Design, Construction, and Assessment of a Self-Sustaining Drainage Ditch

A THESIS

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Abstract

Agricultural drainage is a double-edged sword: helping farmers achieve ever-increasing crop yields to meet consumers' demands, while providing a short-circuit through the soil profile for excess water and nutrients. Drainage ditches are an important pathway as water moves downstream in headwater landscapes. As low order streams, ditches have the potential to remove and assimilate nutrients. In order to operate at their maximum nutrient removal potential, ditches should be healthy, self-sustaining ecosystems that function similarly to natural streams.

The two-stage agricultural drainage ditch is an innovative solution for creating drainage ditches that behave more like natural streams. A low-flow channel provides sediment transport during low-flow periods, while benches within the ditch allow for overbank flow and energy dissipation during high-flow periods. The larger cross-sectional area increases surface contact between water and the ditch at certain flow depths, which likely enhances nutrient removal.

In this study, a two-stage agricultural drainage ditch was designed and then constructed in southern Minnesota, USA in the autumn of 2009. Extensive monitoring of the ditch has been conducted following construction; efforts have focused on establishing an understanding of the geomorphic, water source, and water quality aspects of the ditch. Analysis of field measurements from August 2010 show that between 10 and 15 percent of nitrate N entering the ditch was removed within the

ditch reach. A slight increase in average channel thalweg elevation has been measured, while increased pool-riffle sequencing has also been observed. Channel cross-sectional surveys have showed slight changes in low-flow channel dimensions.

Economic analyses have been performed to measure the feasibility of two-stage ditch construction. There are situations where predicted cost reductions in periodic ditch maintenance provide enough savings to offset two-stage channel construction costs. In other cases, subsidies may be required to economically justify a two-stage system. An analysis was performed to estimate the cost of additional nitrogen (N) removal ($/kg N removed) in two-stage ditches, using increased N removal as a basis for subsidies. Results show situations where N removal costs is less than $3 to $4 kg^{-1} of N removed, which is competitive with other Best Management Practices.

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1 Introduction

Water quality in agricultural watersheds is under greater scrutiny as the landscape and hydrologic pathways are altered to keep up with world food demand. One of the most dramatic landscape changes in the Midwestern United States since the mid-1800s has been the development and proliferation of subsurface drainage and the accompanying open drainage ditches. The drainage water and entrained nutrients are carried downstream to higher order streams, and eventually, some of the nutrients are transported to the Gulf of Mexico and other coastal waters. These coastal waters, including those on the Louisiana shelf near the Mississippi River delta are being depleted of oxygen from excess algae growth and decomposition. The depletion of oxygen has had devastating effects on aquatic life, as well as the economies of coastal communities that rely on commercial fishing. Degraded water quality within low-order tributaries in the upper Mississippi River basin (UMRB) is also becoming an increasing problem.

Two-stage ditches have been installed as a potential solution for improving the quality of drainage ditches areas of the Midwest (namely Ohio, Michigan, and Indiana). The principle of a two-stage ditch is based on creating a more natural, stable channel within a drainage ditch. This approach has shown potential for enhancing nutrient removal and assimilation, increasing stability in ditch systems, and producing a higher-quality stream and riparian environment for many organisms.

The goals of this study were to design, construct, and assess a two-stage ditch in southern Minnesota, USA, a region with intense subsurface tile drainage in the UMRB. The specific objectives are to:

1) Design a self-sustaining, two-stage drainage ditch for a research site located in Mower County;
2) Collect data to assess the stability of the two-stage ditch;
3) Collect data to determine the removal efficiency of nitrate N within the ditch; and
4) Perform an economic analysis to provide a framework for assessing the economic feasibility of two-stage ditch construction on a site-by-site basis.

2 Agricultural Drainage

2.1 History & Importance

For millennia, many societies have worked to improve their agricultural productivity by controlling soil water. van Schilfgaarde (1971) reported traces of agricultural water management in Mesopotamia dating to as early as 7000 BC. There is evidence of extensive drainage ditch systems used by the Greeks and Egyptians dating to 400 BC (Beauchamp, 1987). The Romans filled in the bottoms of open ditches with stones or bundles of brush, which were then covered by the previously excavated material, to create covered ditches, a forerunner of modern subsurface drainage. This form of subsurface drainage was used throughout Europe for many centuries. Modern-day water management systems in Europe are extensive, and include most notably the Netherlands, which has added a considerable amount of farmland over the past several centuries by claiming land below the sea level through a series of dikes and pumping stations.

However, as in ancient Greece and Egypt, much of the earlier agricultural drainage in Europe and the American colonies centered around surface drainage through small trenches or ditches. Beauchamp (1987) reported that this greatly increased the trafficability in fields, but often did not move excess moisture out of the soil profile quickly enough after rain events. Rain events would often lead to prolonged periods with an elevated water table, which caused stress on crops, thereby reducing crop yields.

Beauchamp (1987) also reported that early settlers in New York and New England used subsurface drainage similar to that used in ancient Rome – using stones, brush, and even hollowed-out logs – to created subsurface conduits for the removal of excess water. Gradually, as the United States expanded westward, larger areas of land needed to be drained to provide agriculture land to feed the growing nation. According to Beauchamp (1987):

> "In its natural state, much of the fertile land in northwestern Ohio, northern Indiana, north-central Illinois, north-central Iowa, and southeastern Missouri was either swamp or frequently too wet to farm."

There are inherently two major issues with the Midwest's poorly drained soils that have a negative effect on crop yields: ponded water on the surface of farm fields and elevated water table elevations.

Results reported by Smedema et al. (2004) highlight the reductions in crop yields that can be seen in poorly drained soils. Studies from Hungary, India, Venezuela, and Texas showed that ponded surface water for prolonged periods will generally have a negative effect on crop yields, especially during the spring and summer months. Smedema et al. (2004) also reported on studies from Australia, the United Kingdom, Oregon, and Ohio, which showed relationships between crop yield and water table depth; namely that there is generally an optimal water table depth for a given soil-crop combination, so that the root zone is neither saturated, nor too far above the water table elevation. A similar relationship showed that reductions in crop yield are related to the frequency and severity of events that result in a water table elevation above the optimal water table elevation.

The negative effects of both surface ponding and elevated water tables can be minimized by agricultural drainage. The surfaces of drained fields dry more quickly, enhancing trafficability for large machinery, which can be critical for the timing of planting, harvesting, and other field operations (Herzon and Helenius, 2008; Blann et al., 2009). The risk of soil compaction (and the potential for a corresponding drop in drainage capacity) is also reduced due to improved trafficability (Appelboom and Fouss, 2006; Smith and Pappas, 2007; Blann et al., 2009). Accelerated drainage within the soil column also reduces the amount of time that crop root zones are susceptible to stress due to exposed to high water table elevations and surface ponding (Smedema et al., 2004; Blann et al., 2009). These benefits of subsurface drainage have led to wide-scale implementation throughout the Midwest.

Fouss and Reeve (1987) reported on the recent technological developments in agricultural drainage during late 1900s: “drainage technology changed and modernized more during 1965-1975 than in the previous 100 years.” Major advances include the widespread adoption of plastic drain tile, and the development of machinery that drastically decreased tile installation time. The most recent comprehensive survey of agriculture drainage in the United States was conducted as part of the 1982 U.S. Department of Commerce Census of Agriculture. It was estimated that 24 percent of all cropland in the 48 contiguous states had been improved by some form of agricultural drainage at that point. Statistics on the extent of agricultural drainage in the Midwest are given in Table 1.

Following Table 1, the percentage of cropland that is drained remained approximately constant from 1985 to 1998 throughout much of the Midwest. Growth in drainage was reported in both Iowa and Ohio, while Minnesota doubled the percentage of drained cropland from 20 to 40 percent. The highest concentrations of tile drainage remain in Ohio and Indiana, with 60 and 50 percent of cropland drained, respectively.

Table 1. Percentage of Total Cropland Drained, Midwestern States, 1985 and 1998.

State	1985(1)	1998(2)
Indiana	50	50
Ohio	50	**60**
Illinois	35	35
Michigan	30	30
Iowa	25	**35**
Missouri	25	25
Minnesota	20	**40**
Wisconsin	10	10

(1) adapted from USDA (1987); (2) taken from The Ohio State University Extension Service (1998)

The United States Department of Agriculture (USDA) (1987) estimated the net value of all drainage improvements in the United States accounts for 4 to 6 percent of total value of all farm real estate, while the portion in Iowa and Minnesota was estimated to be 10 percent, and 20 percent in Ohio and Indiana. These figures underscore the economic importance of agricultural drainage. Blann et al. (2009) reported that while the initial investment for subsurface drainage tile is rather high (\$300 to \$600 acre^{-1}, or \$750 to \$1500 ha^{-1}), yield increases in drained fields are on the order of 5-25%.

2.2 From Drainage Tiles to Drainage Ditches

Following the expansion of drainage in the Midwest, headwater streams in many basins have been drastically altered to create drainage ditches for conveying drainage water downstream. Many of these drainage ditches are natural streams, which have been widened, straightened, and deepened (Landwehr and Rhoads, 2003; Powell et al., 2007b; King et al., 2009). Straightening is primarily done to control the natural meanderings of a stream, in order to prevent the loss of cropland, and to provide a more permanent boundary to existing fields. Widening of drainage ditches is generally done to increase the cross-sectional area of the drainage ditch for any given flow depth, thus reducing the potential for overtopping the ditch banks during high flow events.

Taff (1998) reported that 27,000 miles (43,500 km) of the 90,000 miles (145,000 km) of waterways in Minnesota are constructed ditches.

2.2.1 Conventional Drainage Ditch Design

Conventional drainage ditches are constructed as a trapezoidal channel, with a flat bottom and outside banks with side slope ratio generally from 1:1 to 2:1 (horizontal:vertical) (Peterson et al., 2010). A cross-sectional view of this type of ditch is given in Figure 1. A typical conventional drainage ditch (the Mullenbach drainage ditch site, which is the subject of this study) in Minnesota is shown in Figure 2. The depth of a conventional drainage ditch is usually designed to carry a flow with a recurrence interval from 5 to 100 years (Ward and Trimble, 2004).

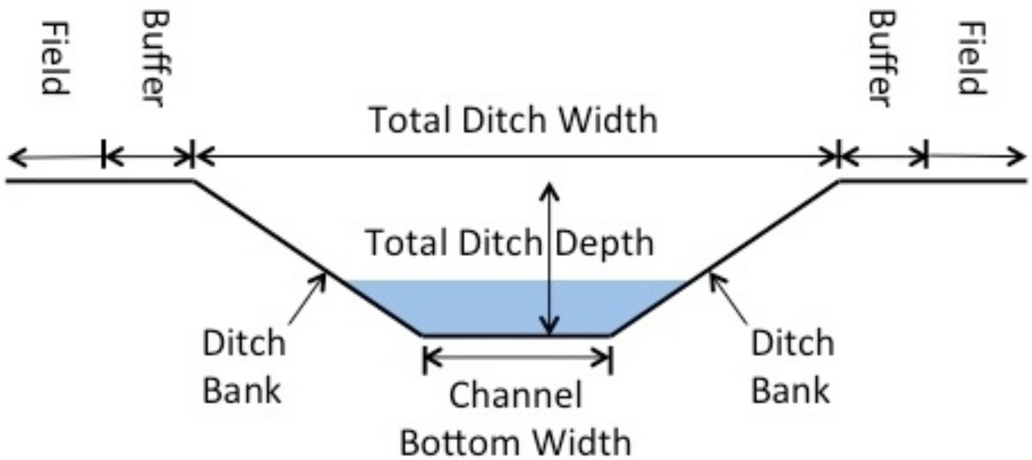

Figure 1. A Typical Conventional (Trapezoidal) Drainage Ditch Cross-Section.

Figure 2. A Typical Agricultural Drainage Ditch in Minnesota – the Mullenbach Drainage Ditch, March 2009. Photo taken by Brad Hansen.

2.2.2 Issues Related to Agricultural Drainage

2.2.2.1 Ditch Stability & Maintenance

The trapezoidal geometry of conventional drainage ditches can result in an over-widened channel bottom (Christner et al., 2004). The wide channel bottom greatly reduces flow velocity in the channel during low flow conditions, which leads to sediment aggradation (Landwehr and Rhoads, 2003; Jayakaran and Ward, 2007). Over long time periods of sediment aggradation and accumulation, a natural low-flow channel will begin to form within the ditch bottom (Christner et al., 2004; Hansen et al., 2006; Jayakaran and Ward, 2007; Rhoads and Massey, 2010). This is part of the ditch's attempt to reach dynamic equilibrium (Yang 1971) by creating a channel cross-section that naturally maintains its dimension and profile over time. This will allow the channel to move both the sediment and water loads provided by its watershed.

Nevertheless, due to limited stream power or excess sediment supply, sediment aggradation will occur on the channel bed. The agricultural community views this sediment deposition as bad for drainage capacity, and therefore costly ditch maintenance (cleanout) is performed periodically to maintain the desired drainage capacity (Christner et al., 2004; Jayakaran and Ward, 2007; Peterson et al., 2010). Ditch cleanout generally involves using heavy machinery (i.e. trackhoes or excavators) to remove deposited material from the ditch channel bottom, and is usually performed with the goal of reestablishing the designed trapezoidal channel pattern to increase drainage capacity (Peterson et al., 2010).

When cleanouts and drainage ditch maintenance are performed, the natural progression toward dynamic equilibrium is interrupted, and indeed, reversed (Magner et al., in review). After deposited sediment is removed and the channel shape is again trapezoidal, the ditch will again begin its movement toward dynamic equilibrium, where sediment supply is in balance with sediment transport capacity. This process will undoubtedly continue until again there is a perceived reduction in water transport capacity, when the ditch will be cleaned out once more. Thus, the cycle of sediment cleanout and aggradation will continue.

Ditch cleanout disrupts the ditch ecology (Kallio et al., 2010) by removing some or all of the existing vegetation and exposing bare soil. Drainage ditch cleanout and maintenance is also expensive. Various estimates of maintenance costs have been reported in the literature:

- $12,000,000 *annual* cost for ditch maintenance and repairs in Minnesota (Hansen et al., 2006),
- \$450 mi^{-1} y^{-1} in Ohio (Hansen et al., 2006),
- $650,000 in 2005, and $1,200,000 in 2006 on open ditch maintenance in Blue Earth County, Minnesota (Hansen et al., 2006), a county that has 163 miles of open county ditches (Blue Earth County, 2008),
- \$65 to \$115 linear mi^{-1} in *annual* ditch maintenance costs in Chippewa County, Minnesota (Christner et al., 2004), and

- \$950 to \$22,000 mi^{-1} for ditch cleanout and maintenance activities (survey of four drainage authorities in Minnesota) (Peterson et al., 2010)

While the above figures show that the cost of drainage ditch maintenance and cleanout can vary greatly, it is clear that maintenance costs have the potential to be costly for landowners, drainage associations, and local governments.

The frequency of ditch cleanout can vary widely from ditch to ditch, and depends on many factors, including: channel bed and bank substrate, hydrologic regime, and channel slope. Some drainage ditches can require cleanouts more often than every 10 years, while other ditches may reach a relatively stable form that is essentially self-sustaining (Christner et al., 2004).

Other problems can arise with the trapezoidal design, primarily concerning the flat bottom of the ditch, which results in the outer ditch banks being continuously exposed to water (Ward et al., 2004). This continuous exposure leads to reduced soil strength along the toe of the ditch bank (bank/bed interface) and can cause mass wasting and bank sloughing (Ward et al., 2004). Shear forces due to flowing water are continuously acting on the ditch bank, which leaves the ditch bank vulnerable to erosion (Ward et al., 2004; Magner et al., in review).

2.2.2.2 Loss of Wetlands

One of the major impacts of the proliferation of tile drainage in the upper Midwest has been the large loss of wetlands across the landscape (Mitsch et al., 2001; Zedler, 2003). In the Midwestern states, 80 percent of wetland acreage has been drained (Mitsch et al., 2001). Mitsch et al. (2001) reported the loss of 14.1 million ha (34.8 million acres) throughout seven Midwestern states (Indiana, Illinois, Iowa, Minnesota, Missouri, Ohio, and Wisconsin) between the 1780s and the 1980s. Zedler (2003) reported similar drastic reductions in wetland area from the 1770s to 1980; Minnesota alone lost nearly 2.6 million ha (6.4 million acres) of wetlands (42 percent of its wetlands). Illinois, Iowa, and Missouri saw much higher losses of wetlands over the same time period: 85, 89, and 87 percent reductions, respectively.

Zedler (2003) summarized two key ecosystem benefits lost with the removal of wetlands: flood reduction and water quality improvement. Flood reduction benefits (on both small and large scales) are due to the capacity of wetlands to temporarily store runoff after a rainfall event. One water quality benefit of wetlands is the high denitrification potential (Mitsch et al., 2001). Reddy et al. (1999) provided a summary of the mechanisms that wetlands employ to retain phosphorus (P); wetlands also have the capability to transform P between bio-available and -unavailable forms. The amount of bio-available P is often found to be a limiting nutrient in freshwater systems (Reddy et al., 1999). The removal of wetlands can therefore increase the amount of bio-available P in freshwater ecosystems.

The conversion of these millions of acres from wetlands to agricultural land has increased the amount of land in agricultural production, while simultaneously reducing the landscape's ability to buffer storm flows and pollutants. This conversion of wetlands to agricultural land has also effectively increased the contributing watershed area for downstream channels (Magner et al., 1993).

2.2.2.3 Nitrate N Loading

Increased eutrophication in various coastal waters around the world in the past several decades has been widely reported (Rabalais et al., 1998; Rabalais et al., 2001; Rabalais et al., 2010; Osterman et al., 2005). Eutrophication is the presence of excess nutrients in water, which leads to hypoxia, or low dissolved oxygen levels (less than 2 mg L^{-1}). Low dissolved oxygen levels can have devastating effects on fish and other aquatic species present in the water. Of particular interest in the United States is the seasonal hypoxic zone that has been measured on the Louisiana continental shelf in the Gulf of Mexico near the delta of the Mississippi – Atchafalaya River Basin (MARB). Seasonal hypoxia occurs from as early as late February through as late as early October, but is generally at its largest extent in June, July, and August.

Osterman et al. (2005) reported that there were several low-oxygen events on the Louisiana shelf between the years of 1817 and 1910 (prior to wide-scale use of fertilizer within the MARB), and that those events were coincident with above-average discharges from the MARB. This evidence suggests there were natural conditions before 1910 that led to hypoxia on the Louisiana shelf. The authors reported that the pre-1910 events appeared to be less severe than the hypoxia that has been recorded over the past several decades.

The hypoxic zone is formed when the rate of oxygen consumed for the aerobic decomposition of organic carbon exceeds that of oxygen re-aeration (Rabalais et al., 2010). Increases in nutrient concentrations and loading within the MARB have been linked to an increase in organic carbon on the Louisiana shelf (Rabalais et al., 1998; Rabalais et al., 2001), and have been shown to cause a corresponding increase in extent and severity of the Louisiana shelf hypoxic zone (Rabalais et al., 2001; Rabalais et al., 2010; Osterman et al., 2005). Turner and Rabalais (1991) reported that there have been significant increases in riverine nitrate N (NO_3-N) and P concentrations and loadings since the early 1900s. Mitsch et al. (2001) reported that nitrogen (N) is most often the limiting macronutrient for productivity in estuary and coastal waters, and that increases in nitrate N concentrations and fluxes in the MARB increased dramatically after 1950, when N fertilizer became widely used. Rabalais et al. (2010) predicted further increases in nutrient loading due to the strain on finite resources from increasing population, increased dependence on crops that require high fertilizer inputs, and urbanization and related effects.

Nitrate N can cause eutrophication in surface water by stimulating algae production (Randall and Mulla, 2001). According to Randall and Mulla:

> "In a soil system, nitrate N is continually supplied through the natural processes of mineralization and nitrification of soil organic matter. Other sources of N include fertilizers, animal manures, municipal sewage waste, agricultural and industrial wastes, atmospheric deposition, and dinitrogen fixation, all of which can be converted to nitrate N through mineralization and nitrification. Nitrate N is mobile and, therefore, can be lost from the soil profile by leaching. Subsequent transport of nitrate N to surface waters occurs through subsurface drainage (tile lines) or base flow."

The mobility of nitrate N within the soil leads to the potential for large nitrate N losses in watersheds with high N inputs. Agricultural watersheds are at a particularly increased risk of nitrate N loss, due to N inputs from fertilizer and animal manure that are employed to enhance crop growth.

The increase in nitrate N loading in the MARB has played a part in the development of the hypoxic zone in the northern Gulf of Mexico (Rabalais et al., 1996; Alexander et al., 2008). The United States Environmental Protection Agency (EPA) Science Advisory Board (2007) noted that a large portion, 82 percent, of nitrate N flux in the MARB originates in the Upper Mississippi and Ohio-Tennessee River basins, while those areas account for just 31 percent of area in the MARB. These areas are the most intensively drained areas in the MARB. Skaggs et al. (2005) reported a large increase in nitrate N losses from tile-drained agricultural fields as the lateral tile spacing decreases.

A nationwide study (Omernik, 1977) of total N (TN) and total phosphorus (TP) concentrations in 928 watersheds showed that the mean TN concentration was 5.345 mg/L in 74 watersheds where agriculture accounted for more than 90% of land use. The 68 watersheds where forest made up more than 90% of the land cover had a mean TN concentration of 0.598 mg/L, or about one order of magnitude lower than that of the predominantly agricultural catchments.

Goolsby et al. (2000) developed a model to determine annual TN yields per area for 42 basins within the MARB; the 42 basins modeled constituted approximately 70 percent of the MARB. The eleven basins with the highest TN loading per area (1501 to 3120 kg km^{-2} y^{-1} (8570 to 17800 lbs mi^{-2} y^{-1})) are all located in the Midwest, and stretch from central Ohio to southwestern Minnesota. David et al. (2010) published similar findings; their predictions of average riverine nitrate N yield (January – June) show the highest losses occurred in an area from southwestern Minnesota into central Iowa, and eastward through Illinois and Indiana into central Ohio. Predicted average riverine nitrate N yields were between 15.01 and 25.00 kg nitrate N ha^{-1} y^{-1} (13.4 to 22.3 lbs nitrate N $acre^{-1}$ y^{-1}) in a large number of counties in the Midwest. In contrast, the authors reported average nitrate N losses of 0.8 and 1.1 kg ha^{-1} y^{-1} (0.7 to 1.0 lbs $acre^{-1}$ y^{-1}) for the Missouri and Lower Mississippi sub-basins, respectively.

Alexander et al. (2000) showed that the N removal capability of streams is highly inversely proportional to the stream size. They reported that differences in N loss coefficients span nearly two orders of magnitude, from 0.005 day^{-1} in the Mississippi River to 0.45 day^{-1} in small streams in the MARB. This evidence, coupled with the evidence that tile-drained agricultural landscapes are contributing disproportionately to excess nitrate N loading in the MARB, underscores the importance of establishing efficient nitrate N removal systems in the small headwater streams of the MARB.

2.2.2.4 Effects on Habitat

Small headwater streams provide an important habitat for a wide variety of biological organisms (King et al., 2009). Streams with greater variability in form and channel bed substrates (i.e. high quality pool-riffle sequences) provide better habitat for aquatic organisms (Christner et al., 2004). Trapezoidal drainage ditch design is dissimilar to natural channel design, and creates an unstable stream system (Christner et al., 2004). Habitat degradation is often closely linked to instability in stream systems (Shields et al., 2003).

The fish index of biotic integrity (IBI) was introduced by Karr (1981), and has proven to be a useful tool for assessing not only the health of organisms within streams, but also as a surrogate for assessing the health of the streams themselves. Karr et al. (1986) reported that negative habitat quality changes due to anthropogenic causes include both streambed substrate instability and more uniform water depth. Smiley et al. (2008) found that fish response variables (measures of diversity, abundance, and species composition in fish communities) in agricultural drainage ditches in Ohio were most strongly influenced by an ‘instream habitat’ metric. The metric was based on herbaceous vegetation within the ditch, mean water depth, and mean wet width. The authors also reported that instream habitat had a greater effect on fish communities than riparian habitat or water chemistry.

Christner et al. (2004) reported on channel bed substrate for two separate reaches of JD #8 in Swift County, Minnesota: a trapezoidal reach (essentially a conventional drainage ditch) and a natural reach. The natural reach was an 800 m section of ditch that had been over-widened during ditch cleanout, and had formed a stable channel geometry, with a low-flow channel and connectivity to a small floodplain within the ditch system. The authors reported that the natural reach had coarser bed material than the trapezoidal reach; the D_{90} was 32 mm (1.26 inches) in the natural reach as opposed to 2 mm (0.08 inches) for the trapezoidal reach. Additionally, particles of 1 mm (0.04 inches) or smaller diameter accounted for 72 percent of bed substrate in the trapezoidal reach, compared to just 37 percent for the natural reach. Increased channel substrate variability (such as that measured in the natural reach of JD #8) results in better aquatic habitat (Christner et al., 2004) due to well-aerated coarse material in riffle sequences. Furthermore, fish habitat availability in streams is indicated by the maximum pool depth (Christner et al. 2004). The poor sediment transport afforded by a wide trapezoidal drainage ditch during low-flow periods leads to poor sediment sorting (Christner et al. 2004); in this case the pool-riffle sequencing that provides high-quality aquatic habitat will be difficult to achieve, and the more uniform water depth that Karr et al. (1986) discussed will tend to occur. Sites with coarse substrate therefore have the potential to provide enhanced fish habitat, which will likely lead to increased fish IBI scores.

2.3 The Two-Stage Solution

Two-stage drainage ditches are low-order streams that are designed to mimic the stable conditions found in natural low-order streams (Ward et al. 2004; USDA-NRCS 2007). Two-stage ditches are usually constructed to replace conventional ditches, and are most beneficial at sites where present conventional drainage ditches are unstable. Two-stage ditches have also been shown to form naturally in over-widened trapezoidal ditch systems, as the likely result of a conventional ditch’s attempt to recover to a stable condition (Landwehr and Rhoads, 2003; Christner et al., 2004; Ward et al., 2004; Hansen et al., 2006).

A natural channel will size its stream geometry and features based on the water and sediment loads delivered to it by its watershed. The channel achieves this state by providing a small low-flow (or bankfull) channel, which can provide sediment transport under low flow conditions, as a result of relatively high velocities achieved due to a small cross-sectional area. Frequent overbank flooding is an important process in natural, low-gradient streams (Ward and Trimble, 2004).

Two-stage ditches are designed fundamentally differently than conventional drainage ditches. Where conventional ditches are over-widened for low-flow conditions, two-stage systems are designed to provide channel features that more closely resemble those found in natural streams. A low-flow channel is sized to replicate that of a natural channel (in the surrounding region) with similar drainage area. Where conventional ditches lack floodplain connectivity, two-stage channels are designed with small benches on both sides of the low-flow channel. The benches allow for dissipation of the fluvial energy associated with high flow rates (Ward et al., 2004). To summarize, a two-stage drainage ditch is constructed to resemble a low-order natural stream system, which will allow the two-stage ditch to behave more like a natural, self-sustaining stream.

The bankfull elevation in natural streams has been shown to correspond to a return period of 1 to 2 years (Shields et al., 2003). Jayakaran and Ward (2007) found that the recurrence interval for bankfull flow in streams in western Ohio that had naturally developed a two-stage channel form could be as low as 0.2 years.

A schematic of a two-stage ditch is given in Figure 3. The sides of the low-flow channel are typically constructed at a slope of 1:1 in cohesive soils, and a slope of 2:1 (horizontal:vertical) is generally used for the outside ditch banks (Ward et al., 2004).

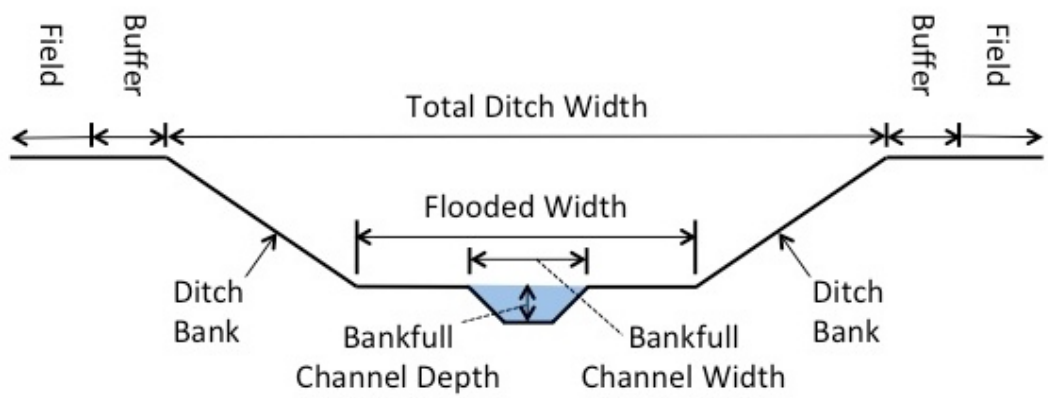

Figure 3. A Two-Stage Drainage Ditch Cross-Section.

2.3.1 Benefits of Two-Stage Ditches

The main value of a two-stage ditch is increased stability in the ditch system. This is mainly embodied in reduced ditch cleanout costs associated with sediment deposition (Ward et al., 2004; Powell et al., 2007a; Peterson et al., 2010). Additionally, a reduction in erosive forces at the toe of the outer ditch bank is expected, as water will usually be confined to the low-flow channel (Ward et al., 2004). Increased stability at the toe of the ditch will also likely reduce the potential for sloughing and mass wasting on the outside ditch banks.

In addition to increased stability, a two-stage channel can potentially provide other benefits to the landscape. There is a potential for enhanced habitat within the low-flow channel, due to deeper water during low-flow periods (vs. conventional ditches) (Ward et al., 2004; Powell et al., 2007a). It should be easier to maintain vegetative cover in a two-stage ditch, due to fewer cleanouts and more stable bench material for plant growth. This vegetative cover not only increases nutrient uptake, specifically nitrate N, but may also provide more shade for fish in the narrower low-flow channel (Ward et al., 2004; Powell et al., 2007a). Improved sediment sorting, with fine materials either carried away by the low-flow channel or deposited on the benches, and coarse materials allowed to settle to the channel bed, may also lead to improved fish habitat (Ward and Trimble, 2004; Ward et al. 2004; Powell et al., 2007a).

Powell and Bouchard (2010) also showed that while the denitrification rates measured in sediment from the channels of trapezoidal and two-stage ditches did not differ significantly, denitrification rates in sediments taken on the benches in two-stage ditches were higher than those in sediments taken from the outside banks of trapezoidal ditches. The authors further showed that denitrification rates on the outside banks of trapezoidal ditches were limited by both nitrate N and carbon concentrations, while denitrification rates on the benches of two-stage ditches were limited only by nitrate N concentration; this difference is due to accumulation of organic matter on the benches of two-stage ditches.

2.3.2 Costs of Two-Stage Ditches

There are two main costs associated with two-stage ditches. Firstly, the earthwork and excavation costs of creating a wider overall ditch system are substantial (Ward et al., 2004). With just a 20-ft (6.1 m) wider ditch system (10 ft or 3.05 m on each bank), there may be upwards of 20,000 cubic yards (15,300 m^3) excavated mi^{-1} (per 1.6 km) of ditch constructed. At a cost of \$1.65 yd^{-3} (\$2.16 m^{-3}), this would cost \$33,000 for excavation alone. Secondly, the cost of the land taken out of production by the widening of the ditch may be considerable: given the same 20-ft (6.1 m) expansion of overall ditch width, the amount of agricultural land taken out of production would be just over 2 acres mi^{-1} (1.29 ha km^{-1}) of ditch constructed. However, improved drainage of wet fields, due to the increased capacity of a two-stage ditch, may offset the cost of any agricultural land that is taken out of production.

A detailed breakdown of the economics of constructing a two-stage drainage ditch is included in Section 5.

2.3.3 Existing Two-Stage Ditches

Kallio et al. (2010) reported on the long-term stability of five two-stage drainage ditches constructed in Indiana, Michigan, and Ohio, USA. The ditches studied were all constructed between 2001 and 2007, and resurveyed in 2009 to measure changes in low-flow channel mean depth, width, and cross-sectional area. Their results showed that all five constructed two-stage ditches were found to be in a stable condition in 2009, and all have required little to no maintenance following construction. The authors did note that some of the surveyed reaches showed sediment aggradation, but it was unclear if the aggradation was due to a flaw in the two-stage design or was simply the result of natural variations in hydrology and sediment transport.

Powell et al. (2007b) reported that the authors have been involved in the design of eight ditches in Indiana, Michigan, and Ohio (some of which overlap with the five reported by Kallio et al. (2010)). The authors stated that ditch stakeholders' satisfaction was high in the years following construction and that few problems had been reported.

3 Mullenbach Two-Stage Drainage Ditch Site

3.1 Watershed Description

The Mullenbach two-stage drainage ditch site (Figure 4 and Figure 5) is located in rural Mower County in southern Minnesota (MN), USA, approximately 8 km southwest of the town of Adams, MN, and 10 km south-southeast of the town of Rose Creek, MN. The ditch is located in the headwaters of the Little Cedar River within the Upper Cedar River watershed (8-digit HUC: 07080201). The Mullenbach ditch empties into the Little Cedar River approximately 4 km downstream of the constructed two-stage ditch reach.

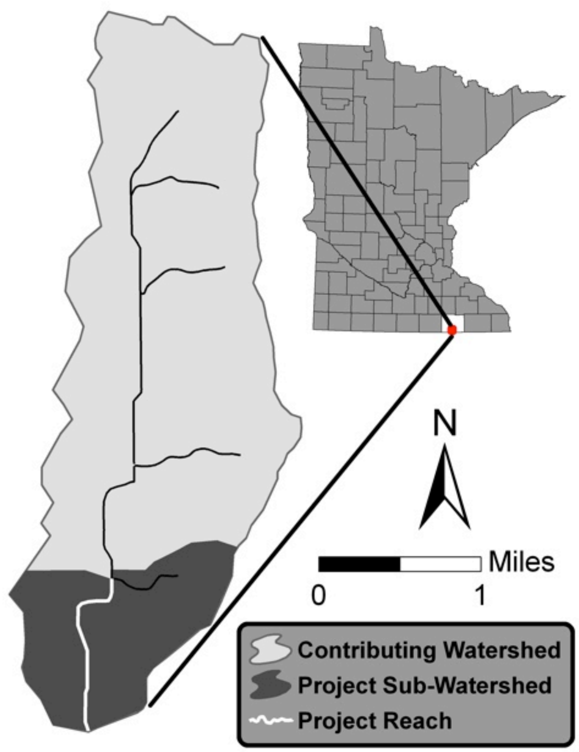

Figure 4. Location of the Mullenbach Drainage Ditch Watershed in Mower County, Minnesota. Map by Joel Peterson and Taken with Permission from Peterson et al., (2010).

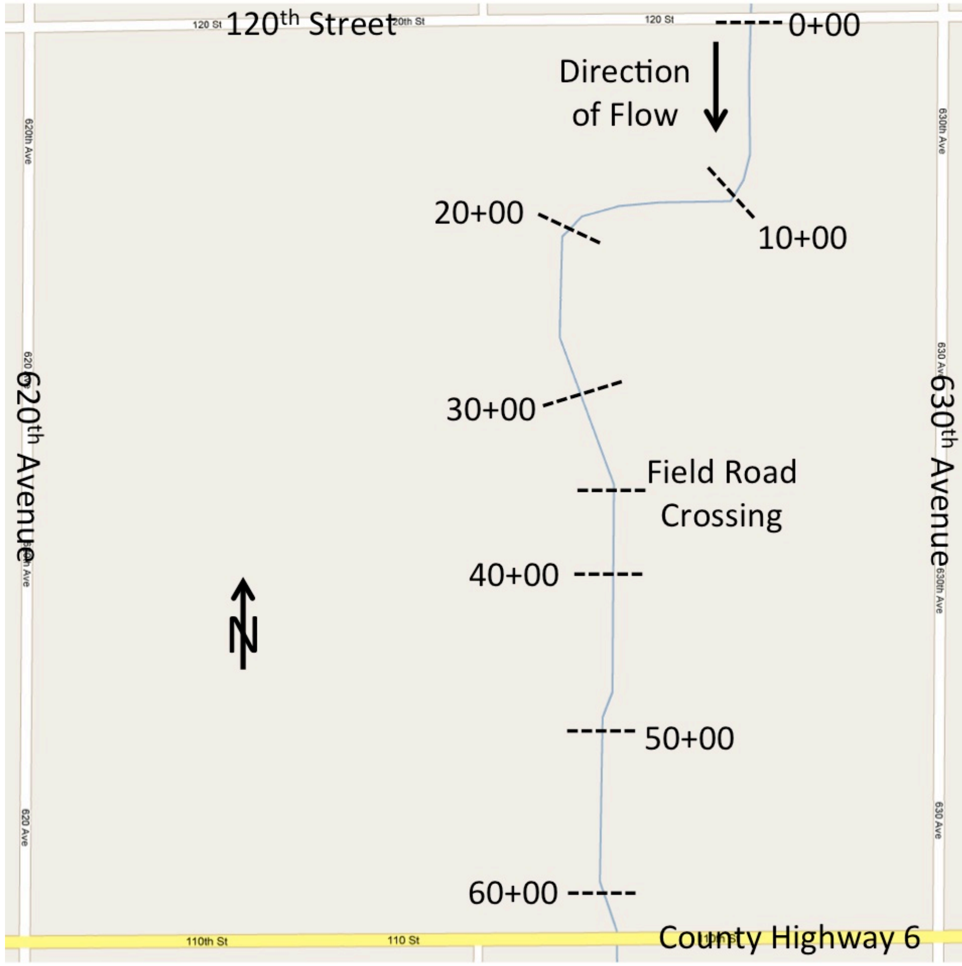

Figure 5. Mullenbach Drainage Ditch Site in Southern Mower County, Minnesota. Base Map © Google.

Peterson et al. (2010) described the project watershed as follows:

> "The project site is located… in the Western Lake section of the Central Lowland physiographic province. The topography is nearly level. Soils in the area comprise mainly poorly drained silty sediment over glacial till and outwash, deposited most likely prior to the Illinoian glaciation. Total annual average precipitation in this region is 80 cm (31.5 in). The watershed area is 12.6 km^2 (3,102 acres). Land use is predominantly row crop agriculture, the main crops being corn and soybeans… The dominant soil type in the ditch is a Clyde silty loam clay (Fine-loamy, mixed, mesic Typic Haplaquolls). The Clyde series developed in shallow depressions and drainageways and is poorly drained with moderate permeability. As

> indicated in the soil survey and evidenced in the field, this soil is typified by bands of pebbles and other coarse material. These bands of coarse material act as conduits, conveying water to the ditch."

The Mullenbach drainage ditch was selected for conversion to a two-stage ditch for a number of reasons. First, the existing trapezoidal ditch was scheduled for maintenance during 2009 or 2010 due to several instabilities within the ditch system: 1) bank instability caused by seepage; 2) planar failure and toe erosion in ditch banks; and 3) tile outlet failures (Peterson et al., 2010). The cost of earthwork for two-stage construction is easier to justify at a site that already requires costly ditch maintenance. A more thorough discussion of the economic factors involved in two-stage drainage ditch construction is given in Section 5. Second, The Nature Conservancy and the Mower County Soil and Water Conservation District identified the Mullenbach site as one where the landowners surrounding the ditch were enthusiastic about implementing the project. Finally, the approval process for changes to this non-public ditch is simpler than work done on a public ditch system.

3.2 Original Conventional Drainage Ditch

Pictures of the pre-existing Mullenbach trapezoidal ditch are included in the Appendix (Figure 37- Figure 42). These pictures highlight issues related to the ditch condition prior to the construction of the two-stage channel.

3.3 Mullenbach Two-Stage Drainage Ditch Design

Ward et al. (2004) and Powell et al. (2007a) suggest that the ratio of the flooded width to the low-flow channel be at least 3:1. Thus, the width of each bench should be at least as great as that of the low-flow channel width. Ratios lower than 3:1 may lead to benches that are not fully developed (although this has not been tested), or have an increased likelihood of being unstable (Ward et al., 2004). Current work (Magner et al., in review) suggests that there are situations where stable fluvial systems can develop with relatively small bench features. Ward et al. (2004) also suggested that the overall width should be no more than 5:1 as ratios greater than 5:1 will tend to create a system that is more likely to meander when overbank flow is initiated. However, Christner et al. (2004) showed evidence of a stable meander pattern in a Minnesota ditch of ration 4:1. This may be a consequence of the differences in site conditions.

Determination of the proper ratio of flooded width to low-flow channel width requires some consideration. The main benefits of a larger ratio are 1) to increase the buffer distance between agricultural fields and the low-flow channel and 2) to increase the channel cross-sectional area, thereby providing more storage within the channel and reducing the risk of flow overtopping onto adjacent fields during periods of high streamflow. The main benefit of a lower ratio is a reduction in the amount of agricultural land that is taken out of production, which results in 1) a reduction in earthmoving costs during two-stage channel construction and 2) a reduction in possible compensation to land owners for agricultural land removed from production. At the Mullenbach site, the minimum recommended ratio of 3:1 was used, in order to reduce excavation costs and reduce the impact on adjacent agricultural fields.

The outside ditch bank slope was designed at 2:1 (horizontal:vertical), and the low-flow channel banks were designed at 1:1, following recommendations of Ward et al. (2004) and Powell et al. (2007a).

Two-stage channel design was performed using a spreadsheet developed by Mecklenburg and Ward (2004). The low-flow channel cross-section was dimensioned according to regional curves previously developed by Joseph Magner and Ken Brooks. The curves used in this study were developed for riffle sections in south-central Minnesota streams, and come from unpublished data (J.A. Magner, personal communication, 2009) that was collected in a manner similar to other work done by Magner and Brooks (see Magner and Brooks, 2007) in Minnesota. Measurements of bankfull channel width and bankfull channel depth were made at riffle sections of varying drainage areas; bankfull cross-sectional area was then computed for each site. Each of the three channel dimensions (bankfull width, bankfull depth, and bankfull cross-sectional area) was then plotted against drainage area to create regional curves. The resulting channel dimensions obtained from the regional curve are reported in Table 2 as the design dimension.

The design low-flow channel width was often more narrow, and the design depth deeper, than the existing pre-construction cross-section dimensions. It was deemed beneficial to leave the streambed largely intact during construction, which offered several advantages, including: 1) a reduction in excavation costs; 2) allowing the existing benthic ecosystem to remain intact; and 3) allowing existing vegetation in the channel and on the edges of the channel to remain, and thereby stabilize the banks.

In order to satisfy the design low-flow channel cross-sectional area, while at the same time leaving the streambed intact, two options remained: 1) leave the channel width as is and reduce the channel depth to attempt to reach the design low-flow channel cross-sectional area, or 2) attempt to narrow the channel in order to achieve the design low-flow channel width. The decision was made to take two different approaches with separate reaches within the ditch to address the over-widened low-flow channel: one reach (where the low-flow channel width was just slightly larger than the design width) would be left as is, and in two other reaches (where the channel width was much greater than the designed channel width), spoil material from excavation would be added to the low-flow channel sides and packed to extend the bench width and thereby reduce the low-flow channel width. Where the low-flow channel was left intact (wider than the design width), we hypothesized that sediment transport would naturally effectively narrow the channel through the development of an inner berm; a phenomenon noted in natural streams.

The adjustments in design low-flow dimensions show the difficulty of constructing a two-stage ditch system within an active ditch system. The design and adjusted channel dimensions are given in Table 2. A typical cross-sectional view is shown in Figure 6.

Table 2. Design and Constructed Channel Dimensions.

Channel Feature	Designed, ft (m)	Constructed, ft (m)
Low-flow channel depth	1.88 (0.57)	Approximately 2.0 (0.61) (varies with changes in bed)
Low-flow channel top width	10.75 (3.28)	9 to 11 (2.74 to 3.35)
Low-flow channel cross-sectional area	19.89 (6.06)	Varies throughout reach
Bench width	10.75 (3.28)	9 to 11 (2.74 to 3.35)
Flooded width	32.25 (9.83)	32 (9.75)
Overall ditch top width (approximate; depends on outside bank height)	63 (19.2)	63 (19.2)

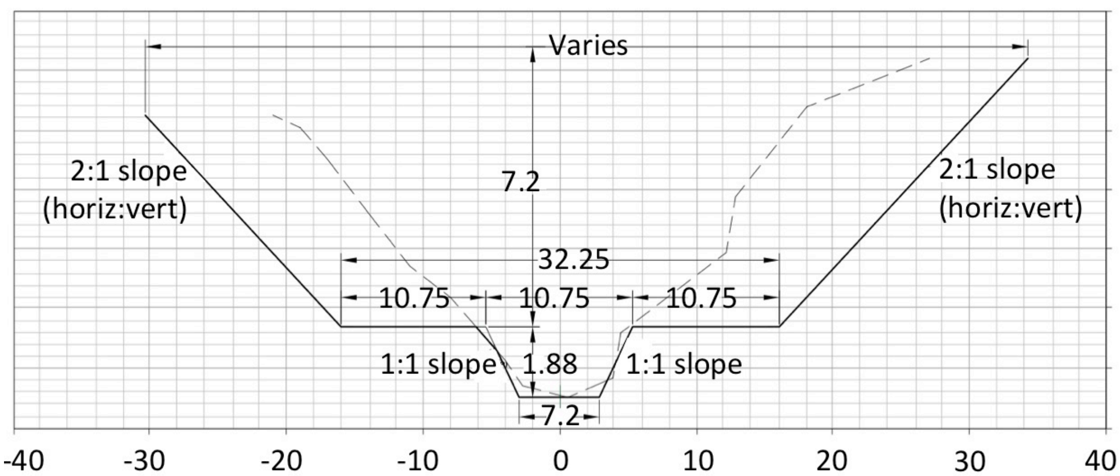

Figure 6. Typical Cross-Section Showing Pre-existing and Two-Stage Channel Design Dimensions. The Existing (Before Construction) Channel Cross-Section is Shown by the Dashed Line. All Dimensions are in Feet. Drawing by Joel Peterson.

3.4 Two-Stage Ditch Construction

Two-stage ditch construction began in late September, 2009. Construction was funded by The Nature Conservancy, and performed by Freeborn Construction of Albert Lea, MN. Excavation began at the north end (0+00) of the ditch site. Machinery was used to create level ditch benches following excavation. Bench elevations and slopes were dictated by channel bed surveys taken at the site before and during construction; slopes were adjusted for every 100-ft (30.5 m) interval to ensure that the constructed low-flow channel depth matched the design low-flow channel depth as closely as possible along the entire ditch reach. As mentioned above, soil was added to the low-flow channel to narrow the channel in two reaches: from 0+00 to 7+00 and from 39+00 to 61+00. In the reach from 0+00 to 7+00, soil was added on the right (west) bench of the ditch only. In the reach from 39+00 to 61+00, soil was added to both benches to narrow the low-flow channel. The benches in the reach from 0+00 to 7+00 were constructed in outwash sand and gravel. Thus, vegetation will play a crucial role in providing stability within the low-flow channel, due to the intrinsic lack of shear resistance in the existing material.

As discussed previously, the original design dimensions were altered to fit within the constraints of the existing ditch dimensions. Natural variation in streambed elevation meant that the actual low-flow channel depth varied around the constructed depth of 2 ft (0.61 m), and channel width varied as previously discussed. There is inherent variability (across sites within the same region) in channel dimensions that regional curves must ignore in order to simplify the stream channel dimensions for a large area. The channel dimensions generated by the regional curves should be considered only an estimate for the actual low-flow channel dimensions. The ditch should be allowed to modify its dimensions by natural fluvial processes to achieve its proper low-flow channel dimensions. Altogether, the constructed low-flow channel closely matched the dimensions suggested by the regional curves.

Construction was completed in early November 2009. Pictures from the two-stage construction are included in the Appendix (Figure 43 - Figure 56).

3.5 Channel Structures

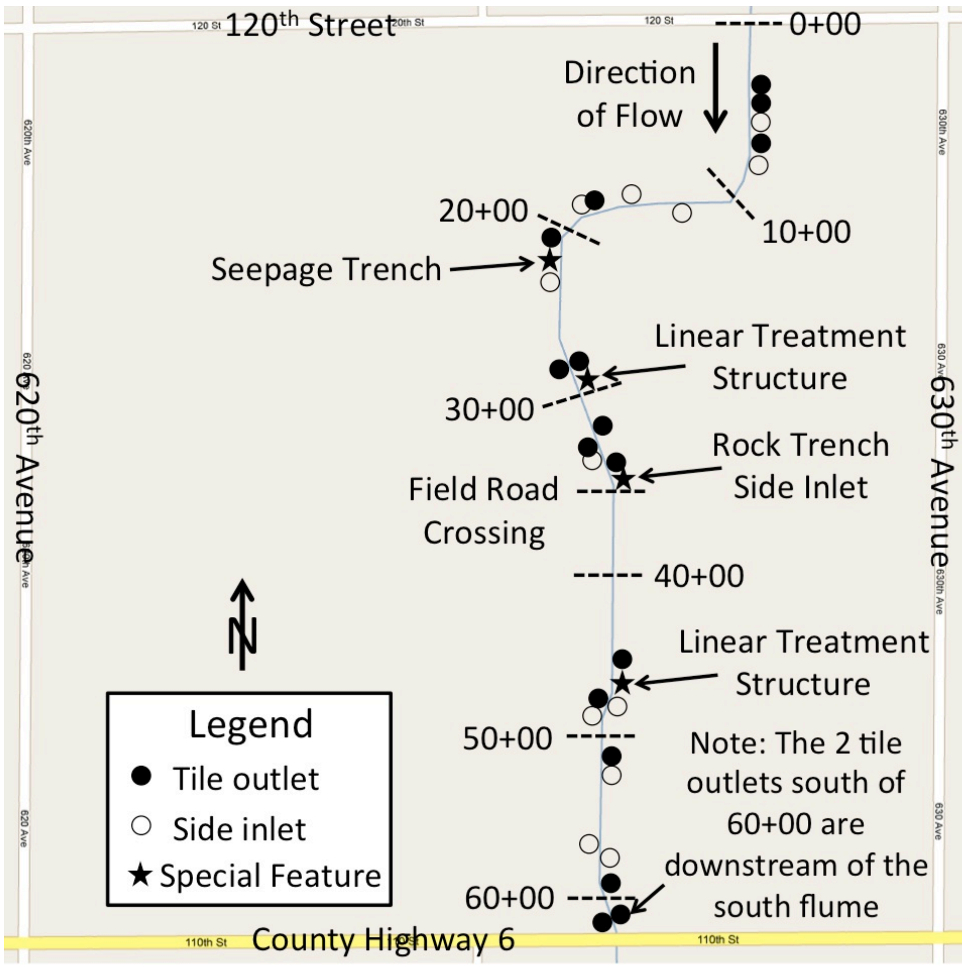

Figure 7. Map of the Mullenbach Two-Stage Drainage Ditch Site Showing the Locations of Ditch Structures. Markers Such As 20+00 Indicate the Distance (in feet) Downstream from the North End of the Project Reach, and Special Features are Labeled within the Map. Base Map © Google.

3.5.1 Tile Outlets

There are 16 subsurface tile drainage outlets that discharge into the ditch reach between 120th Street and Mower County Highway 6. Figure 7 shows the location of each tile outlet within the project reach. As shown in Figure 8, each of the tile outlets was cut off and trimmed to the new ditch bank, and riprap reinforcements were added to protect the ditch benches as the drainage water moves from the tile outlet across the bench into the low-flow channel (see Figure 49 in the Appendix for a photograph of a completed tile outlet). In an effort to improve the nutrient removal capabilities of the two-stage ditch system, two of the 16 tile outlets were constructed with linear treatment systems (see Section 3.5.3 for more information).

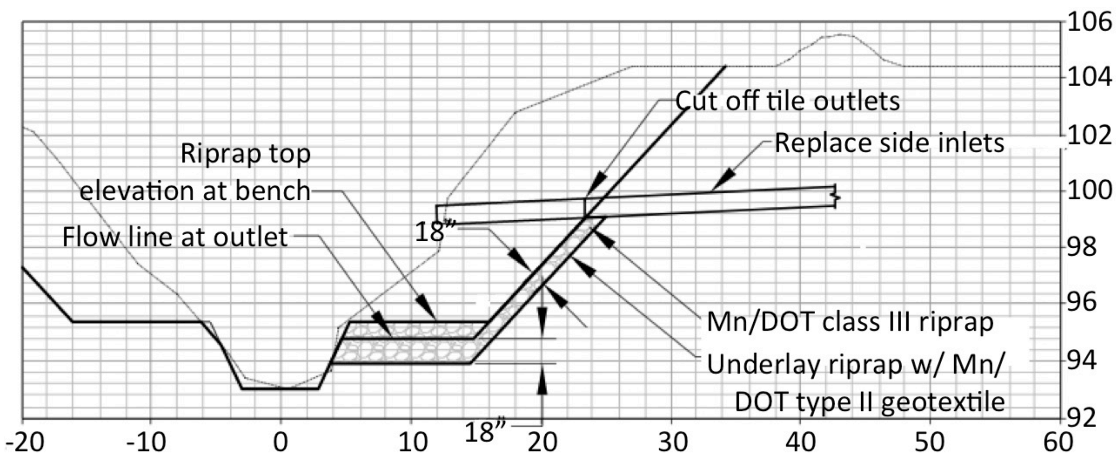

Figure 8. Schematic Showing the Construction Modifications to Each Tile Outlet. Dimensions are in Feet and Inches. Drawing by Joel Peterson.

The two most southerly tile outlets (61+00 and 61+05) are located just upstream of Mower County Highway 6, where the two-stage channel transitions to a conventional ditch in before flowing through the double box culvert that is located under Mower County Highway 6. The water balance and nitrate N removal calculations that are discussed later in this paper will not include the outflow from the tiles at 61+00 and 61+05, as the flow from those tile lines entered the channel downstream of the south flume where in-stream discharge is measured.

3.5.2 Side Inlets

There are 12 side inlets, which are designed to convey surface water from the surrounding landscape into the ditch, in the study reach. All of the side inlets were replaced during construction, and each was sized according to the area from which it receives surface runoff. Riprap was also used to protect the ditch benches from water discharging into the ditch from side inlets. A view of two completed side inlets (before riprap reinforcement was placed) can be seen in Figure 47 in the Appendix. The locations of all side inlets are shown in Figure 7.

3.5.3 Linear Treatment Systems

Two experimental linear treatment systems were constructed within the ditch as a water quality improvement measure. Each linear treatment structure was constructed at a tile outlet. Standard tile outlets discharge across riprap armoring on the ditch banks and into the low-flow channel, while the linear treatment structures route tile discharge water through a 200-ft (61 m) long constructed channel along the outside of the ditch bench (adjacent to the outside ditch bank). A detailed view of a tile outlet entering a linear treatment structure is shown in Figure 9. After flowing through the constructed treatment system for 200 ft (61 m), the channel is routed across the bench toward the low-flow channel, through a reinforced riprap section, where the water empties into the low-flow channel. The intent of the linear treatment structures is to enhance nutrient reduction in tile outlet water by increasing residence time and water-plant contact. Nutrient removal may be enhanced by the introduction of wetland-type species that can tolerate continuous saturation. The locations of the two linear treatment systems are shown in Figure 7. A picture of the constructed linear treatment system at the tile outlet at 28+00 is in included in the appendix (Figure 48).

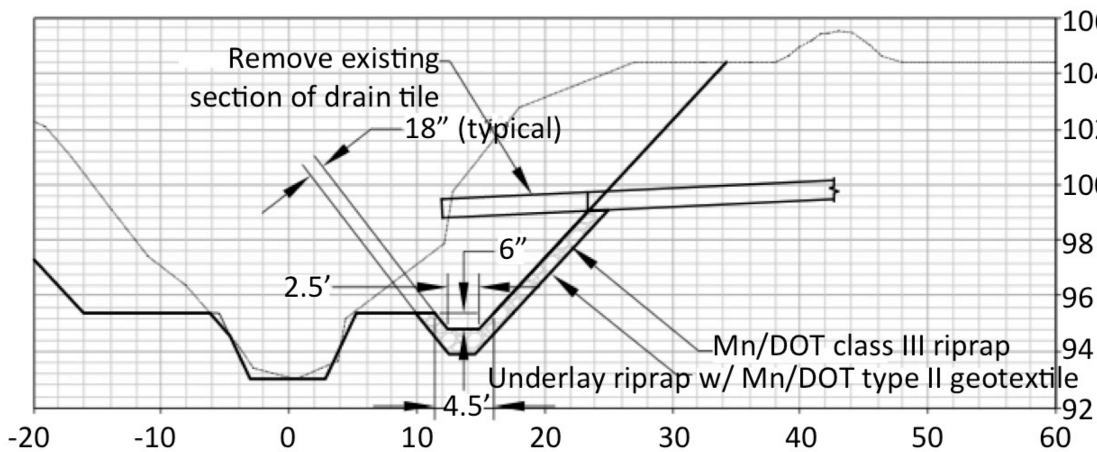

Figure 9. Detail of Tile Outlet to a Linear Treatment System. Dimensions in Feet and Inches. Drawing by Joel Peterson.

3.5.4 Seepage Trench

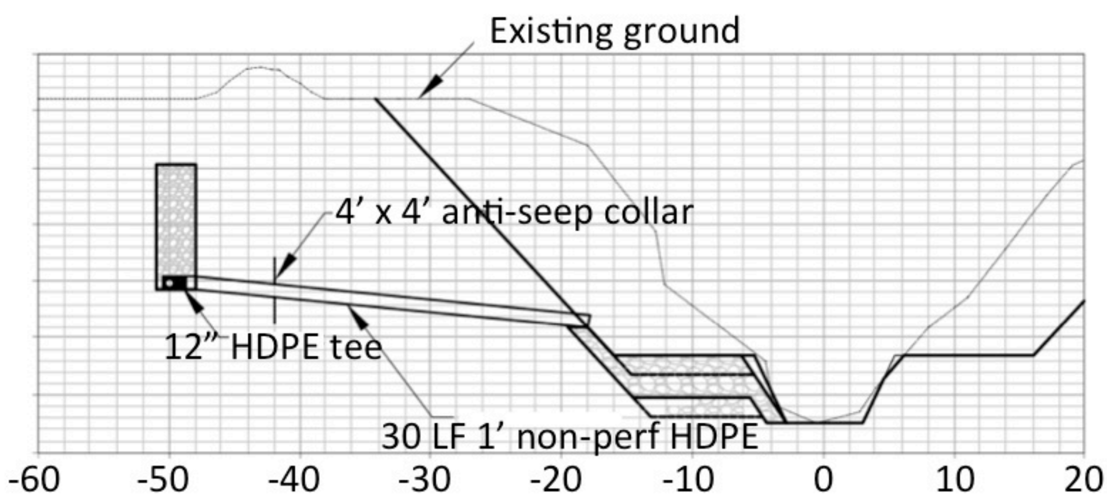

Figure 10. Two-Stage Channel Cross-Section Showing the Seepage Trench. Dimensions of the Axis are in Feet and Inches. Drawing by Joel Peterson.

A seepage trench was also included in the construction process as an experimental method for reducing the problem of bank sloughing in drainage ditches, which can be caused by excessive soil moisture and lateral subsurface flow in ditch banks. The seepage trench is designed to collect excess soil moisture and lateral seepage flow before it can reach the ditch bank, thereby reducing soil moisture and increasing stability in the adjacent ditch bank.

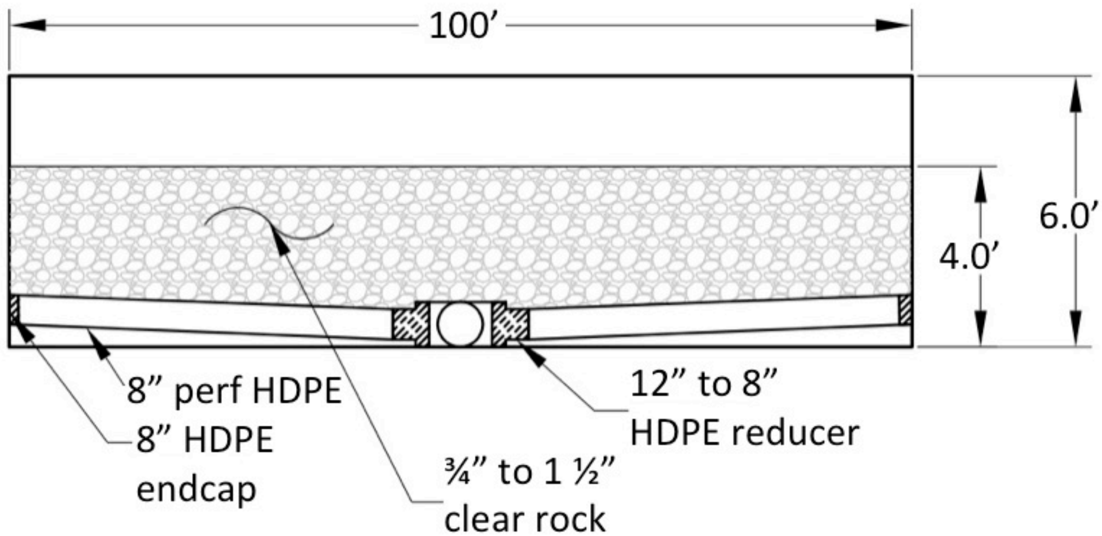

Figure 11. Side View of the Seepage Trench. Dimensions in Feet and Inches. Drawing by Joel Peterson.

As Figure 11 shows, the trench was constructed at 100 ft (30.5 m) long and 3 ft (91 cm) wide, and extended from 2 ft (61 cm) below the soil surface to 6 ft (1.83 m) below the soil surface, and was filled with rock with a diameter of ¾ to 1½ inches (19 to 38 mm). A perforated 8-inch (20.3 cm) plastic collector pipe was located in the middle of the seepage trench at the bottom (6 ft or 1.83 m below the soil surface) to carry intercepted water to the seepage trench outlet pipe. The outlet pipe (12-inch or 30.5 cm plastic) was laid perpendicular to the seepage trench to carry water through the ditch bank and into the channel. Riprap armoring was used beneath the outlet pipe and across the ditch bench to minimize scour and erosion. Figure 10 shows the location of the seepage trench in the outside ditch bank. See the appendix for several pictures (Figure 53 - Figure 56) of the seepage trench construction.

3.5.5 Rock Trench Side Inlet

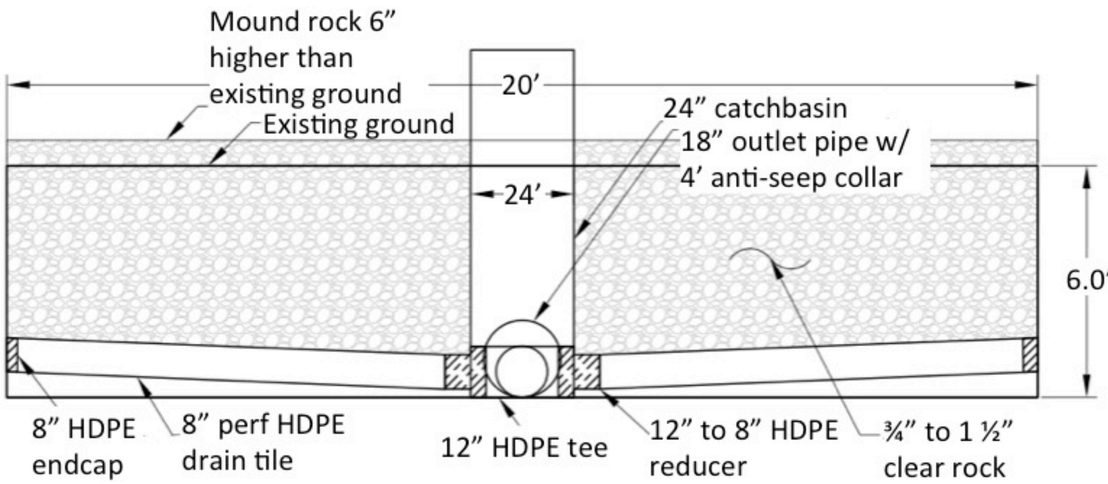

Figure 12. Side View of the Rock Trench Side Inlet. Drawing by Joel Peterson.

A rock trench side inlet (Figure 12 and Figure 13) was also installed at the site. The rock trench side inlet was designed to incorporate a seepage trench into the replacement of a standard side inlet. Like the installed seepage trench, the rock trench extends to 6 ft (1.83 m) below the soil surface. However, the rock trench differs in that it extends to the soil surface to allow for overland flow to enter the trench. Gravel was mounted to a height of 6 inches (15 cm) above the soil surface to slow down surface water as it approaches and enters the rock trench. An auxiliary surface tile inlet (24 inch or 61 cm diameter) was included, with an inlet elevation 2 ft (61 cm) higher than the soil surface. This structure allows water to bypass the rock trench during high runoff events. As with the seepage trench, the rock trench side inlet structure outlets to a horizontal pipe, which enters the ditch bench on an armored surface. The rock trench side inlet was constructed parallel to the channel and at a length of 20 ft (6.1 m) and a width of 3 ft (91 cm). Rock used to fill the rock trench had a diameter of ¾ to 1½ inches (19 to 38 mm). The location of the rock trench side inlet is shown in Figure 7. A picture of the constructed rock trench side inlet is included in the Appendix (Figure 50).

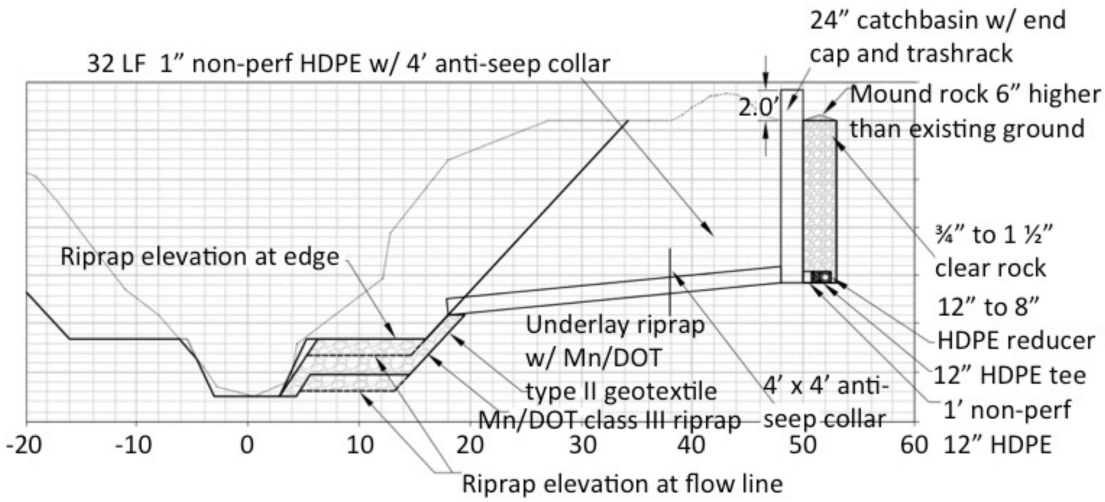

Figure 13. Cross-Section of the Two-Stage Ditch Showing the Rock Trench Side Inlet Structure. Dimensions of the Axis are Given in Feet. Drawing by Joel Peterson.

3.5.6 Field Road Crossing

Agricultural producers along both sides of the ditch have historically allotted a thin strip of land between the top of the ditch banks and their agricultural fields for field roads. These field roads allow farmers to move machinery more efficiently to their fields located on both sides of the ditch. A field road crossing through the ditch also existed approximately half-way between the ends of the ditch reach (as shown in Figure 7) and the land owners expressed their desire to be able to continue crossing the ditch channel at that location. The field road crossing consisted of a rock base before construction, and the replacement crossing was designed similarly, with a new layer of rock (approximately 3-4 inches in diameter) making up the surface. A picture of the completed field road crossing is included in the Appendix (Figure 64).

3.5.7 Erosion Control Measures

Erosion control was handled by anchoring erosion control blankets along the channel benches and outside slopes. Blankets were installed by a subcontractor selected by Freeborn Construction. All erosion control blankets were selected following Minnesota Department of Transportation guidelines (MnDOT, 2005). 500 yd^2 (418 m^2) and 210 yd^2 (176 m^2) of Category 3 and Category 2 (MnDOT, 2005) geotextile blankets were used in the installation of riprap below side inlet and tile outlets. 48,400 yd^2 (40,500 m^2) of MnDOT Category 2 erosion control blankets were used to line the exposed ditch benches and outside ditch banks following construction.

3.5.8 Seeding of vegetation

In order to establish stable vegetation, a seed mixture was applied on the soil surface after construction. The seed mixture applied was developed and recommend by the Minnesota State Board of Water and Soil Resources (BWSR). A complete list of all species present in the mixture is given in Table 26 in the Appendix.

Unfortunately, due to wet conditions throughout the autumn of 2009 that delayed construction, little vegetation was established before the winter of 2009/2010. When the project was completed in early November, very little vegetation had emerged through the erosion control blanket. With the hopes of establishing vegetation before the winter season, and due to unseasonably warm temperatures following the end of construction, an additional application of rye seed was made in early November 2009. The rye seed was selected as the species with the highest likelihood of emergence before the winter season. The rye seed generally did not emerge before snowfall, but did emerge in the spring and summer of 2010. Vegetation in the ditch became fully established during the 2010 growing season. Detailed vegetation monitoring was conducted at the ditch site during the summer of 2010, but those results are not discussed in this paper.

4 Mullenbach Two-Stage Drainage Ditch Monitoring and Analysis

An important aspect of the Mullenbach two-stage ditch project is measuring the performance and stability of the ditch over time. In order to measure the ditch's performance, and to track changes in the ditch over time, several aspects of the ditch's performance have been measured, including: geomorphic changes, water quality sampling to study nutrient dynamics, and isotope sampling to assess hydraulic residence and source waters contributing to flow throughout the reach.

4.1 Channel Geomorphic Monitoring and Stability Analysis

Several profile and cross-sectional surveys were conducted at the Mullenbach two-stage drainage ditch site. The purpose of these surveys was to monitor the current ditch conditions, as well as to measure changes that took place within the ditch from the pre-construction initial condition through the construction process to October 2010. This section describes the types of geomorphic measurements and presents the analysis of these measurements to assess channel stability.

All channel cross-sections were completed with a laser level or a total station (locked with a level sighting) and a stadia rod. Cross-sections were staked with wooden stakes and distance across the channel was measured with a 100- or 200-ft (30.5 m or 61 m) long field tape measure. For the initial site survey in April 2009, a ¾" (19 mm) metal rod was pounded into the soil near the culvert at Mower County Highway 6 to provide an arbitrary project benchmark elevation. The benchmark was lost in the fall of 2009 when the culvert beneath Mower County Highway 6 was replaced (the new double-box culvert can be seen in Figure 57 in Appendix A.3). The top of the culvert beneath 120th Street was selected as a new benchmark, as its elevation had been measured as part of the initial site survey, and was therefore tied to the original project datum of 100 ft (30.48 m). For clarification, the current benchmark on top of the culvert is the top of the lower lip (the most downstream extent of the culvert), and the elevation is 109.63 (33.42 m). No attempt has thus far been made to tie the current or former project benchmarks to sea level.

4.1.1 Channel Cross-Section Surveys

Seven cross-sectional surveys were conducted as part of the initial pre-construction site survey in April 2009 (shown in Figure 14). Four of the cross-sections were re-measured in August 2010. All seven cross-sections were surveyed in October 2010. Although the cross-sections were not surveyed immediately following construction, design specifications and field notes provide useful information on constructed cross-sections in late 2009. As outlined previously, bench building was conducted in two reaches in order to narrow the low-flow channel. No bench building was performed in the section from 7+00 to 39+00, so it is possible to estimate the constructed cross-sections at 14+67 and 28+25 by combining the April 2009 survey data with the constructed bench elevation. For the remaining five cross-sections, notes during construction allow an approximate constructed low-flow channel width to be reported. These dimensions are included in Table 3, along with a summary of all cross-section surveys completed.

Table 3. Summary of Cross-Section Surveys Conducted at the Mullenbach Ditch.

	Survey Date			
Location	*Pre-Construction*	*As-Constructed Estimated Cross-Section*	*August 2010*	*October 2010*
5+16	Yes	Benches constructed (11 to 12 ft channel width)	Yes	Yes
14+67	Yes	Yes	Yes	Yes
28+25	Yes	Yes	No	Yes
40+04	Yes	Benches constructed (8 ft channel width)	No	Yes
46+54	Yes	Benches constructed (8 ft channel width)	No	Yes
54+83	Yes	Benches constructed (8 ft channel width)	Yes	Yes
58+39	Yes	Benches constructed (8 ft channel width)	Yes	Yes

The cross-sectional data were analyzed using RiverMorph version 4.3 (RiverMorph, LLC, 2010). This software package was used to compute dimensions such as low-flow width and cross-sectional area, bench elevation, and mean and maximum depth in the low-flow channel. A summary of the dimensions is given in Table 4.

Three of the four sites surveyed in both August and October 2010 showed reductions in low-flow channel width, depth (maximum and mean), and cross-sectional area from August to October 2010. The cross-section at 14+67 (riffle) showed large reductions in all of the parameters, most noticeably low-flow channel cross-sectional area, where a 24.6% decrease was measured. Figure 15 shows a dramatic change in the low-flow channel depth at the cross-section 14+67 from prior to construction to October 2010.

Table 4. Summary of Measured Low-Flow Channel Features from Channel Surveys at Construction and in August and October 2010.

		Dimensions of low-flow channel features				
Cross-Section		*Elevation (ft)*	*Width (ft)*	*Area (ft^2)*	*Mean Depth (ft)*	*Max Depth (ft)*
5+16						
Constructed		104.06	*11 to 12 (approx.)*	N/A	N/A	N/A
August 2010		104.4	11.98	13.64	1.14	2.1
October 2010		104.35	10.33	11.41	1.1	1.89
% change	August '10 - October '10		-13.8	-16.3	-3.5	-10.0
14+67						
Constructed		101.58	12.44	17.98	1.45	2.19
August 2010		101.92	12.64	14.21	1.12	1.69
October 2010		101.98	12.09	10.72	0.89	1.46
% change	Construction - August '10		+1.6	-21.0	-22.8	-22.8
	August '10 - October '10		-4.4	-24.6	-20.5	-13.6
	Construction - October '10		-2.8	-40.4	-38.6	-33.3
28+25						
Constructed		98.88	10.84	17.05	1.57	2.28
October 2010		99.45	11.83	19.76	1.67	2.61
% change	Construction - October '10		+9.1	+15.9	+6.4	+14.5
40+04						
Constructed		97.16	*8 (approx.)*	N/A	N/A	N/A
October 2010		97.71	9.41	11.71	1.24	2.11
46+54						
Constructed		96.59	*8 (approx.)*	N/A	N/A	N/A
October 2010		96.5	11.59	16.52	1.43	2.05
54+83						
Constructed		95.27	*8 (approx.)*	N/A	N/A	N/A
August 2010		95.55	13.96	22.98	1.65	2.52
October 2010		96.07	12.77	20.23	1.58	2.31
% change	August '10 - October '10		-8.5	-12.0	-4.2	-8.3
58+39						
Constructed		94.7	*8 (approx.)*	N/A	N/A	N/A
August 2010		94.51	12.98	16.27	1.25	2.16
October 2010		94.7	13.51	17.57	1.3	2.23
% change	August '10 - October '10		+4.1	+8.0	+4.0	+3.2

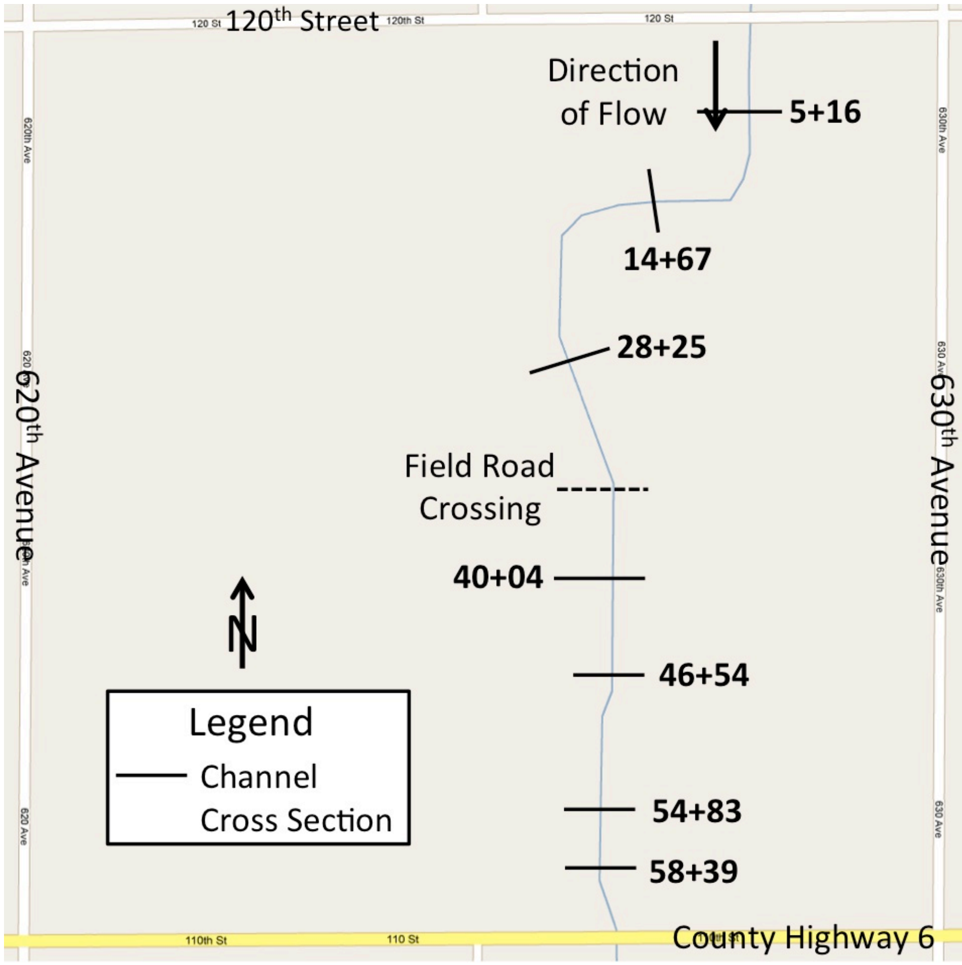

Figure 14. Mullenbach Two-Stage Ditch Site Showing Measured Channel Cross-Sections. Base Map © Google.

The large changes measured in low-flow channel dimensions from August to October 2010 may have been driven by a large storm event that took place on September 22-23, 2010. 4.54 inches (115 mm) of precipitation were measured at the Mullenbach site between 17:11 on September 22nd and 16:58 on September 23rd. The discharge that resulted in the two-stage channel from the rainfall event is uncertain, but the stage within the ditch channel was observed at a level of 2 to 3 ft (61 to 91 cm) below the top ditch bank (field level) on the afternoon of September 23rd (Figure 63 in the Appendix).

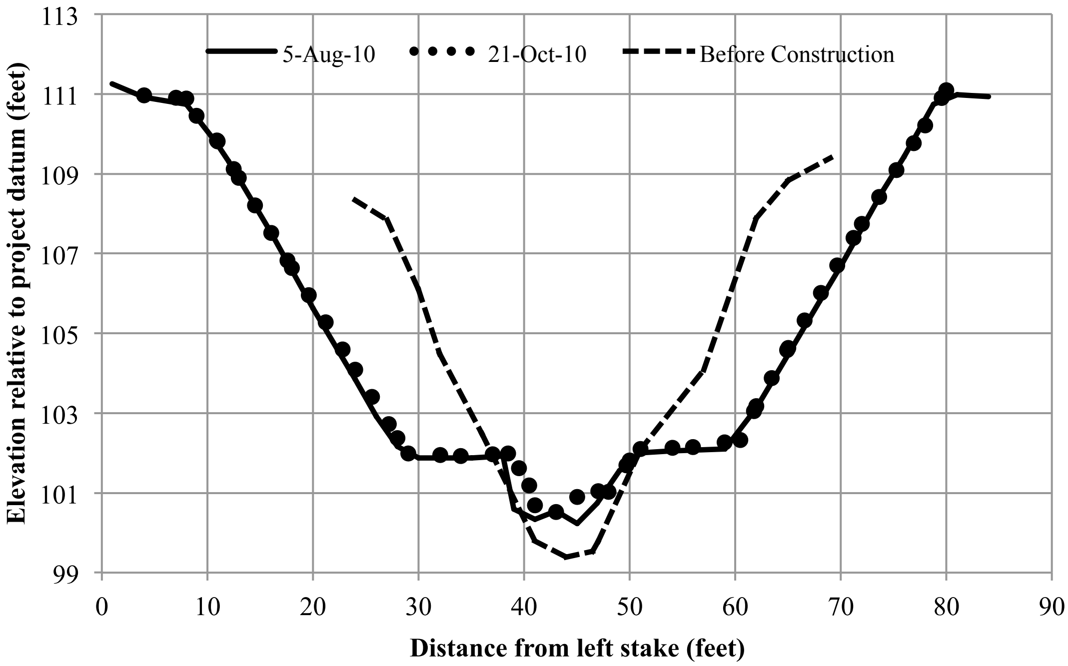

Figure 15. Three Cross-Section Measurements Taken at the Station 14+67.

The development of an inner berm within the low-flow channel has been noted (both through observations at the ditch and through cross-section analysis) along nearly the entire ditch reach, including those areas where no fill material was added to the low-flow channel to reduce channel width. This further reinforces and supports the reduction of cross-sectional area in many cross-sections, as well as a shift towards the proposed design low-flow channel width that resulted from the regional curves. The inner berm development also suggests that the channel may be shifting toward a multi-stage channel with a smaller inset channel within the constructed low-flow channel.

This may be a consequence of the types of sediment and water loads being carried by the channel, and how the channel hydraulics are interacting with the substrate in the low-flow channel.

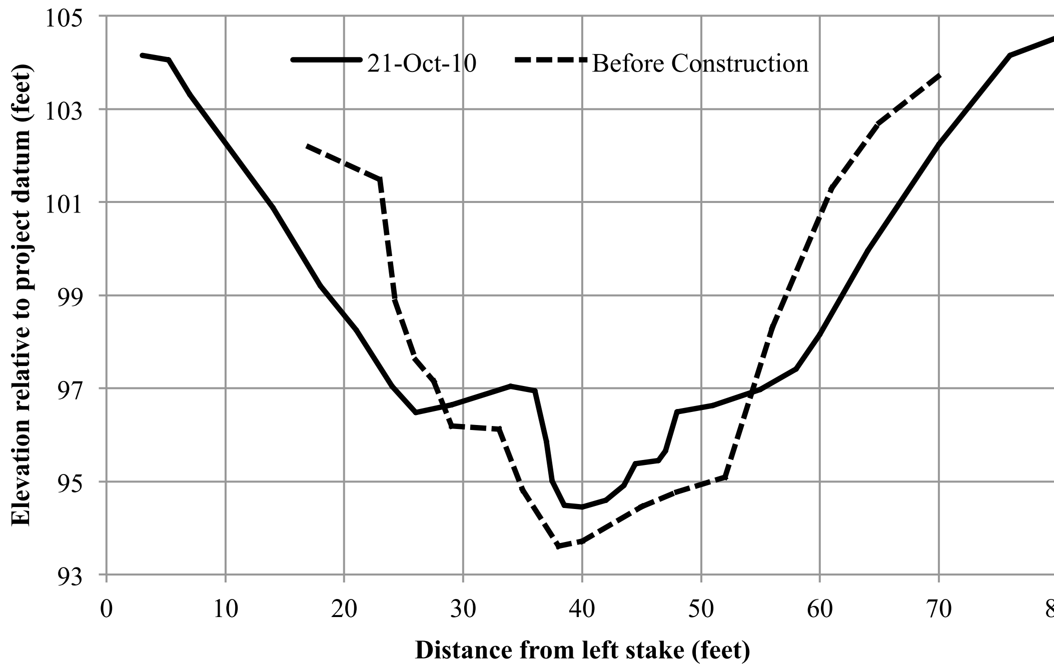

Figure 16. Two Cross-Section Measurements Taken at the Station 46+54.

Figure 16 shows the two cross-sections measured at the station 46+54. Note that soil was added to both sides of the channel at this cross-section during construction. Also note that this cross-section intersects a linear treatment system, which can be seen on the edge of the left bench adjacent to the outside ditch bank. Construction notes show that the low-flow channel width at 46+54 was reduced to approximately 8 ft (2.44 m) during the construction process. The measured low-flow channel width in October 2010 was 11.59 ft (3.53 m), which suggests that some of the soil that was added to the low-flow channel has been eroded. Future measurements are required to determine the success of the bench building that was done to narrow the low-flow channel. The widening of the channel at 46+54 suggests that the bench building was not as successful as was hoped. The channel remains much narrower, however, than before construction. Continued monitoring will show whether this cross-section stabilizes, or continues to widen. Additionally, the channel thalweg increased in elevation by approximately 6 inches (15 cm) from before construction to October 2010.

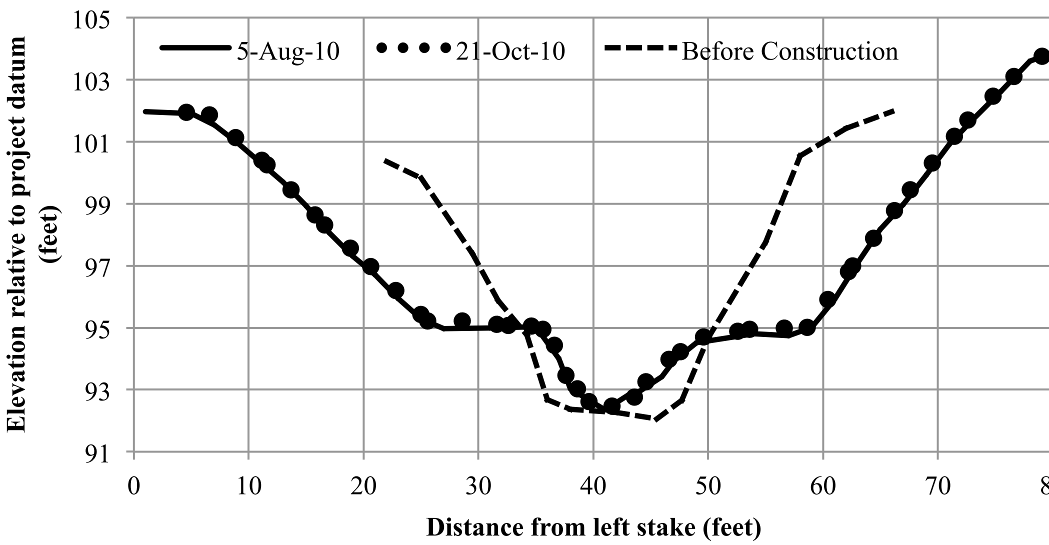

Figure 17. Three Cross-Section Measurements Taken at the Station 58+39.

Figure 17 shows the cross-section surveys that were conducted at the 58+39 cross-section. Soil was added to both sides of the low-flow channel during construction to reduce the low-flow channel width to approximately 8 ft (2.44 m). The measured low-flow channel width in October 2010 had increased to 13.51 ft (4.12 m), a dramatic increase. The graph of the three cross-sections surveys seems to show that the low-flow channel has nearly reverted back to its pre-construction width, although the channel cross-sectional area is noticeably smaller. This suggests that the material added to the channel to narrow the low-flow channel may not be stable at higher flows (when the stage nears the top of the low-flow channel), but may be stable during very low flows, where stream power is low. This shows that building benches to reduce the low-flow channel width may be more successful in situations where vegetation can be quickly established to increase bank stability and erosion resistance along the sides of the low-flow channel.

Figure 70 through Figure 73 in the Appendix show cross-sectional surveys for the four cross-sections not discussed in the text.

4.1.2 Channel Longitudinal Profile Surveys

Longitudinal channel thalweg surveys were conducted on three different occasions: prior to construction (April 2009), during construction (October – November 2009), and approximately one year after completion of construction (October 2010).

Table 5. Summary of Data Collected for Three Longitudinal Surveys.

Survey Date	Start station	End station	Number of survey points	Method
April 2009 (prior to construction)	3+55	62+00	39	Constant interval: 200ft (with some variation)
October – November 2009 (during construction)	8+00	61+00	54	Constant interval: 100ft
October 2010	0+00	61+00	125	Based on channel features

Longitudinal surveys were conducted with a laser level and stadia rod, and measured the channel thalweg along the project reach. The measurement interval for the first two surveys was a pre-set distance (given in Table 5). The October 2010 survey was a more thorough channel survey based on channel features (riffle and pool indicators) and points were taken to capture all of the slope changes within the reach. A summary of the channel thalweg data collected is presented in Table 5. Graphical representations of the channel thalweg longitudinal surveys are included in Figure 18 and Figure 19. Figure 18 shows the changes in channel bed form from April 2009 to construction (October – November 2009). There are few visible changes over this time period. Figure 19 shows the changes in channel bed form after construction, as measured in August 2010. There is a considerably higher pool-riffle sequence frequency evident in the August 2010 survey. There are also reaches where the channel bed elevation has noticeably increased (notably upstream of 13+00 and downstream of the field road crossing at 35+45). The results also appear to show that the areas with newly developed pool-riffle sequences correspond to areas where the channel bed has aggraded. This suggests that pool-riffle sequences have developed in newly deposited bed sediments.

Bed material diameter was also measured at each of the 119 survey points from the longitudinal survey; the particle size distribution is summarized in Figure 20. The measured D_{50} was 1.4 mm.

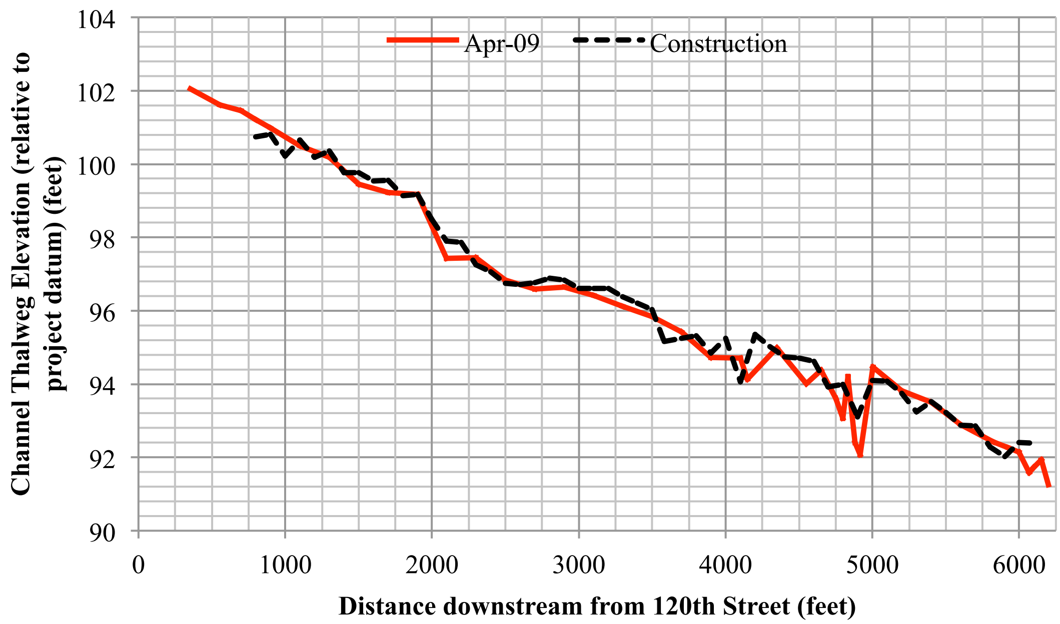

Figure 18. Longitudinal Channel Thalweg Elevations from April 2009 and Construction Surveys.

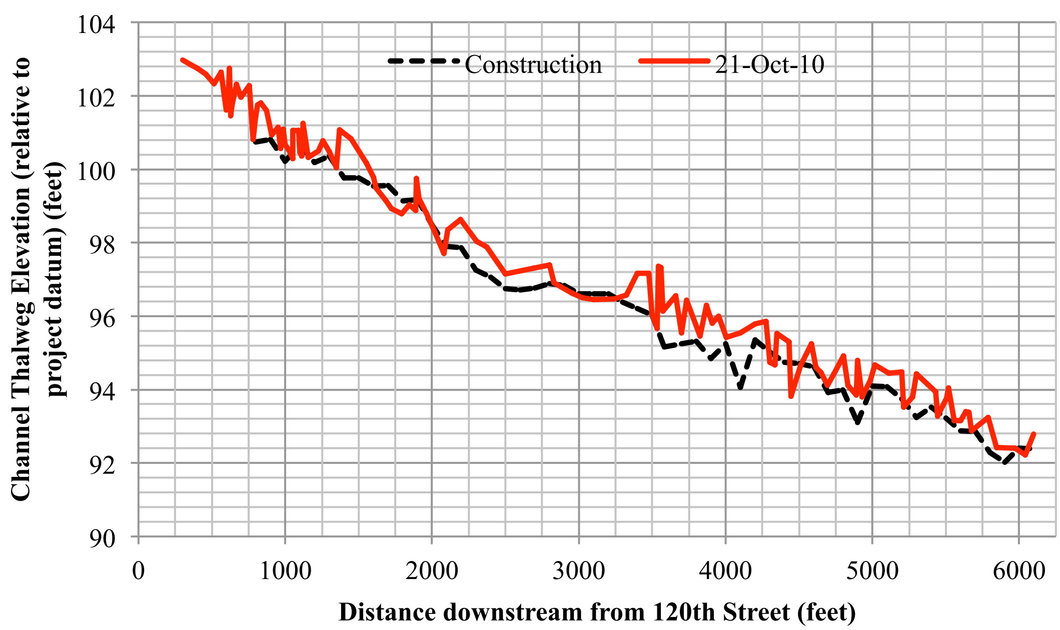

Figure 19. Longitudinal Channel Thalweg Elevations from Construction and October 2010 Surveys.

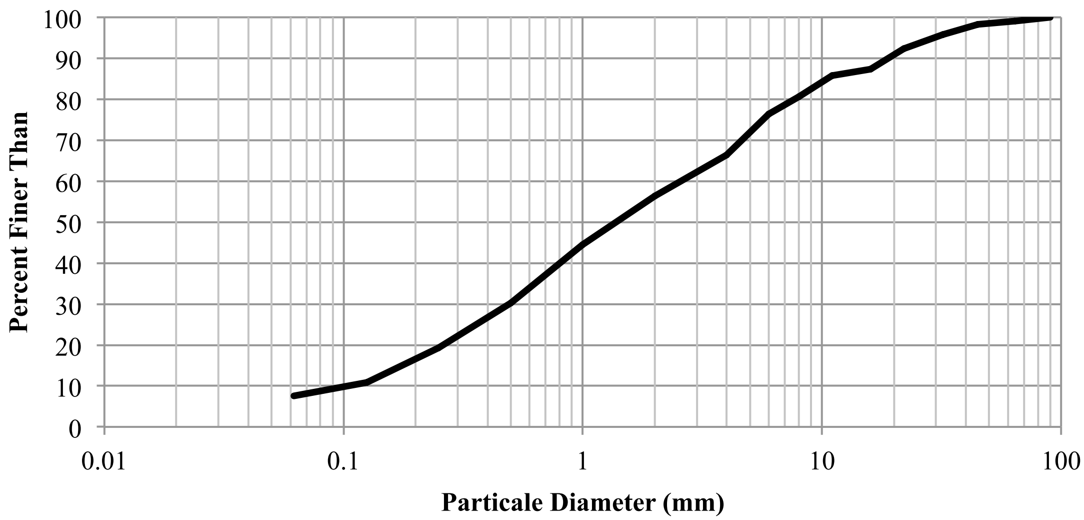

Figure 20. Particle Size Distribution from Pebble Count Conducted During Longitudinal Profile Survey, October 2010.

In order to measure changes that may have taken place in the ditch, an analysis was carried out to measure vertical changes in the channel bed. In order to assess whether the channel has aggraded, degraded, or remained stable, the trapezoidal rule was used to calculate the area (along the channel profile) of the streambed relative to the project datum. The calculated area was then divided by the length of the channel survey to determine average bed elevation.

To compare the average bed elevation, the surveyed channel reach must be the same for all three surveys. Unfortunately (following Table 5), each of the longitudinal surveys began at a different location. The survey conducted during construction did not begin at station 0+00, but at 8+00. In order to compare the three surveys, the other two surveys were modified to exclude points upstream of 8+00. Further complicating the issue, survey points were not taken at exactly 8+00 for the April 2009 and October

2010 surveys. Therefore, the furthest upstream point (8+00) for the April 2009 and October 2010 surveys was estimated by linear interpolation between the two nearest (one point upstream and one point downstream) measured thalweg elevations. The average elevations over the reach from 8+00 to 61+00 for each survey are presented in Table 6.

Table 6. Average Elevation from 8+00 to 61+00 for Three Longitudinal Profiles.

Measurement date	Average Elevation, ft (m)
Prior to construction (April 2009)	96.06 (29.29)
During construction (October-November 2009)	96.16 (29.32)
October 2010	96.57 (29.44)

**Naturally, the average bed elevation for those intervals excluding the upper section of the reach will tend to be lower than the average elevation estimated for those intervals extending further upstream.*

A comparison shows an aggradation of 0.51 ft (15.5 cm) from prior to construction to October 2010 over the interval 8+00 to 61+00 interval, while it was somewhat lower, at 0.41 ft (12.5 cm), from construction to October 2010. This agrees with the observations of recently constructed two-stage drainage ditches in Ohio, Michigan and Indiana made by Kallio et al. (2010).

Because the two earliest surveys were done in a somewhat coarse, less-detailed manner than the October 2010 survey, it is necessary to quantify the measurement error introduced when using a survey consisting of fewer points. In order to estimate the error, the October 2010 data set was modified by removing points so that a computational ‘coarser’ data set was generated for comparison to the more complete data set (Table 7).

Table 7. Calculated Average Bed Elevations for Two Longitudinal Channel Surveys from 8+00 to 61+00.

	Calculated average elevations over survey reach, ft (m)		
Survey Date	Based on channel features (thorough)	Constant interval: 100 ft	Constant interval: 200 ft
Survey at Ditch Construction	N/A	96.157 (29.316)	96.169* (29.320)
October 2010	96.568 (29.442)	96.560* (29.439)	96.558* (29.438)

** denotes that this average bed elevation corresponds to a synthetic data set that was created from a more robust data set*

The error associated with a constant-interval survey can be estimated by using the above average elevations from the original October 2010 survey, and the average elevations obtained by removing points to mimic a survey based on a constant interval. The error is calculated as:

$$error = \frac{\overline{Elev}_{cf} - \overline{Elev}_{ci}}{Elev_{max} - Elev_{bed}} \qquad (1)$$

where $\overline{Elev}_{cf}$ is the average elevation as determined by a thorough survey based on channel features, $\overline{Elev}_{ci}$ is the average elevation as determined by a channel survey of constant interval, $Elev_{max}$ is the maximum channel thalweg elevation measured in the channel survey, and $Elev_{min}$ is the minimum channel thalweg elevation measured in the channel survey. The errors are quantified in Table 8.

Maximum and minimum measured elevations for the October 2010 survey were 101.78 ft (31.02 m), and 92.31 ft (28.14 m), respectively. Average elevations are reported for the interval from 8+00 to 61+00.

Table 8. Error Associated with Calculating Average Bed Elevation from Points Sampled on a Constant Interval for the October, 2010 Longitudinal Survey.

Complete Data Set	**100 ft (30.5 m) interval Data Set**			**200-ft (61 m) interval Data Set**		
Average Elevation, ft (m)	Average Elevation, ft (m)	Absolute Error, ft (m)	Percent Error	Average Elevation, ft (m)	Absolute Error, ft (m)	Percent Error
96.57 (29.43)	96.56 (29.43)	0.01 (0.003)	0.106	96.56 (29.43)	0.01 (0.003)	0.106

Following the error analysis, the errors associated with both of the constant interval surveys are 0.1% of the difference between maximum and minimum elevation measurements. Absolute errors for the two earlier (constant interval) surveys are estimated in Table 9.

Table 9. Computed Average Elevation Errors from Two Channel Surveys (8+00 to 61+00) Based on Constant Measurement Interval.

Survey Date and Method	**Average Elevation, ft (m)**	**Maximum thalweg elevation, ft (m)**	**Minimum thalweg elevation, ft (m)**	**Percent Error**	**Absolute Error, ft (m)**
Pre-construction (200-ft interval)	96.06 (29.29)	101.22 (30.86)	91.59 (27.92)	0.1056	0.010 (0.003)
During Construction (100-ft interval)	96.16 (29.32)	100.82 (30.74)	92.02 (28.05)	0.1056	0.009 (0.003)

The absolute errors of 0.009 ft (0.003 m) and 0.010 ft (0.003 m) in Table 9 are approximately equal to the measurement resolution of 0.01 ft (0.003 m) of the stadia rod, and show that the coarse survey method is sufficient to determine average bed elevations for estimating changes in bed profile.

4.1.3 Rosgen Channel Type Classification

The following stream classification was performed using the Rosgen classification system (Rosgen, 2006), with information from channel longitudinal and cross-section surveys conducted in October 2010. Data from the cross-section at 14+67 were used because 14+67 was the only benchmarked cross-section located at a riffle. A riffle is the desired stream feature to use in conjunction with the Rosgen classification system. The channel dimensions and data are presented in Table 10.

Table 10. Summary of Channel Dimensions for the Rosgen Channel Classification.

Channel feature	Measurement (Imperial)	Measurement (metric)
Low-flow channel depth(1)	0.8 ft	24 cm
Low-flow channel cross-sectional area(1)	9.8 ft^2	0.91 m^2
Width/depth ratio (low-flow channel)(1)	13.7	13.7
Maximum depth(1)	1.4 ft	43 cm
Width of flood-prone area(1)	36 ft	11 m
Entrenchment ratio(1)	3.1	3.1
Channel materials D_{50}(2)	0.055 in	1.4 mm
Water surface slope(2)	0.0033	0.0033
Channel sinuosity(3)	1	1
Rosgen channel classification	C4 to E4	C4 to E4

(1) value obtained using the Mecklenburg macro spreadsheet (Mecklenburg, 1998) to analyze cross-section information from the channel survey, (2) value obtained from channel longitudinal survey, where a pebble count was completed for the entire channel length. The D_{50} value was not collected and analyzed for the cross-section, but is likely larger than the reported value of 1.4mm, as the cross-section is a riffle, and (3) channel sinuosity is estimated to be ~1 as the channel is still almost entirely confined within the original constructed geometry, which included no meandering.

The Rosgen classification system yielded a channel type of C4-E4. Sinuosity was ignored in the classification due to the fact that the Mullenbach was recently constructed and has not had time to adjust its sinuosity. The entrenchment ratio and slope (> 2.2 and < 0.2, respectively) in C and E channels are equal, and so ignoring sinuosity the only difference between C and E channels is the width-to-depth ratio. C channels have a width-to-depth ratio of greater than 12, while the ratio is less than 12 in E channels. The width to depth ratio of 13.7 in the riffle section gives a stream channel type of C, while this value would likely be lower at other cross-sections, due to greater channel depth. Because the width to depth ratio is relatively close to 12, it is difficult to make a decisive conclusion between an C4 and an E4 channel.

4.2 Precipitation Monitoring

Precipitation at the Mullenbach ditch site was measured with a Campbell Scientific tipping bucket rain gauge. The rain gauge was installed near 120th Street at the north end of the ditch (Figure 21). Each tip recorded precipitation of 0.01 inches and the number of tips was recorded every minute by a Campbell Scientific CR10X data logger.

4.3 Volumetric Flow Monitoring

Wooden box flumes were installed at both the upstream and downstream ends of the study reach in the spring of 2010 (see Figure 21). The upstream flume was installed just downstream of the culvert underneath at approximate station 1+50, and the downstream flume was installed just upstream of the downstream culvert, at approximate station 60+95. Each flume bed was an 8 ft (2.44 m) (across the channel) by 4 ft (1.22 m) (along the direction of flow) sheet of plywood. The bed was attached to 2 ft (61 cm) tall vertical walls. Wing walls were then attached to the vertical walls, and extended at an approximate 45-degree angle outward toward the low-flow channel banks, in order to concentrate flow through the flume.

Druck pressure transducers were installed in both flumes to measure water depth. A pressure transducer was also installed at the downstream end of the 120th Street culvert for the estimation of water discharge at the inlet of the ditch. It was mounted at an elevation equal to that of the downstream end of the culvert. All depth readings measured by the pressure transducers were logged on Campbell Scientific CR10X data loggers. Figure 21 shows the monitoring setup used during 2010.

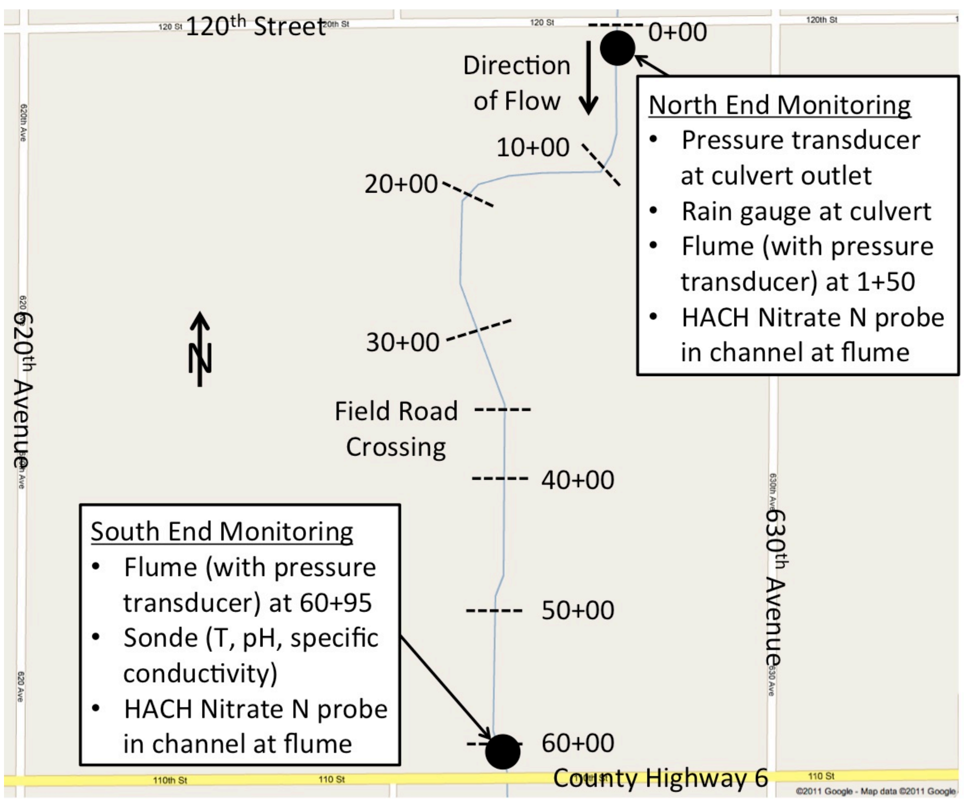

Figure 21. Map of the Mullenbach Two-Stage Ditch Site Showing the Monitoring Equipment Used During the 2010 Field Season. Base Map © Google.

The pressure transducers installed in each of the flumes were set up to take water depth readings every 15 minutes for the monitoring season. The sensors were installed on April 27 and removed from the field on October 21, 2010. Equipment problems caused some issues in storing data from the sensors, such that depth data are missing for the north flume from April 27 through June 3, and from June 3 through June 29 for the south flume. Thus, uninterrupted 15-minute data were collected for both flumes from June 29 through October 21, 2010.

4.3.1 Stage-Discharge Relationship Development

To determine water depth/discharge relationships for both installed flumes, depth-discharge measurements were taken on seven days during the summer and fall of 2010. On six days, measurements were taken using the float method (Robins and Crawford, 1954) with an apple to estimate the surface velocity through the flume. Surface velocity is related to discharge using Equation 2 (Robins and Crawford, 1954):

$$Q = \frac{awdL}{t} \tag{2}$$

where *Q* is volumetric stream discharge, *w* is average stream width, *d* is average stream depth, *L* is the length of section measured (along the direction of flow), *t* is the time for the floating object to travel the distance *L*, and *a* is an adjustment factor. Robins and Crawford (1954) suggest an *a*-value of 0.8 for a rough bed (gravel and coarse rocks) and 0.9 for a smooth bed (mud, sand, hardpan). A value of 0.85 was used for the flumes at the Mullenbach site as the flumes are constructed of smooth plywood, while the channel bed near both flumes is rougher (gravel and coarse sand).

Table 11. Summary of Depth-Discharge Measurements Conducted in 2010.

		North Flume		South Flume	
Date	**Measurement method**	**Measured flow depth, ft**	**Corrected Discharge, cfs**	**Measured flow depth, ft**	**Corrected Discharge, cfs**
June 29	Float	1.066	4.15	0.750*	10.20*
July 17	Float	0.317	0.73	0.354	1.84
August 2	ADV	0.640	1.85	0.708	4.16
August 4	Float	0.600*	2.66*	0.410*	3.26*
August 8	Float	0.453	1.28	0.323	2.29
August 24	Float	0.472	1.28	0.365	0.98
October 21	Float (multiple points)	0.550*	0.08*	0.510*	0.17*

**denotes data that were excluded in the development of the rating curves*

An acoustic Doppler velocimeter (ADV) was used on one occasion to more accurately measure the stream flow across the entire cross-sections of the flumes. ADVs operate on the principle of a Doppler shift, and are capable of measuring velocity fluctuations and average velocity at high frequency. The ADV device used in this study was a SonTek FlowTracker handheld ADV device. The FlowTracker device can measure over a range of 0.001 to 4.0 m s^{-1} and has an accuracy of +/- 1% of measured velocity or +/- 0.25 cm s^{-1} (SonTek/YSI, 2011). The signal transmitter and receivers are places at a height above the channel bed corresponding to 40% of the channel flow depth. Using this height allows for a single point measurement to provide the average water velocity over the entire water column. Velocity and depth in the flumes was measured on one-ft (30.5 cm) intervals across the channel width and combined to estimate volumetric discharge. A summary of data collected is given in Table 11.

Ward and Trimble (2004) showed that streamflow characteristics (average depth, average width, and velocity) are related to the channel discharge by power equations, as follows:

$$w = c_1 Q^e \tag{3}$$

$$h = c_2 Q^f \tag{4}$$

$$v = c_2 Q^g \tag{5}$$

where w is width, h is depth, and v is average velocity; c_1, c_2, c_3, e, f, and g are site-specific parameters. Equation (4) can be rearranged to produce a rating curve of the form:

$$Q = ah^b \tag{6}$$

where a and b are channel-specific parameters. Rating curves following Equation (6) were produced for both flumes using the data summarized in Table 11, with h in units of feet and Q in units of cubic feet s^{-1}. For both of the flumes, the data collected on August 4th were omitted from the rating curve regression, as the measurements varied significantly from those obtained on August 2nd with the more accurate ADV instrument. The measurements taken on October 21st were also excluded from the regression, as both flumes were measured to have very low discharges, while the depths were around 6 inches. The reasons for the low discharge measurements on October 21st are unclear, although it is possible that a large storm event on September 22nd and 23rd resulted in significant alteration of channel form near the flumes.

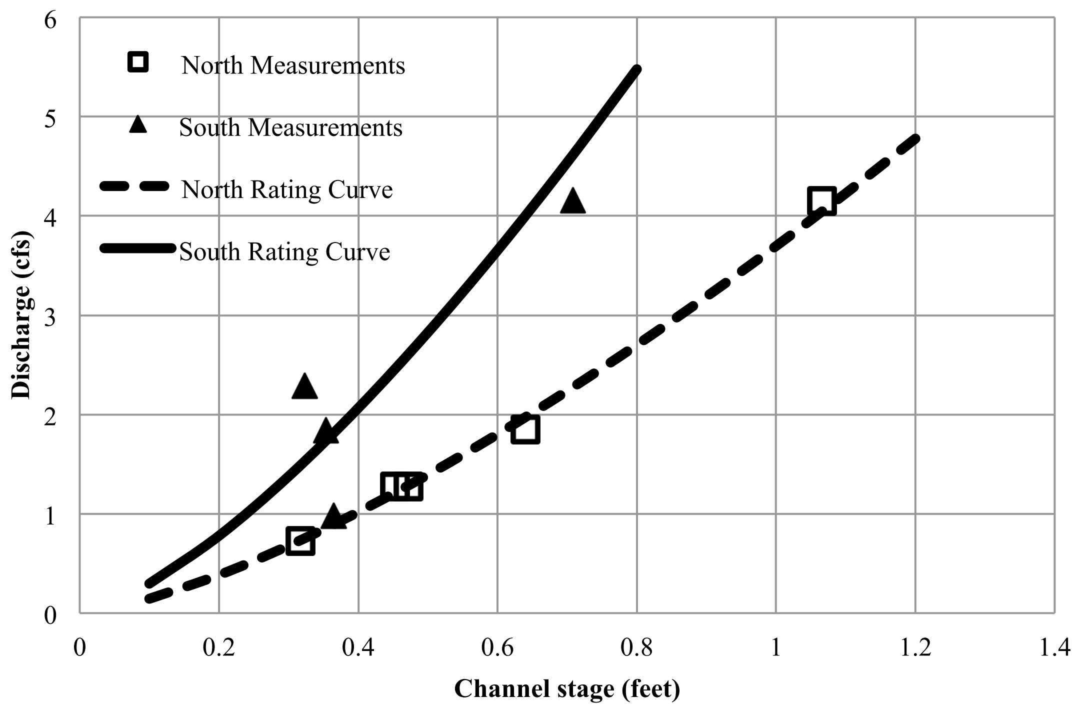

Figure 22. Stage-Discharge Relationships for the North and South Flumes.

As Figure 22 shows, the rating curve fits the measured data very well for the north flume. An additional data point (June 29th) was removed from the data set for the south flume prior to regression analysis. The June 29th discharge measurement is likely to be unreliable as the depth is only slightly higher than that of August 2nd (ADV measurement), while the discharge was measured to be nearly 150% higher. Unlike the depth-discharge data collected for the north flume, there was significant variation present in the data for the south flume. For this reason, and because the flume dimensions do not vary between the north and south flumes, it was decided to fix the *b* term (1.407) from the north flume rating curve for use in the south flume rating curve. The rating curve for the south flume was developed by averaging the respective a-values (obtained by rearranging Equation (6)) from each of the remaining four data points.

The rating curves for both flumes are summarized in Figure 22 and Table 12. The normalized mean square error (NMSE) (Wilson, 2001) for the north flume is low (0.004), while the NMSE for the south flume is higher (0.231). Compared to the north flume, there is a considerable amount of uncertainty in the rating curve for the south flume.

Because no flow measurements were taken while out-of-bank flow was occurring, there is no data to extend the rating curves beyond the depth of the channel benches. Thus, the rating curves are valid only for the low-flow channel, which corresponds to depths of 1.2 ft (36.5 cm) at the north flume and 1.0 ft (30.5 cm) at the south flume.

Table 12. Rating Curve Coefficients and Normalized Mean Square Error for the North and South Flumes.

Flume	a	b	NMSE (normalized mean square error)
North flume (inflow)	3.698	1.407	0.004
South flume (outflow)	7.491	1.407	0.231

4.4 Water Quality Monitoring and Sampling

Table 13. Summary of Water Quality Grab Samples and Field Measurements Conducted Through the End of 2010.

		Sonde				*Laboratory Analyses*							*Other Analyses*		
Year	Date	Temperature	Specific Conductance	pH	Dissolved Oxygen	Nitrate - Nitrite	Total Phosphorus	pH	Specific Conductance	Full Suite of Metals	Chloride	Isotope Analysis	Longitudinal temperature profile	HACH Nitrate N Probe	Tile Q
2009	28-Oct	-	-	-	-	5	5	5	5	-	-	5	-	-	-
	20-Nov	19	19	19	19	20	20	-	-	-	-	20	-	-	-
2010	24-Mar	-	-	-	-	7	7	7	7	-	-	7	-	-	-
	22-Apr	-	-	-	-	4	4	4	4	-	-	4	-	11	13
	13-May	-	-	-	-	6	6	6	6	-	-	4	-	-	-
	14-Jun	-	-	-	-	-	-	-	-	-	-	-	-	-	7
	16-Jun	-	-	-	-	-	-	-	-	-	-	-	-	24	17
	24-Jun	11	11	11	11	11	11	-	-	11	-	11	-	-	-
	14-Jul	-	-	-	-	-	-	-	-	-	-	-	X	-	-
	3-Aug	-	-	-	-	-	-	-	-	-	-	39	-	35	17
	4-Aug	-	-	-	-	-	-	-	-	-	-	-	X	24	17
	8-Aug	-	-	-	-	-	-	-	-	-	-	-	-	22	18
	21-Oct	-	-	-	-	8	8	-	-	-	8	-	-	-	-
	Sum	**30**	**30**	**30**	**30**	**61**	**61**	**22**	**22**	**11**	**8**	**90**	**2**	**116**	**89**

Water quality monitoring was performed in order to assess the quality of water from various sources within the ditch system, as well as to assess the performance of the ditch in improving water quality. The various water samples and water quality measurement techniques used in this study are included in the following sections. An overview of all data collected (dates and analyses performed) is given in Table 13; the full set of water quality data collected in this study is included in Appendix E.

4.4.1 In-Field Monitoring

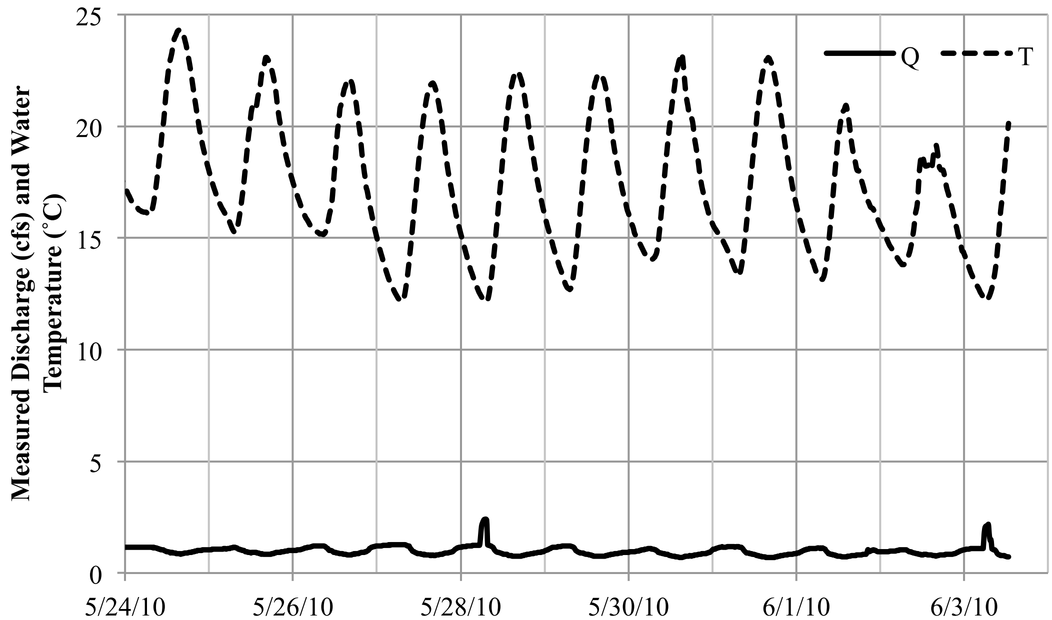

Figure 23. Sample Water Temperature Data Collected with the YSI Sonde in the South Flume during May and June, 2010, Shown with Measured Discharge (Q) through the South Flume.

A HACH Nitratax plus sc nitrate N probe was installed at both of the flumes to obtain hourly measurements of the nitrate N concentrations of water entering and exiting the ditch reach. The HACH nitrate N probes were purchased in early 2010, and were pre-calibrated from the factory. The factory calibration of each probe was tested with a standard solution of known nitrate N concentration before installation at the Mullenbach site, and the calibration for both probes was found to be correct. A YSI 6800 Sonde water quality probe was installed to measure pH, specific conductance, and water temperature. Measurements taken by the HACH nitrate N probes and the YSI Sonde were recorded every hour by a Campbell Scientific CR10X datalogger. The locations of the HACH Nitrate N probes and the YSI Sonde are shown in Figure

21. Sample data collected with the YSI Sonde are presented in Figure 23 and Figure 24. Figure 23 shows daily fluctuation in water temperature in the south flume, while Figure 24 shows pH and specific conductance trends before, during and after a rainfall event on August 31-September 1, 2010. An instrumentation error prevented water temperature data from being stored after June 3, 2010. Notice the drops in both pH and specific conductance as discharge increases late on August 31.

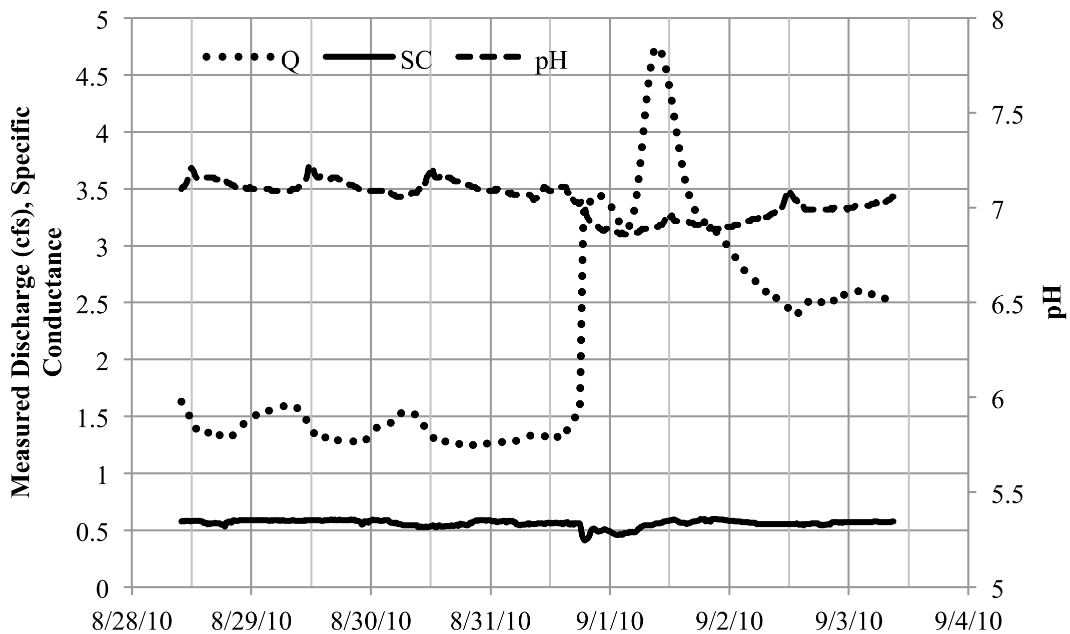

Figure 24. Sample pH and Specific Conductance Data Collected with the YSI Sonde. The Chart Shows pH and Specific Conductance Before, During, and After a Rainfall Event on August 31-September 1, 2010. Shown With Measured Discharge (Q) Through The South Flume.

On several occasions grab samples from running tile outlets, locations along the ditch channel, and seepage water were analyzed on site for nitrate N concentration using the HACH nitrate N probes. Another YSI Sonde was used to measure several water quality parameters (temperature, specific conductance, pH, and dissolved oxygen) on two occasions. A summary of all the water quality analyses performed on different sample sets is given in Table 13.

4.4.2 Laboratory Testing

In addition to on-site testing of various grab samples for nitrate N concentration, many samples were collected in the field and submitted for laboratory analysis. Samples were generally gathered from running tile outlets, the ditch channel, and seepage water observed in the ditch system.

All laboratory analyses were completed at the University of Minnesota Research Analytical Laboratory (UMN-RAL), with the exception of stable isotope analysis, which were analyzed at the University of Minnesota Department of Soil, Water and Climate; and those samples collected on October 21st, 2010, which were analyzed by Pace Analytical Laboratories.

Tests conducted at the UMN-RAL were:

- nitrate – nitrite concentration
- total phosphorus concentration
- pH
- specific conductance, and
- concentrations of various metals (Al, B, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Pb, and Zn)

Nitrate – nitrite testing was conducted by colorimetric analysis by the cadmium reduction method (UMN-RAL, 2011). Total phosphorus was obtained in the following manner:

> "A 10 mL aliquot of a water sample is digested with sulfuric acid and mercuric oxide in a 50mL Folin-Wu tube. The two hour digestion takes place in an electrically heated aluminum block at 380°C. The digested sample is analyzed colorimetrically for total phosphorus as orthophosphate" (UMN-RAL, 2011).

pH was measured using a pH meter equipped with glass and calomel reference electrodes (UMN-RAL, 2011). Specific conductance was measured using a Beckman Solubridge and a conductivity cells with a 1.0 cell constant (UMN-RAL, 2011). The broad list of metals (Al, B, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Pb, and Zn) was analyzed via multi-element ICP atomic emission spectrometry (UMN-RAL, 2011).

Analyses carried out by Pace Analytical Laboratories were:

- Nitrate – nitrite concentration,
- Total phosphorus concentration, and
- Chloride concentration

All measurements conducted at Pace Analytical Laboratories were conducted according to standard methods developed by the American Public Health Association, American Water Works Association, and the Water Environment Federation (Eaton et al., 2005). A summary of all analyzed water quality samples (dates and types of analyses performed) is included in Table 13; the full water quality data set is included in Appendix E.

4.5 Water Stable Isotope Sampling

Many water samples were collected at the Mullenbach ditch site for stable isotope (^{18}O and ^{2}H) analysis. Stable ^{18}O and ^{2}H isotopes have been shown to be useful in gaining information about the different source waters of a system, as well as the pathways and hydraulic residence times (HRT) of those source waters (Simpkins, 1995; Clark and Fritz, 1997; Magner and Alexander, 2008). Stable isotope values are reported as $\delta^{18}O$ and $\delta^{2}H$, which are the deviations (usually given as per mil, ‰) of ^{18}O and ^{2}H concentrations from those of Vienna Standard Mean Ocean Water (VSMOW) (Clark and Fritz, 1997).

Craig (1961) showed that there is a linear relationship between $\delta^{18}O$ and $\delta^{2}H$ in meteoric water (which has not been subject to excessive evaporation) from around the world. This relationship has come to be known as the global meteoric water line (MWL) (Clark and Fritz, 1997). While the relationship developed by Craig provides a reasonable fit for most meteoric water around the world, regional MWLs have been developed to more accurately show the relationship between $\delta^{18}O$ and $\delta^{2}H$ in precipitation on smaller scales. Relationships germane to this study have been developed for Minnesota (Magner and Regan, 1994) and Iowa (Simpkins, 1995).

Clark and Fritz (1997) discussed the relationship between earth surface air temperature and ^{18}O concentration in precipitation water. The authors noted that although there is a strong linear relationship between long-term average ^{18}O concentrations in precipitation and surface air temperature, ^{18}O concentrations within storm events and across small areal scales can vary widely. Following the dependence of $\delta^{18}O$ on temperature, ^{18}O concentration in precipitation will tend to follow a cycle similar to that of average temperature throughout the year. Magner and Alexander (2008) presented an equation, which allows for the computation of HRT based on annual variation in ^{18}O concentrations for precipitation and site water:

$$HRT = \mu^{-1}[(A/B)^2 - 1]^{0.5} \tag{7}$$

where HRT is hydraulic residence time (days), μ is the angular frequency of variation (2π/365), A is the input (precipitation) amplitude, and B is the output (site water) amplitude. The amplitude values are obtained from best-fit isotopic sine curves for precipitation and site water, and are essentially the expected annual ranges of variation for ^{18}O concentration values for precipitation and site water.

Equation (7) shows that HRT is proportional to the ratio (A/B), which means that output waters will tend to have shorter HRTs with increased variation in concentration of ^{18}O in output water throughout the year. This relationship can be useful for determining the HRT for a number of separate water sources at a single site. For instance, the HRTs for groundwater and tile drainage discharge could be determined to gain a better understanding of water source characteristics in a drainage ditch setting.

Stable isotope samples were analyzed by a laboratory in the Department of Soil, Water and Climate at the University of Minnesota. The analysis was conducted using a laser spectroscopy system (LGR Liquid Water Analyzer DLT-100, Los Gatos Research, Inc.) coupled to an autosampler (HT-300A, HTA s.r.l.) for simultaneous $^2H/^1H$ and

$^{18}O/^{16}O$ measurement (N. Schultz, personal communication, June 30, 2011). Please see Appendix D for a full description of the methods used in determining stable isotope concentrations.

4.6 Nitrate N Loading Calculations

Measurements from the installed HACH nitrate N probes were combined with volumetric discharge at both the north and south flumes to calculate nitrate N loading through the ditch from April to October 2010. Unfortunately, several equipment problems limit the amount of useful data. As previously mentioned, there is a period of missing water level data for each flume, such that the only period with continuous data from both sensors is that from June 29 – October 21. Similar technical issues also caused the loss of some data from the HACH nitrate N sensors at the site. Power supply issues at the south flume caused nitrate N concentration to only be logged during daylight hours for much of the season. For this reason, it is not possible to calculate hourly nitrate N loading for the south flume.

The solution is to create a daily average nitrate N concentration from all of the measured values on each day. This average nitrate N concentration are then coupled with average daily discharge to approximate daily nitrate N export through the south end of the ditch system. To calculate the error associated with this simplification, a 10-day period (June 4 – 13) was selected for analysis of data from the north flume, where there were hourly nitrate N concentration and 15-minute discharge measurements available for analysis. First, daily loading was calculated using hourly discharge and nitrate N concentrations. Then, daily loading was estimated with a limited set of data: hourly discharge measurements were averaged to obtain a daily average discharge, and hourly nitrate N concentrations measured from 7 a.m. to 8 p.m. (to mimic the data recorded by the HACH nitrate N probe installed at the south flume) were averaged to obtain an estimated daily average nitrate N concentration. Average daily nitrate N and discharge were then multiplied to calculated daily loading. A comparison of results from the two approaches is given in Table 14.

Table 14. Error in Daily and 10-Day Nitrate N Loading when Calculated with Average Daily Discharge and Nitrate N Concentrations.

Nitrate N (NO_3-N) Loading (kg d^{-1})			
	Calculation Method		
Date	Hourly data	Daily (average) data	Percent error
4-Jun	33.1	33.8	-2.15
5-Jun	55.2	56.0	-1.34
6-Jun	55.8	56.1	-0.631
7-Jun	51.6	51.5	0.107
8-Jun	56.7	56.1	1.03
9-Jun	61.8	62.2	-0.727
10-Jun	66.0	65.5	0.725
11-Jun	105	104	0.652
12-Jun	143	142	0.511
13-Jun	125	126	-0.360
10-day sum	**752.8**	753.0	-0.022

The cumulative 10-day nitrate N loading calculated using hourly data was 752.8 kg, while the load was estimated to be 753.0 kg using average daily discharge and average daily nitrate N concentration, an error of 0.022% for the 10-day period. Seven of the daily loads calculated with daily-averaged data were within 1.00% of the loads calculated with hourly data, while the largest difference in daily loading was 2.15%.

The relatively small errors show that using the daily average values will provide an acceptable approach for calculating nitrate N loading in the south flume.

4.7 Nitrate N Removal Calculations: August, 2010

Determining the amount of nitrate N removed in the ditch reach is an important way of quantifying the performance of the alternative drainage ditch design. It is therefore important to quantify the individual components that contribute nitrate N to the ditch system. With the goal of quantifying nitrate N removal within the ditch, a one-week period in August 2010 was selected for thorough tile outlet discharge and nitrate N concentration monitoring. Tile outlet discharges and nitrate N concentrations were measured on three days (August 3rd, 4th, and 8th).

4.7.1 Water and Nitrate N Mass Balance Approach

For steady-state conditions (neglecting changes in water storage within the ditch reach), the water mass flow rate balance for the ditch reach is defined as:

$$\rho_{ds} Q_{ds} = \rho_{us} Q_{us} + \rho_t Q_t + \rho_{gw} Q_{gw} \tag{8}$$

where Q_{ds} is the volumetric discharge passing through the flume at the downstream end of the ditch, Q_{us} is the volumetric discharge entering the ditch reach through the flume at the upstream end of the ditch reach, Q_t is the cumulative discharge into the ditch from all the tiles between the upstream and downstream flumes, Q_{gw} is the flux into or out of the ditch reach from groundwater sources, and ρ_{ds}, ρ_{us}, ρ_t, and ρ_{gw} are the densities corresponding to the respective discharges. Q_{gw} is defined as positive when the flow of groundwater is into the stream channel. By neglecting changes in density amongst the four water discharge terms, the mass flow rate balance simplifies to a volumetric flow rate balance:

$$Q_{ds} = Q_{us} + Q_t + Q_{gw} \tag{9}$$

Nitrate N inputs (mass loading rate) to the ditch reach can be written as:

$$\dot{N}_{in} = Q_{us}[N]_{us} + Q_t[N]_t + Q_{gw}[N]_{gw} \qquad \text{for } Q_{gw} > 0 \tag{10}$$

$$\dot{N}_{in} = Q_{us}[N]_{us} + Q_t[N]_t \qquad \text{for } Q_{gw} < 0 \tag{11}$$

where $\dot{N}_{in}$ is the cumulative nitrate N load (mass per time)entering the ditch, and $[N]_{us}$, $[N]_t$, and $[N]_{gw}$ are the nitrate N concentrations corresponding to Q_{us}, Q_t, and Q_{gw}, respectively. The $[N]_t$ and $[N]_{gw}$ terms are taken as flow-weighted mean concentrations. Please note that in this paper $\dot{N}$ terms correspond to nitrate N mass flow rates (mass per time), Q terms correspond to water volumetric flow rates (volume per time), and *[N]* terms correspond to nitrate N concentrations (mass per volume). Nitrate N losses from the ditch reach can be written as:

$$\dot{N}_{out} = Q_{ds}[N]_{ds} + \dot{N}_{removed} \qquad \text{for } Q_{gw} > 0 \tag{12}$$

$$\dot{N}_{out} = Q_{ds}[N]_{ds} - Q_{gw}[N]_{chan} + \dot{N}_{removed} \qquad \text{for } Q_{gw} < 0 \tag{13}$$

where $\dot{N}_{out}$ is the cumulative nitrate N load (mass per time) exiting the ditch reach, either by outflow through the south flume, nitrate N removed from stream water by denitrification or assimilation, or groundwater flow out of the channel. For the purposes of calculating nitrate N removal, nitrate N removed via the $\dot{N}_{out}$ pathway is not available for conversion back to an available nitrate N form. $\dot{N}_{out}$ is defined as positive for nitrate N loads exiting the ditch reach. $[N]_{ds}$ is the nitrate N concentration corresponding to Q_{ds}, $[N]_{chan}$ is the nitrate N concentration in the channel, which represents the nitrate N concentration associated with groundwater loss from the ditch reach, and $\dot{N}_{removed}$ is the mass of nitrate N removed by denitrification and/or assimilation. $[N]_{chan}$ is defined as follows:

$$[N]_{chan} = \frac{Q_{us}[N]_{us} + Q_t[N]_t}{Q_{us} + Q_t} \qquad (14)$$

where the nitrate N loads from upstream and tile lines are assumed to be well mixed within the reach.

To remove the Q_{ds} term, Equations (12) and (13) can be rewritten as

$$\dot{N}_{out} = \left(Q_{us} + Q_t + Q_{gw}\right)[N]_{ds} + \dot{N}_{removed} \qquad \text{for } Q_{gw} > 0 \qquad (15)$$

$$\dot{N}_{out} = \left(Q_{us} + Q_t + Q_{gw}\right)[N]_{ds} - Q_{gw}[N]_{chan} + \dot{N}_{removed} \qquad \text{for } Q_{gw} < 0 \qquad (16)$$

Ignoring changes in the storage of nitrate N in the channel (steady-state conditions), it can be shown by conservation of mass that

$$\dot{N}_{in} = \dot{N}_{out} \qquad (17)$$

By combining Equations (10), (11), (15), (16), and (17), a solution for nitrate N removal in the ditch reach for steady-state conditions can be obtained:

$$\dot{N}_{removed} = Q_{us}([N]_{us} - [N]_{ds}) + Q_t([N]_t - [N]_{ds}) + Q_{gw}\big([N]_{gw} - [N]_{ds}\big) \quad \text{for } Q_{gw} > 0 \tag{18}$$

$$\dot{N}_{removed} = Q_{us}([N]_{us} - [N]_{ds}) + Q_t([N]_t - [N]_{ds}) + Q_{gw}([N]_{chan} + [N]_{ds}) \quad \text{for } Q_{gw} < 0 \tag{19}$$

and the nitrate N fraction removed can be defined as the ratio of the amount of nitrate N removed to the net nitrate N inputs to the ditch.

$$f_N = \frac{\dot{N}_{removed}}{\dot{N}_{in}} \tag{20}$$

where f_N is defined as the fraction of nitrate N that enters the system that is removed within the system. Following Equations (9), (18), and (19), there are three unknowns ($\dot{N}_{removed}$, Q_{gw}, and either $[N]_{gw}$ or $[N]_{chan}$, depending on the direction of groundwater flux) and only two equations (Equation (9) and either Equation (18) or (19)). Thus, further information is needed to complete the analysis. Before continuing, it is important to discuss further limitations of the data at hand.

An important step in the analysis is determination of the groundwater flow component, Q_{gw}. Because groundwater flow was not measured, Equation (9) can be used to estimate groundwater flow based on measured Q_{us}, Q_t, and Q_{ds} values. However, an examination of the data shown in Table 15 reveals problems in using this method to calculate the groundwater discharge for August 3rd, 4th, and 8th. The volumetric water balance suggests that groundwater discharge is actually negative (movement out of the stream channel and into the surrounding soil profile) on August 3 and 4, while a positive value is obtained for August 8. These values seem unreasonable, as it is unlikely for the groundwater discharge to change rather drastically (from -0.25 to +0.18) within five days.

Table 15. Average Daily Discharges of Various Flow Components, August 2010.

	Measured				Calculated
Date	Daily precipitation (in)	North flume	South flume	Summation of tile flow	Groundwater Discharge
8/3/10	0	2.28	2.60	0.57	-0.25
8/4/10	0	1.98	2.28	0.43	-0.13
8/5/10	0	1.73	2.06	N/A	N/A
8/6/10	0	1.55	1.84	N/A	N/A
8/7/10	0.14	1.39	1.69	N/A	N/A
8/8/10	0.69	1.21	1.53	0.14	0.18
8/9/10	0.11	2.03	3.05	N/A	N/A

The uncertainty surrounding these data points calls attention to the somewhat poor regression equation that was fit to the stage-discharge measurements in the south flume (see Table 12 and Figure 22). It appears likely that the error in the rating curve results in unreliable estimates of groundwater flow. This prevents the simple computation of groundwater flow using Equation (9). Thus, a more complex solution approach is required to determine the groundwater flow rate.

4.7.2 ^{18}O Mass Balance Approach

The additional data available to determine the groundwater flow are somewhat limited. Temperature data that were collected on August 4th were considered for inclusion to provide a second equation for the two unknowns in the water balance equation. However, the temperature-energy balance would have also introduced an additional unknown, the temperature corresponding to groundwater flow.

The use of stable water isotope data, namely ^{18}O concentrations, also provides a framework for estimating groundwater flux. Clark and Fritz (1997) provide an approach for determining the relative contributions of runoff, soil water, and groundwater following a rainfall event, given as:

$$Q_{tot}[^{18}O]_{tot} = Q_r[^{18}O]_r + Q_s[^{18}O]_s + Q_{gw}[^{18}O]_{gw} \qquad \text{for } Q_{gw} > 0 \tag{21}$$

Where Q is volumetric discharge and $[^{18}O]$ is concentration of ^{18}O in the various flow components; the subscripts *tot*, *r*, *s*, and *gw* represent total streamflow, runoff, soil water, and groundwater flow, respectively. The analysis presented by Clark and Fritz includes a second stable chemical tracer and an volumetric water balance (similar to that of Equation (9)), where the total streamflow is known, and the three-equation set is needed to determine the three components contributing to streamflow. As is discussed later, the use of one tracer is adequate for our study, as both the upstream and tile discharges are known from field measurements.

4.7.2.1 Solution for the Mullenbach Ditch

A variation on the ^{18}O tracer methodology presented by Clark and Fritz (1997) is presented for the Mullenbach two-stage ditch in Equations (22) and (23).

$$Q_{ds}[^{18}O]_{ds} = Q_{us}[^{18}O]_{us} + Q_t[^{18}O]_t + Q_{gw}[^{18}O]_{gw} \qquad \text{for } Q_{gw} > 0 \qquad (22)$$

$$Q_{ds}[^{18}O]_{ds} = Q_{us}[^{18}O]_{us} + Q_t[^{18}O]_t - Q_{gw}[^{18}O]_{chan} \qquad \text{for } Q_{gw} < 0 \qquad (23)$$

where $[^{18}O]_{ds}$, $[^{18}O]_{us}$, $[^{18}O]_t$, and $[^{18}O]_{gw}$ are the ^{18}O concentrations corresponding to the downstream, upstream, tile, and groundwater flows, respectively. Equations (22) and (23) assume a steady-state condition within the system, where there is negligible change in storage within the system with time. Concentrations again correspond to flow-weighted mean concentrations. Two equations are necessary due to of the uncertainty surrounding the direction of groundwater flow in the system. $[^{18}O]_{chan}$ is a flow-weighted mean ^{18}O concentration that is associated with groundwater that is leaving the ditch channel (for the condition where $Q_{gw} < 0$), and is defined as:

$$[^{18}O]_{chan} = \frac{Q_{us}[^{18}O]_{us} + Q_t[^{18}O]_t}{Q_{us}+Q_t} \qquad (24)$$

where the inputs from upstream and tile sources are assumed to be well mixed within the reach. By substituting Equation (9) into Equations (22) and (23), Q_{gw} equations can be defined for four distinct situations that may be encountered in the field data (Table 16).

Table 16. Summary of Solutions for Determining Groundwater Flow for Specific Groundwater and Tile Flow Conditions.

	General case where $Q_t > 0$	Special case where $Q_t = 0$
$Q_{gw} > 0$	$Q_{gw} = \frac{Q_t([O]_{ds} - [O]_t) + Q_{us}([O]_{ds} - [O]_{us})}{[O]_{gw} - [O]_{ds}}$ (25)	$Q_{gw} = \frac{Q_{us}([O]_{ds} - [O]_{us})}{[O]_{gw} - [O]_{ds}}$ (26)
$Q_{gw} < 0$	$Q_{gw} = \frac{Q_t([O]_{ds} - [O]_t) + Q_{us}([O]_{ds} - [O]_{us})}{\left[\frac{Q_t([O]_{ds} - [O]_t) + Q_{us}([O]_{ds} - [O]_{us})}{-(Q_{us} + Q_t)}\right]}$ $= -(Q_{us} + Q_t)$ (27)	$Q_{gw} = \frac{([O]_{ds} - [O]_{us})}{[O]_{us} - [O]_{ds}}$ $= -Q_{us}$ (28)

The use of the appropriate equation from Equations (25) - (28) with Equation (9) provides two equations, but six unknowns (Q_{ds}, Q_{gw}, $[^{18}O]_{ds}$, $[^{18}O]_{us}$, $[^{18}O]_t$, and $[^{18}O]_{gw}$) remain. Estimates of ^{18}O concentrations are therefore needed to solve the equation set to determine the relative contributions from various water sources and establish an estimate of nitrate N removal in the ditch reach.

4.7.2.2 ^{18}O Data Available for Analysis

During the week of August 3rd – 8th, only those water samples collected on August 3rd were submitted for stable isotope analysis. The samples collected on August 3rd were taken from all running tile lines, as well as from in-stream water every 300 ft (91.4 m) along the ditch. The $\delta^{18}O$ values from in-stream samples collected on August 3rd, 2010 are presented in Figure 25. Three samples were also collected from seepage water standing on the ditch benches near the station 30+00. Because the only seepage water samples that were collected were taken from one small area within the ditch, information cannot be reliably extrapolated to the rest of the ditch. Stable isotope concentrations in the sampled seepage water may also have differed from that of the actual seepage water, as the isotopic composition of the water may have been altered through evaporation while the water was ponded on the ditch benches. Furthermore, the pockets of coarse material in the local soils (and within the channel bed, benches, and banks) in the ditch reach complicate the use of isotope samples.

Table 17. Summary of δ^{18}O Data Collected on August 3rd, 2010.

Upstream Station	Downstream Station	Number of Running Tile Lines	$\delta^{18}O_{us}$	$\delta^{18}O_{t}$	$\delta^{18}O_{ds}$
3+00	6+00	3	-7.964	-8.324	-7.765
6+00	9+00	0	-7.765	-	-7.660
9+00	12+00	0	-7.660	-	-8.305
12+00	15+00	0	-8.305	-	-7.942
15+00	18+00	0	-7.942	-	-8.224
18+00	21+00	2	-8.224	-8.599	-8.367
21+00	24+00	0	-8.367	-	-8.553
24+00	27+00	0	-8.553	-	-8.532
27+00	30+00	1	-8.532	-8.440	-8.523
30+00	33+00	2	-8.523	-8.507	-8.464
33+00	36+00	1	-8.464	-8.382	-8.522
36+00	39+00	0	-8.522	-	-7.887
39+00	42+00	0	-7.887	-	-7.693
42+00	45+00	0	-7.693	-	-7.852
45+00	48+00	1	-7.852	-8.331	-7.476
48+00	51+00	1	-7.476	-8.743	-8.642
51+00	54+00	0	-8.642	-	-8.561
54+00	57+00	0	-8.561	-	-8.757
57+00	60+00	1	-8.757	-8.947	-8.538

Reliable data are available to estimate *[^{18}O]$_{us}$, [^{18}O]$_{ds}$,* and *[^{18}O]$_t$*, and are given in Table 17. As discussed above, insufficient field data are available to estimate *[^{18}O]$_{gw}$*. The estimate of $\delta^{18}O_{gw}$ is therefore based on the weighted mean $\delta^{18}O$ that is expected in groundwater discharge at the Mullenbach site. As shown in Equation (7), HRT can be used to estimate temporal variation (amplitude of variation around the annual mean) in $\delta^{18}O$ values that is expected in groundwater discharge. However, the HRT for groundwater in the Mullenbach ditch system is not known, and thus estimates of $\delta^{18}O$ values will be made based on weighted annual $\delta^{18}O$ reported in other studies (discussed in the following section).

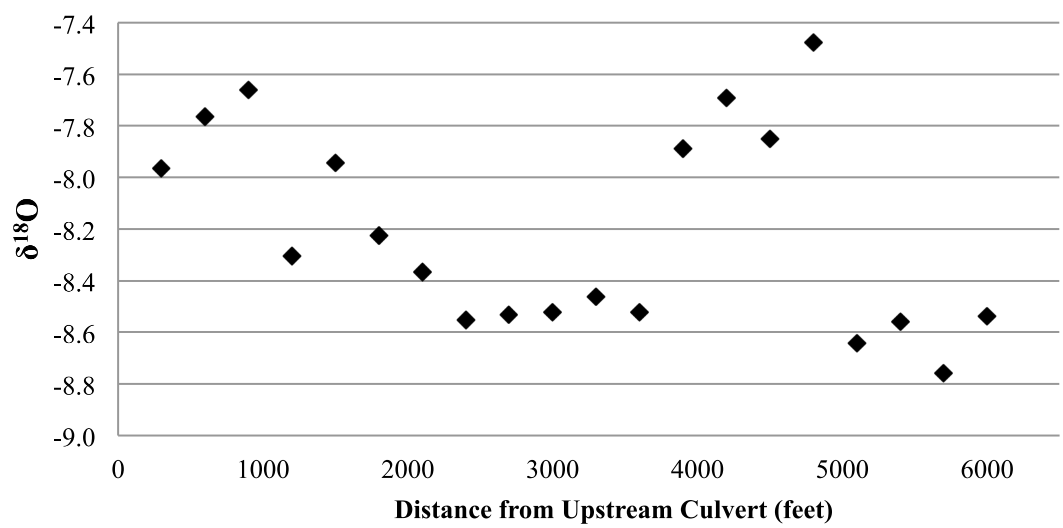

Figure 25. Summary of $\delta^{18}O$ Values from In-Channel Samples Collected on August 3rd, 2010.

4.7.2.3 $\delta^{18}O_{gw}$ Estimation

Simpkins (1995) reported weighted annual mean $\delta^{18}O$ values between -7.24‰ (for Gulf-derived precipitation) and -10.80‰ (for Gulf-Pacific derived precipitation) for central Iowa, USA. As a point of reference, Des Moines, in central Iowa, is approximately 140 miles (220 km) from the Mullenbach ditch site. The significant difference in annual mean values is a result of the variation in precipitation characteristics based on water source and atmospheric effects. Joseph Magner (Minnesota Pollution Control Agency), who has a great deal of familiarity with stable water isotopes in Minnesota, suggests that the values for southern Minnesota may be slightly more negative, at approximately -12‰ to -10‰ (J.A. Magner, personal communication, July, 2011). This is to be expected, as the $\delta^{18}O$ will generally tend to become more negative as distance from the equator increases. Clark and Fritz (1997) presented a map of worldwide average $\delta^{18}O$ values, which suggests that the average is approximately -11‰ for southern Minnesota. Based on the reported values, a $\delta^{18}O$ value of -11‰ is used for all groundwater entering the Mullenbach two-stage ditch system.

4.7.2.4 Preferred Solution Approach

To fully utilize the data collected on August 3rd, an initial solution approach was devised to segment the ditch into 300-ft (91.4 m) sections along the channel's length (following the 300-ft, 91.4 m, sampling interval for $\delta^{18}O$ in the low-flow channel). For the initial segment (3+00 to 6+00), the upstream discharge value, Q_{us}, is taken as equal to the discharge through the upstream flume, which is located at approximately 1+50 (150 ft or 45.7 m upstream of 3+00). $[^{18}O]_{us}$ (at 3+00) and $[^{18}O]_{ds}$ (at 6+00) are known for the reach, as are Q_t and $[^{18}O]_t$. $[^{18}O]_{gw}$ is taken as -11‰, as discussed previously. Q_{gw} and Q_{ds} remain unknown. Q_{gw} is estimated using the appropriate equation from Table 16; Q_{ds} is subsequently calculated using Equation (9).

After all variables are known, the next segment can be analyzed by setting Q_{us} equal to Q_{ds} from the previous segment. Then Q_{gw} and Q_{ds} can be determined for the second segment as they were for the first. This pattern can repeat itself until all flow components (Q_{us}, Q_t, Q_{gw}, and Q_{ds}) are known for each 300-ft (91.4 m) segment. After flow components have been determined for the entire ditch reach, nitrate N removal calculations can be carried out using Equations (18)-(20).

4.7.2.5 Solution Limitations

The framework that has thus far been presented has several drawbacks when applied to the Mullenbach site on August 3, 2010. As Figure 25 shows, $\delta^{18}O$ values in the channel generally become more negative as the distance downstream from the culvert at 120th Street increases (although the values vary widely throughout the reach). The $\delta^{18}O$ values at 3+00 and 60+00 were -7.964‰ and -8.538‰, respectively. Examination of Table 17 shows that $\delta^{18}O$ values for running tile lines are generally only slightly more negative than the values measured for in-channel samples. The flow-weighted mean $\delta^{18}O$ value for all tile flow entering the ditch between 3+00 and 60+00 was -8.528. Given the relatively low influence of tile flow (Q_t:Q_{us} = approximately 1:4) and the similarity between $\delta^{18}O$ values for tile flow and channel flow, it is likely that there is an overall positive groundwater flow component in the ditch, and that the accompanying $\delta^{18}O_{gw}$ value is not insignificantly different from those measured in channel flow.

Further examination of the data in Table 17 shows that on a smaller scale, there are significant fluctuations in $\delta^{18}O$ values along the channel reach. There are several ditch segments (i.e. 3+00 to 6+00 and 30+00 to 33+00) where the $\delta^{18}O$ values for upstream and tile contributions are *both* more negative than the downstream value. Adding to this issue is the fact that $\delta^{18}O$ value for groundwater flow (from our estimate based on literature, and from evidence collected at the site) is estimated as being *even more negative* than channel and tile waters. This shows a fundamental flaw in applying the solution framework outlined here on small spatial scales when the $\delta^{18}O_{gw}$ value is unknown. Similarly, other segments without tile flow contributions (i.e. 12+00 to 15+00 and 36+00 to 39+00) show that the downstream $\delta^{18}O$ value is less negative than both the upstream and estimated groundwater values. This will fundamentally prevent the calculation of groundwater flow based on $\delta^{18}O$ values. For a mass balance of this sort to make sense physically, at least *one* input water source must have a $\delta^{18}O$ value that is *less negative* than that of the outflow, while at least *one* must also have a $\delta^{18}O$ value that is *more negative* than the outflow (assuming that all $\delta^{18}O$ values are not equal).

Furthermore, examination of Equations (27) and (28) shows that the mathematical approach outlined for negative groundwater flows is fundamentally unstable (highlighted by the simplified equation forms given in Table 16), and will not provide a solution for groundwater and downstream flow rates.

One explanation for the difficulty in finding a solution for the groundwater discharge is the possibility of an incorrect choice for the $\delta^{18}O_{gw}$ value. Examination of the 11 300-ft channel sections that have a $Q_t = 0$ shows that six of these segments have a $\delta^{18}O_{ds}$ that is *less negative* than $\delta^{18}O_{us}$. The only way that the $\delta^{18}O_{ds}$ can be less negative than $\delta^{18}O_{us}$ value is for another input to contribute a flow rate that has an associated $\delta^{18}O$ value that is less negative than the $\delta^{18}O_{ds}$ value. This means, in many cases, that the $\delta^{18}O_{gw}$ value has to be less negative than approximately -7‰. This is possible, given that rainfall during warm summer months in Minnesota often has a $\delta^{18}O$ value in the range of -7‰ to -5‰ (J.A. Magner, personal communication, July 2011), and would suggest that the HRT in many parts of the ditch may be quite low.

4.7.2.6 Estimate of Nitrate N Loading and Removal

While $\delta^{18}O$ values measured for channel flow cannot be used to accurately estimate Q_{gw} and Q_{ds} on small scales within the ditch, evidence discussed previously suggests that the estimate of $\delta^{18}O_{gw}$ presented here is reasonable. Due to the overall shift towards more negative $\delta^{18}O$ values as distance downstream increases, it is likely that total groundwater flow in the reach is positive (into the reach). For these reasons, the attempt to quantify water discharges on many small scales will be abandoned, and a single analysis will be done on the entire ditch reach using upstream (3+00), downstream (60+00), and tile outlet flow-weighted mean $\delta^{18}O$ values. Q_{us} and Q_t are known for the reach, and thus Equation (25) can be used to estimate total groundwater flow between 3+00 and 60+00.

This solution approach ignores the small-scale fluctuations in $\delta^{18}O$ value for channel water, and further assumes that $\delta^{18}O$ values for various groundwater sources are equal. Given the information that is available, this will provide an approximation of the overall flow contributions in the ditch reach.

Table 18. Summary of $\delta^{18}O$ Values and Volumetric Discharges for August 3, 2010.

Parameter	Upstream (3+00)	Tile	Groundwater	Downstream (60+00)
$\delta^{18}O$	-7.964	-8.528	-11.000	-8.538
Q, cfs (L s^{-1})	2.28 (64.6)	0.56 (15.9)	0.53 (15.0)	3.38 (95.7)

Table 18 shows the relative contributions of different source waters in the ditch on August 3, 2010. Total groundwater discharge is approximately equal to that of the tile lines (0.53 cfs (15.0 L s^{-1}) and 0.56 cfs (15.9 L s^{-1}), respectively), and upstream discharge accounts for approximately two-thirds of the discharge at the downstream flume.

With the estimates of groundwater and downstream discharge, there are still two unknowns in the remaining nitrate N removal relationships (Equations (18) and (20)), $\dot{N}_{removed}$ and $[N]_{gw}$. A further assumption of groundwater nitrate N concentration, $[N]_{gw}$, is needed to calculate the nitrate N removal within the ditch system.

Because nitrate N concentrations at the upstream end of the ditch were generally measured to be around 20 mg/L during 2010, the upper bound for $[N]_{gw}$ is likely 20 mg/L. The upstream nitrate N concentration is the result of mixing of water from tile and groundwater sources (ignoring surface runoff from rainfall events). Tile water (that was measured within our ditch reach) exhibits a flow-weighted mean nitrate N concentration over 20 mg/L, and for many tile lines approaches 30 mg/L. Because the mixed water that enters the ditch reach through the north flume tends to have a nitrate N concentration around 20 mg/L, it is assumed that the groundwater flow component would be less than 20 mg/L in order to dilute the tile water to a level around 20 mg/L. This assumption is further supported by the longer HRT through the groundwater profile compared to the tile system. The increased HRT may result in increased nitrate N removal within the groundwater profile, and the corresponding nitrate N concentrations in groundwater entering the ditch could be lower than tile water.

Applying the results given in Table 18 to Equations (18) and (20) produces estimates of nitrate N loading and removal, given in Table 19.

Table 19. Nitrate N Loads for the Reach from 3+00 to 60+00 on August 3rd, 2010 ($\delta^{18}O_{gw}$ = -11‰ and $[N]_{gw}$ = 20mg L^{-1}).

	Inflow Components			**Outflow**
	Upstream	Tile	Groundwater	Downstream
Discharge (cfs)	2.28	0.56	0.53	3.38
[NO_3-N] (mg L^{-1})	20.2	21.7	20.0	17.5
Nitrate N Load (kg d^{-1})	112.7	29.84	26.12	144.6
Cumulative Load (kg d^{-1})		**168.6**		**144.6**
Load Removed (kg d^{-1})			**24.1**	
Removal Fraction			**0.143**	

Table 19 shows the nitrate N loads corresponding to each of the flow components in the ditch reach on August 3rd, 2010. The cumulative load from upstream, tile, and groundwater sources is estimated to be 168.6 kg nitrate N d^{-1}, while the estimated nitrate N load carried out of the ditch at the downstream end of the reach is estimated to be 144.6 kg d^{-1}. This results in an estimated nitrate N removal fraction of 0.143, which means that 14.3% of nitrate N that enters the system through the upstream flume, tile flow, and groundwater flow is removed from channel flow by denitrification or assimilation.

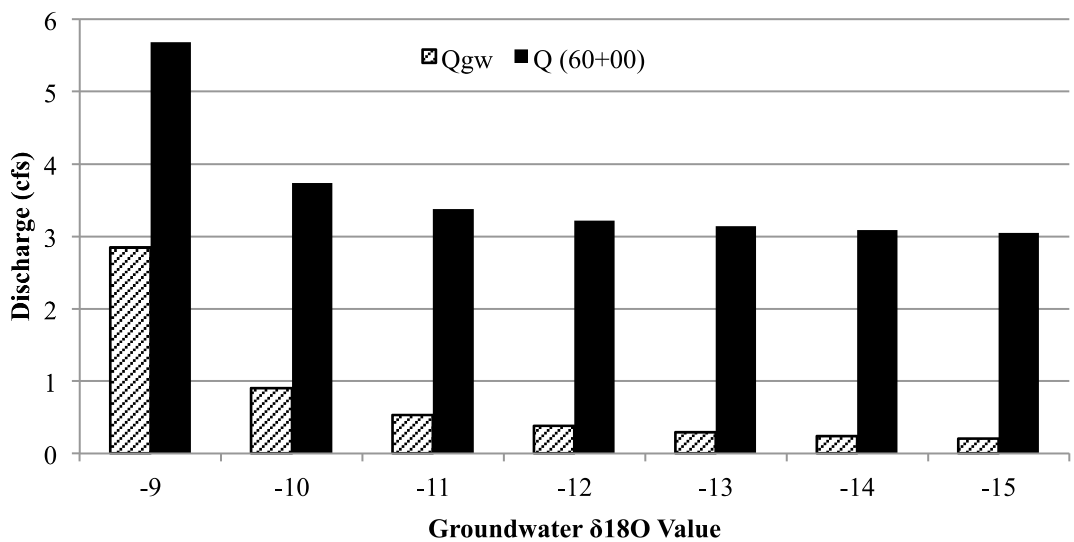

Figure 26. Total Groundwater Discharge and Downstream Channel Discharge at 60+00 as a Function of the Assumed $\delta^{18}O$ Value for Groundwater.

An analysis was done to assess the impact of the uncertainty in the estimates of $\delta^{18}O$ and $[N]_{gw}$. Figure 26 shows the relationship between total groundwater discharge and $\delta^{18}O$ value for groundwater. The total groundwater discharge becomes more sensitive as $\delta^{18}O$ values become less negative. The solution approach breaks down as $\delta^{18}O$

approaches -8‰ because the denominator in Equation (25) approaches zero. The range of groundwater $\delta^{18}O$ values from -11‰ to -15‰ shows little effect on the downstream channel discharge.

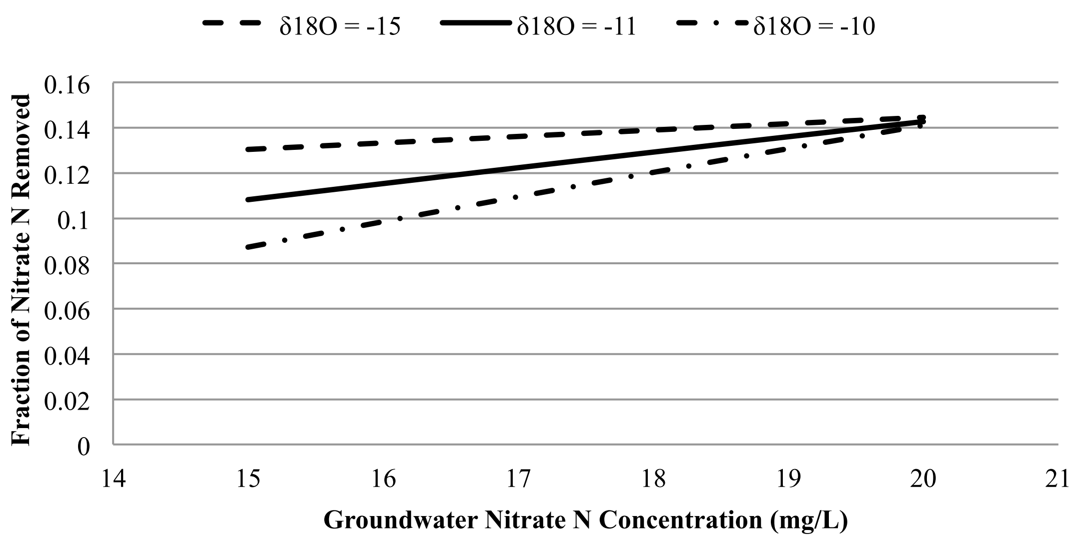

Figure 27. August 3, 2010 Nitrate N Removal Fraction as a Function of $\delta^{18}O$ and Nitrate N Concentration in Groundwater. Each Curve Represents a Distinct Groundwater $\delta^{18}O$ Value, According to the Legend.

Figure 27 shows the impact of $\delta^{18}O_{gw}$ and $[N]_{gw}$ values on nitrate N removal. For $\delta^{18}O$ values that are more negative than -11‰, the nitrate N removal is relatively insensitive to variations in both $\delta^{18}O_{gw}$ and $[N]_{gw}$ values. The removal rates are more sensitive to changes in $[N]_{gw}$ for $\delta^{18}O_{gw}$ values that are less negative than -11‰. Based on the previous discuss of the selection of $\delta^{18}O_{gw}$ and $[N]_{gw}$ values, these results suggest the nitrate N removal in the ditch is likely in the range of 10 to 15 percent.

5 Two-Stage Ditch Economic Analysis

5.1 Mullenbach Two-Stage Ditch Construction Costs

Table 20. Costs of Construction of the Mullenbach Two-Stage Ditch.

Item #	Item description	Units	Quantity	Unit price	Total cost per item
1	Common Channel Excavation (PV)	yd^3	30100	$1.65	$49,665.00
2	Clearing & Grubbing	hours	10.5	$155.00	$1,627.50
3	12" Non-Perforated HDPE (Goldline or Equal)	Linear feet	40	$38.00	$1,520.00
4	8" Perforated HDPE Drain Tile (Goldline or Equal)	Linear feet	120	$36.00	$4,320.00
5	12" CMP	Linear feet	40	$38.00	$1,520.00
6	15" CMP	Linear feet	40	$40.00	$1,600.00
7	18" CMP	Linear feet	160	$50.00	$8,000.00
8	24" CMP	Linear feet	200	$55.00	$11,000.00
9	30" CMP	Linear feet	80	$67.50	$5,400.00
10	24" HDPE Catch Basin, 8 LF w/ Endcap & Trashrack	Each	1	$2,200.00	$2,200.00
11	4'x4' HDPE Anti-Seep Collar	Each	10	$600.00	$6,000.00
12	3'x3' HDPE Anti-Seep Collar	Each	0	$575.00	$0.00
13	12" Tee (Goldline or Equal)	Each	2	$155.00	$310.00
14	12" Reducing Tee (Goldline or Equal)	Each	4	$155.00	$620.00
15	8" HDPE End Cap (Goldline or Equal)	Each	4	$18.00	$72.00
16	3/4" to 1 1/2" Clear Washed Rock	Tons	69.09	$40.00	$2,763.60
17	Mn/DOT CL III Rip Rap	Tons	239.12	$65.00	$15,542.80
18	Mn/DOT Type III Geotextile	yd^2	500	$3.50	$1,750.00
19	Mn/DOT Type II Geotextile	yd^2	215	$3.50	$752.50
20	Seeding For Underneath Blanket	Acres	10	$550.00	$5,500.00
21	Blanket - Category 2 w/ Staple Pattern Designed for 2-1 Slopes	yd^2	48400	$1.35	$65,340.00
22	Sediment Trap Low End & Rip Rap (Maintenance by Hour)	Each	0.5	$3,500.00	$1,750.00
23	Permit & Administration of Paperwork for Diaries	Each	1	$3,500.00	$3,500.00
24	Mobilization	Each	5	$500.00	$2,500.00
25	5'x5' HDPE Anti-Seep Collar	Each	2	$720.00	$1,440.00
26	Seeding 1 Rod Grass Strip w/ Waterway Mix	Acres	4.54	$550.00	$2,497.00
				Total	**$197,190.40**

Construction cost is an important consideration in the implementation of a two-stage drainage ditch design. A detailed list of construction costs associated with the Mullenbach two-stage ditch is given in Table 20. Costs are itemized and provide insight into the various materials that were used to construct the ditch. Costs were also classified into five categories and are presented in Table 21. Total excavation costs account for only \$8.55 linear ft^{-1} (\$28.04 linear m^{-1}). Miscellaneous costs included \$2,500 for mobilization of equipment and \$3,500 for permitting and associated paperwork; these costs could be considered as additional costs associated with excavation, as they likely would have been necessary if only earthwork had been done.

Table 21. Breakdown of Costs for Mullenbach Two-Stage Ditch Construction, Allocated by Project Component.

Description	Total Cost	Cost Linear ft^{-1}	Cost Linear m^{-1}
Excavation	\$51,292.50	\$8.55	\$28.04
Structures (tiles, side inlets)	\$64,810.90	\$10.80	\$35.42
Erosion Control	\$67,090.00	\$11.18	\$36.67
Seeding	\$7,997.00	\$1.33	\$4.36
Miscellaneous	\$6,000.00	\$1.00	\$3.28

Replacement of tile outlets, side inlets, and work related to riprap and geotextile armoring where those structures outlet onto the ditch benches accounted for \$10.80 linear ft^{-1} (\$35.42 linear m^{-1}). Erosion control blankets that were used to cover the ditch benches and outside ditch banks constituted a cost of \$11.18 linear ft^{-1} (\$36.67 linear m^{-1}), the most expensive single project component.

A great amount of caution was used in the selection of erosion control blankets for the site, and due to the ditch's being used as a research site, erosion control provided was more than would otherwise have been needed. Specific numbers have not been researched, but there are other options, such as a straw cover that is crimped into the soil, that are approximately one order of magnitude less expensive than the Category II blankets used for the ditch benches and outside banks. Substantial savings could have been achieved by pursuing different erosion control methods; however in this study it was critical to provide a higher level of protection for demonstration purposes.

Delays in the start date of construction placed further demands on erosion protection. Erosion control blankets for the ditch constituted nearly one-third of construction costs. A more cost-effective method for preventing soil erosion on exposed ditch banks following excavation is needed.

Powell et al. (2007b) reported a range of construction costs from \$10 to \$68 linear ft^{-1} (\$33 to \$218 linear m^{-1}) for four constructed two-stage channels in northwestern Ohio and southeastern Michigan. The cost to construct the Mullenbach ditch system (neglecting design and data collection costs) is approximately \$32 linear ft^{-1} (\$105 linear m^{-1}), which falls within the range of costs reported by Powell et al. (2007b).

5.2 General Two-Stage Ditch Construction Economic Analysis

The general cost-benefit analysis for a two-stage drainage ditch requires consideration of a multitude of factors. The major costs associated with the construction of a two-stage ditch are:

- earthwork required to widen the ditch system,
- erosion control and prevention measures,
- land lost to widening of the ditch system, and
- side inlet and tile outlet modifications and reinforcements.

Potential economic benefits from a two-stage agricultural drainage ditch include:

- reduced ditch maintenance and cleanout frequency,
- water quality benefits,
- improved habitat for both aquatic and terrestrial organisms, and
- a more aesthetically pleasing landscape.

Ditch cleanouts constitute the main cost of maintaining many existing conventional drainage ditch. For this reason, cleanout costs are the main determining factor of the cost effectiveness of a two-stage drainage ditch. Drainage ditches that require relatively frequent cleanouts will tend to be the best candidates for two-stage systems. To provide a method of estimating the financial feasibility of a two-stage drainage project, an economic analysis is presented for determining the economic feasibility of two-stage drainage ditches for a number of site conditions.

The analysis employs a spreadsheet developed by Bill Lazarus in the Department of Applied Economics at the University of Minnesota (B.N. Wilson, personal communication, 2009). The spreadsheet determines the economically preferable ditch system (trapezoidal vs. two-stage) based on a number of factors. Based on the input parameters for a given site, the net present value (NPV) (net value of all future costs expected over the project life) is calculated for both the current trapezoidal ditch system and a potential constructed two-stage ditch. The ditch system with the lower NPV is thereby determined economically preferable.

The input parameters used in this analysis are presented in Table 22. A project life of 100 years is used in this study. A ditch cleanout cost of \$3 linear ft^{-1} (\$9.84 linear m^{-1}) for a conventional ditch in Ohio (before conversion to a two-stage channel) was reported by Powell et al. (2007b). Few other estimates are available that specify the cost of maintenance activities per cleanout event, as most literature reviewed cites annual average ditch maintenance costs for counties or states (i.e. Hansen et al., 2006 and Christner et al., 2004). Peterson et al., (2010) reported cleanout and maintenance costs of \$950 - \$22,000 mi^{-1} (\$590 to \$13670 km^{-1}). The value reported by Powell et al. (2007b) is used in this analysis.

Table 22. Summary of Inputs Used to Determine the Economic Feasibility of Two-Stage Ditch Construction Given Existing Trapezoidal Ditch Conditions.

Variable	Value	Source
Discount rate (interest rate)	Varies (part of analysis)	N/A
	Current ditch design	
Annual Cleanout Cost, \$ linear ft^{-1} (\$ linear m^{-1})	\$3 (\$9.84)	Powell et al. (2007b)
Years before next cleanout	1	Assumed
Cleanout interval (years)	Varies (part of analysis)	N/A
Ditch Width, ft (m)	43 (13.1)	Mullenbach Ditch
	Two-Stage Ditch	
Construction cost	Varies (part of analysis)	USDA-NRCS (2007), Mullenbach Ditch
Value of land taken out of production	Varies (part of analysis)	N/A
Ditch Width, ft (m)	63 (19.2)	Mullenbach Ditch

As discussed previously, two-stage ditch earthwork costs vary widely. A value of \$8.55 ft^{-1} (\$28 m^{-1}) is reported in this study; the USDA-NRCS (2007) reported a range of \$5 to \$20 ft^{-1} (\$16 to \$66 m^{-1}) for the construction of two-stage drainage ditches. The following analysis will be completed for three separate two-stage drainage ditch construction costs: \$5, \$10, and \$15 ft^{-1} (\$16, \$33, and \$49 m^{-1}), which will provide results for a range of construction costs.

As mentioned previously, the erosion control costs following construction of the Mullenbach two-stage ditch could likely have been reduced substantially. Similar erosion control measures are necessary following cleanouts in trapezoidal ditches. Thus, erosion control costs are assumed to be approximately equal for two-stage ditch construction and trapezoidal ditch cleanouts, and are excluded from this analysis. Associated seeding and spraying costs are also assumed to be approximately equal, and are thus excluded from both conventional ditch and two-stage ditch analysis. Furthermore, the Mullenbach ditch site contains many more side inlets and tile outlets than average drainage ditches. For simplification purposes, the construction costs related to side inlets and tile outlets will therefore also be ignored in this analysis. A further assumption will be made that the two-stage ditch will not require cleanout over the project life. The cost of agricultural land taken out of production (due to ditch widening) in the two-stage construction process is accounted for by the change in overall width between the trapezoidal ditch and the two-stage ditch. This analysis is conducted without considering ditch length, as the costs considered are all given in cost per length.

Results are reported as a break-even discount (interest) rate based on the factors outlined in Table 22. The break-even discount rate is the interest rate at which the NPVs of the existing trapezoidal ditch and the proposed two-stage ditch are equal. Therefore, if a discount rate lower than that of the break-even rate can be obtained for two-stage construction, the NPV of the two-stage ditch will be lower than that of the trapezoidal ditch over the project life.

Figure 28 through Figure 30 show the break-even discount (interest) rates for various agricultural land prices, cleanout intervals, and two-stage ditch construction costs. For example, given a two-stage ditch construction cost of \$10 linear ft^{-1} (\$32.80 linear m^{-1}) (Figure 29), adjacent land value of \$3,000 acre^{-1} (\$7,500 ha^{-1}) and an approximate conventional ditch cleanout interval of 10 years, the break-even discount rate is approximately 3%. This means that if a discount rate of less than 3% can be achieved, the two-stage ditch will be economically feasible.

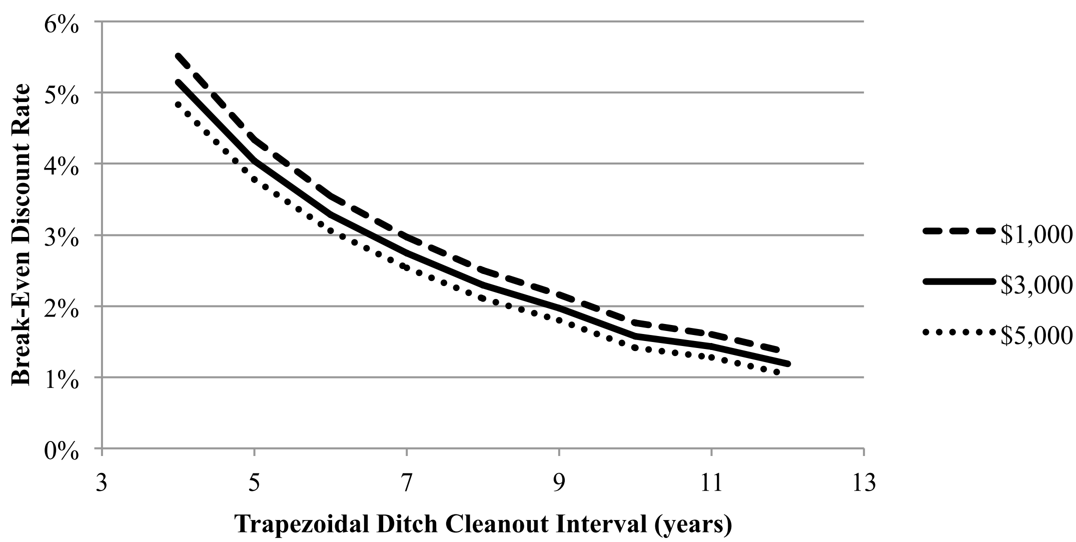

Figure 28. Break-Even Discount Rate vs. Cleanout Interval for Three Adjacent Land Prices and a Two-Stage Ditch Construction Cost of \$15 linear ft^{-1}.

Achievable interest rate will have a large bearing on project feasibility (perhaps more so than cleanout frequency or land price), and small reductions in the interest rate may tip the economic analysis in favor of a two-stage channel in some cases. Because there is considerable uncertainty surrounding the interest rate that is expected for two-stage ditch construction practices, presenting the break-even interest rate based on other factors may be especially useful for considering the feasibility of two-stage ditch projects in the future.

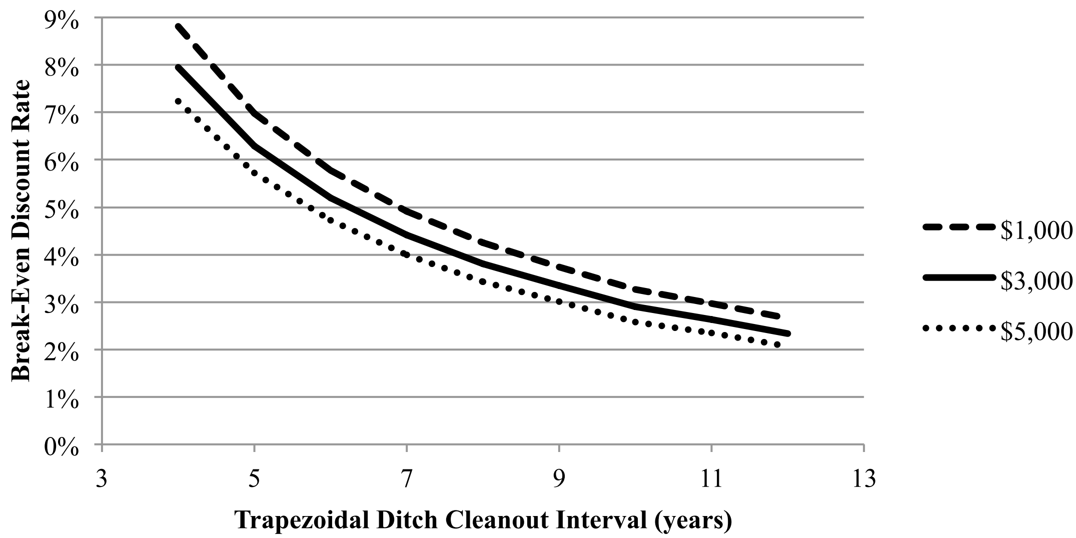

Figure 29. Break-Even Discount Rate vs. Cleanout Interval for Three Adjacent Land Prices and a Two-Stage Ditch Construction Cost of \$10 linear ft^{-1}.

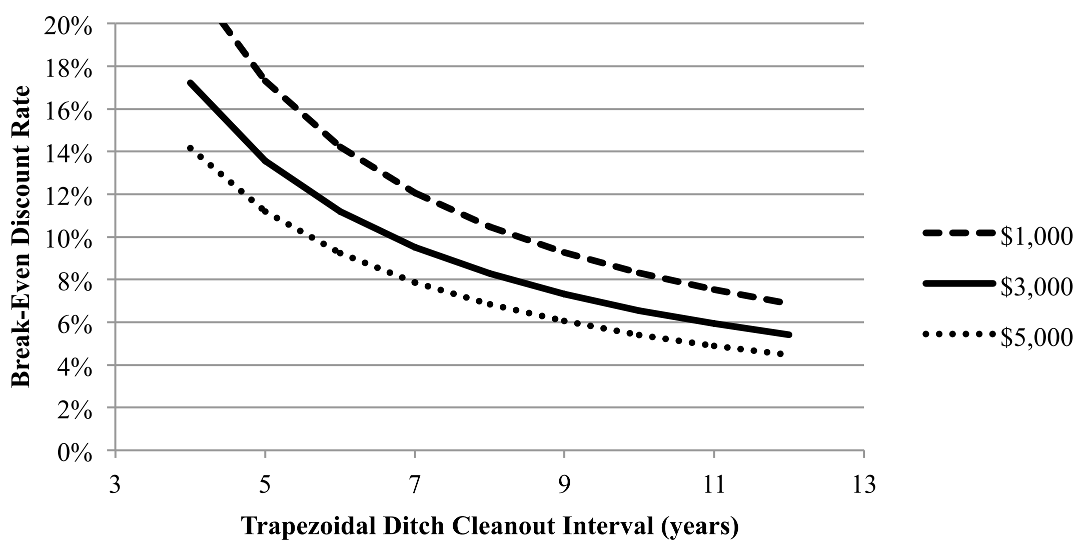

Figure 30. Break-Even Discount Rate vs. Cleanout Interval for Three Adjacent Land Prices and a Two-Stage Ditch Construction Cost of \$5 linear ft^{-1}.

5.3 Expanded Economic Analysis with Subsidies

Based on the initial analysis discussed in Section 5.2, there may be certain settings (such as those where frequent trapezoidal ditch cleanout and maintenance is necessary) where a two-stage ditch will be economically preferable to the existing trapezoidal ditch. However, for situations where a two-stage ditch is initially found to be costlier than a conventional ditch, it is useful to have a method for determining the amount of subsidies necessary to reduce the NPV of the two-stage ditch to a level equal to the NPV of the current trapezoidal ditch. Subsidies based on the potential for increased ditch stability, improved water quality, and enhanced habitat may be available. Subsidies may offset two-stage construction costs enough to reduce the NPV of a two-stage system to a level below that of the existing trapezoidal ditch.

The results presented here are based on the analysis presented in Section 5.2, and the values used are taken from Table 22. The results are given as a break-even subsidy (given in \$ linear ft^{-1} of constructed two-stage ditch), which is the amount of subsidy (paid at the time of construction) necessary to reduce the NPV of the proposed two-stage ditch to equal the NPV of the existing trapezoidal ditch over the project life. If subsidies meeting or exceeding the break-even subsidy amount are available, the two-stage ditch will be economically preferable. To simplify the presentation of results, this analysis assumes an adjacent land value of \$5000 $acre^{-1}$. The break-even subsidy values for various scenarios are presented in Figure 31 through Figure 33.

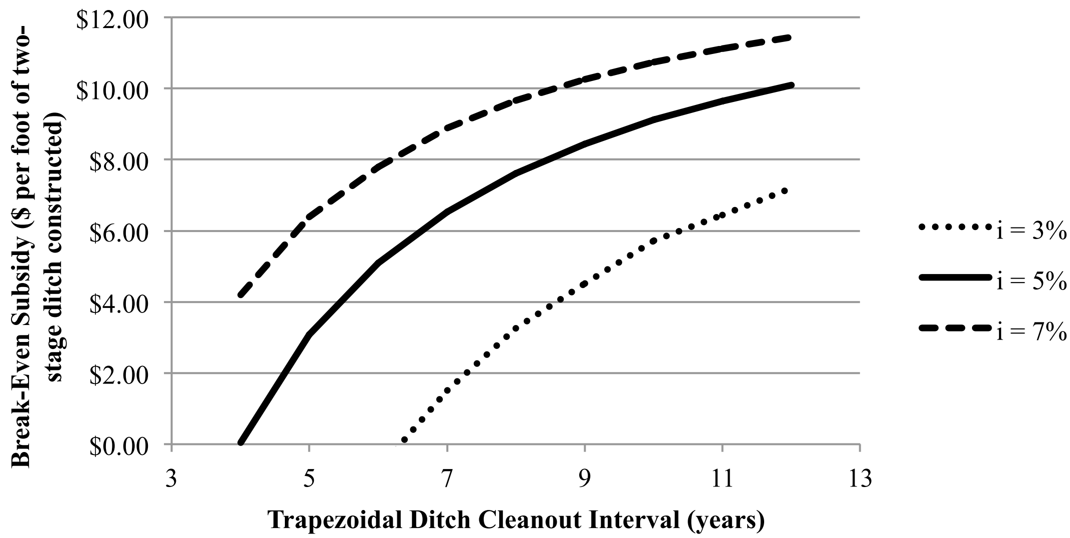

Figure 31. Break-Even Subsidy as a Function of Trapezoidal Ditch Cleanout Interval for Various Discount Rates at a Two-Stage Ditch Construction Cost of \$15 linear ft^{-1} and an Adjacent Land Price of \$5,000 Acre^{-1}.

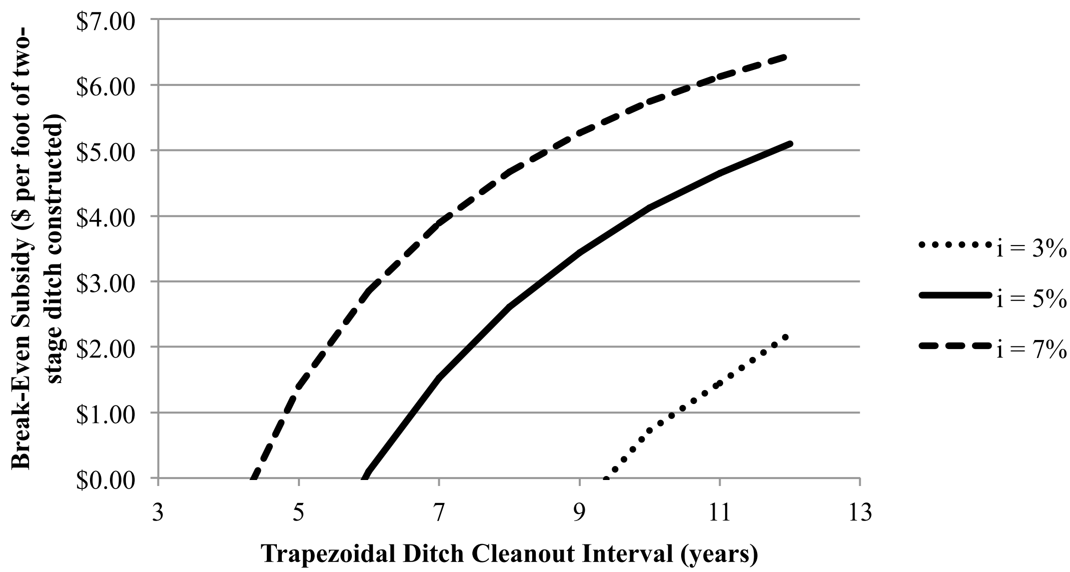

Figure 32. Break-Even Subsidy as a Function of Trapezoidal Ditch Cleanout Interval for Various Discount Rates at a Two-Stage Ditch Construction Cost of \$10 linear ft^{-1} and an Adjacent Land Price of \$5,000 Acre^{-1}.

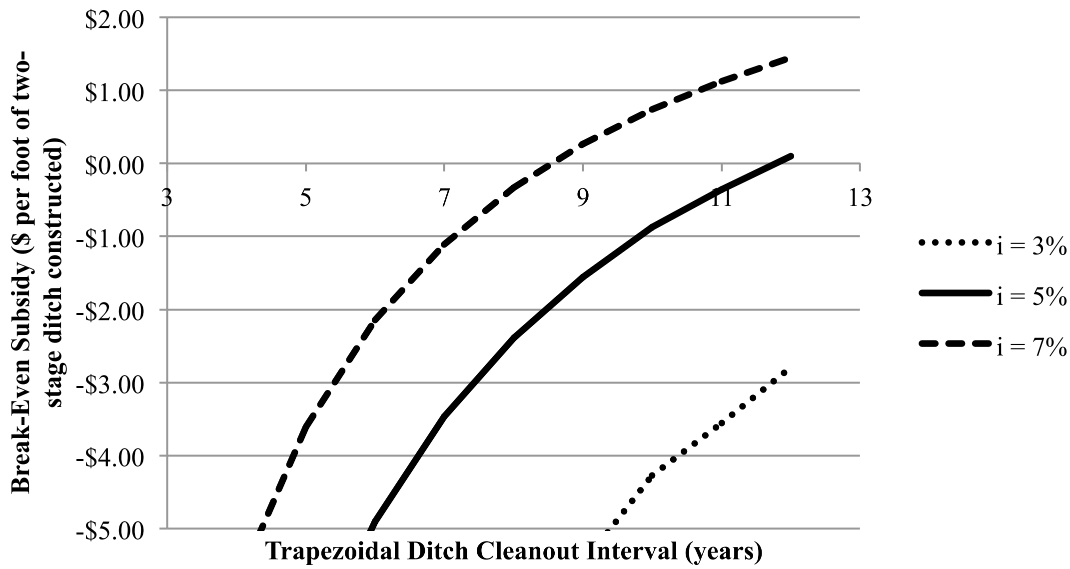

Figure 33. Break-Even Subsidy as a Function of Trapezoidal Ditch Cleanout Interval for Various Discount Rates at a Two-Stage Ditch Construction Cost of \$5 linear ft^{-1} and an Adjacent Land Price of \$5,000 Acre^{-1}.

For example, following Figure 32, the break-even subsidy for a ditch with i = 5%, a trapezoidal ditch cleanout frequency of 9 years, a two-stage construction cost of \$10 linear ft^{-1} (\$32.80 linear m^{-1}), and an adjacent land value of \$5000 acre^{-1} (\$12,500 ha^{-1}) is approximately \$3.50 linear ft^{-1} (\$11.50 linear m^{-1}) of ditch constructed. This corresponds to a subsidy of \$18,480 mi^{-1} (\$11,500 km^{-1}).

5.4 Expanded Economic Analysis with Nitrogen Removal Subsidies

Measureable water quality benefits may provide justification for subsidies in some cases. To demonstrate this point, an additional analysis is presented to demonstrate the potential of water quality improvements, specifically increased N removal, to provide a basis for subsidies. This analysis focuses on N removal as the only justification for subsidies, and will not attempt to account for possible subsidies resulting from the potential for enhanced habitat or other water quality improvements. This analysis continues from the previous analysis by using the break-even subsidy in conjunction with estimates of increased N removal (vs. a conventional ditch) in a two-stage system to calculate the break-even N removal cost ($/kg N removed). This provides a useful framework for comparing the costs of N removal in two-stage ditches to the costs of N removal with other BMPs. If N removal can be achieved less expensively (i.e. more N removed per dollar of subsidy) with a two-stage ditch than with other BMPs, a two-stage ditch will likely be economically feasible if subsidies are available.

The break-even subsidies presented in Section 5.3 (Figure 31 through Figure 33) are essentially equivalent to the NPV of all subsidies required over the project life. This is useful for determining the level of subsidy that may be necessary to justify two-stage construction, but is not useful for determining the cost of N removal (based on the subsidy) on an annual basis. For this reason, Figure 31, Figure 32, and Figure 33 are modified and reprinted here (Figure 34 through Figure 36) to present the *annualized* break-even subsidy.

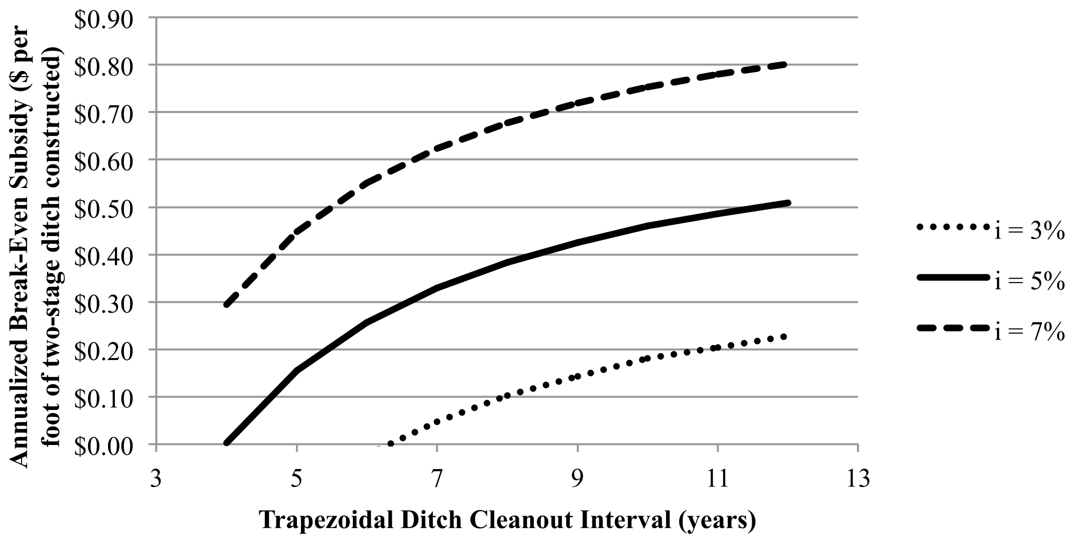

Figure 34. Annualized Break-Even Subsidy as a Function of Trapezoidal Ditch Cleanout Interval for Various Discount Rates at a Two-Stage Ditch Construction Cost of $15 linear ft^{-1} and an Adjacent Land Price of $5,000 Acre^{-1}.

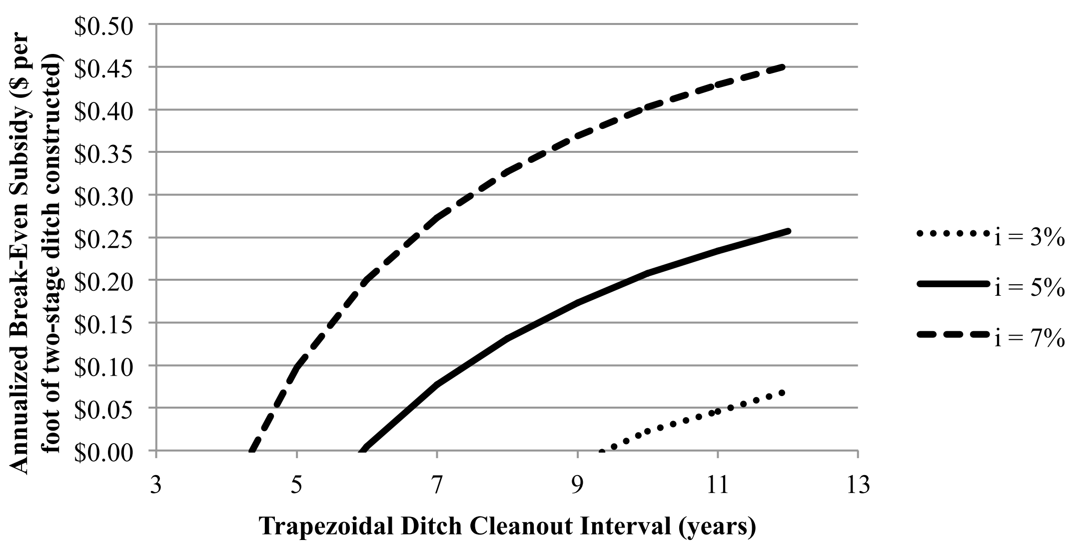

Figure 35. Annualized Break-Even Subsidy as a Function of Trapezoidal Ditch Cleanout Interval for Various Discount Rates at a Two-Stage Ditch Construction Cost of $10 linear ft^{-1} and an Adjacent Land Price of $5,000 Acre^{-1}.

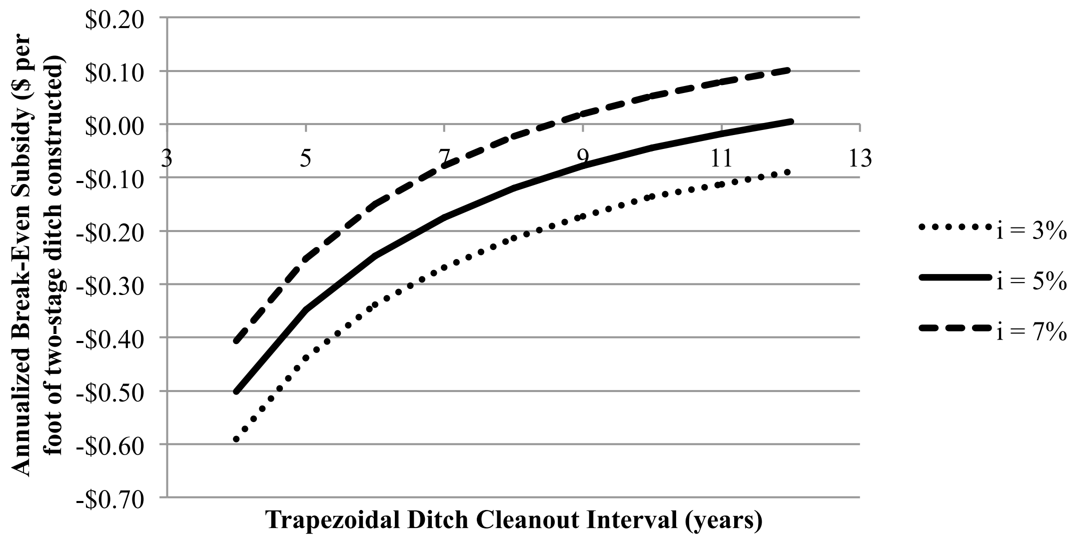

Figure 36. Annualized Break-Even Subsidy as a Function of Trapezoidal Ditch Cleanout Interval for Various Discount Rates at a Two-Stage Ditch Construction Cost of linear ft^{-1} and an Adjacent Land Price of \$5,000 Acre^{-1}.

To illustrate the results shown in Figure 34 through Figure 36, the annualized subsidy (determined from Figure 35) for the conditions discussed in Section 5.3 (i = 5%, trapezoidal ditch cleanout frequency of 9 years, two-stage construction cost of \$10 linear ft^{-1} (\$32.80 linear m^{-1}), and an adjacent land value of \$5000 acre^{-1} (\$12,500 ha^{-1}) is approximately \$0.17 linear ft^{-1} (compared to the NPV of approximately \$3.50 linear ft^{-1} or \$11.48 linear m^{-1} of the break-even subsidy).

The results in Figure 34 through Figure 36 allow for the calculation of N removal costs on an annual basis. The break-even N removal cost is determined as follows:

$$C_{NR} = \frac{S_{ABE}}{D_{marginal}} \tag{29}$$

where C_{NR} is the break-even N removal cost (\$ kg^{-1} of additional N removed in the two-stage ditch), S_{ABE} is the annual break-even subsidy (\$ y^{-1} linear ft^{-1} of two-stage ditch constructed, as presented in Figure 34 through Figure 36), and $D_{marginal}$ is the marginal N removal (kg N removed y^{-1} linear ft^{-1} of two-stage channel constructed) obtained by construction of a two-stage ditch. $D_{marginal}$ is defined as follows:

$$D_{marginal} = D_{2st} - D_{trap} \tag{30}$$

where D_{2st} and D_{trap} are the annual N removal rates (kg N removed y^{-1} linear ft^{-1}) removed achieved with two-stage and trapezoidal ditches, respectively. The following section presents data for the estimation of annual N removal in trapezoidal and two-stage ditches.

5.4.1 Additional Nitrogen Removal in Two-Stage Ditches

Roley et al. (2008), Tank et al. (2009), and Roley et al. (in press) reported results from a study of channel and bank/bench denitrification in a 600 m two-stage ditch reach (Indiana, USA) before and after two-stage channel construction. Measured sediment denitrification rates within the low-flow channel did not change following two-stage construction, and in-channel denitrification rates in the trapezoidal and two-stage systems will be considered equal in this analysis.

There was a small, naturally formed floodplain (756 m^2) present in the existing trapezoidal ditch. Two-stage channel construction increased the floodplain area to 3520 m^2. An estimate of floodplain (bench) denitrification rate was made based on denitrification rates measured in sediment samples taken from the upper 5 cm of the floodplain. Based on the increase in floodplain area, a 400% increase (708 g N d^{-1} vs. 142 g N d^{-1}) in nitrate N removal on ditch floodplains (benches) was estimated following two-stage construction (Roley et al., 2008).

Based on the methods of measuring denitrification, these denitrification rates are only valid when the ditch benches are inundated (overbank flow), and the authors considered denitrification on the ditch benches to be zero during low-flow periods (Roley et al., in press). This estimate of nitrate N removal due to the increased area of the ditch benches is conservative for two reasons: denitrification occurring below a depth of 5 cm on the benches is neglected, and possible removal benefits from the ditch benches under low-flow conditions are neglected. Thus, the reported values provide a conservative estimate of denitrification rates on two-stage ditch floodplains, and corresponding predicted nitrate N removal costs are likely higher than actual costs, and should be taken as an initial estimate only.

Based on the methodology reported by Roley et al. (2008), Tank et al. (2009), and Roley et al. (in press), the annual N removal rate for the trapezoidal ditch floodplain can be defined as:

$$D_{trap} = D_{trap,daily} T_{overbank} \tag{31}$$

where $D_{trap,daily}$ is the daily nitrate N removal (kg nitrate N d^{-1} linear ft^{-1}) in the trapezoidal ditch and $T_{overbank}$ is the annual number of days of overbank (flooded) flow within the ditch system. Annual N removal rate for the two-stage ditch benches is defined as:

$$D_{2st} = D_{2st,daily} T_{overbank} \tag{32}$$

where $D_{2st,daily}$ is the daily nitrate N removal (kg nitrate N d^{-1} linear ft^{-1}) and $T_{overbank}$ is the annual number of days of overbank (flooded) flow within the ditch system. Nitrate N is used as a surrogate for total N in this analysis, and denitrification is assumed to be the dominant nitrate N removal pathway.

Floodplain denitrification rates of 142 g d^{-1} and 708 g d^{-1} were reported for the 600 m ditch section in trapezoidal and two-stage forms, respectively (Roley et al., 2008). This results in denitrification rates of 0.24 g d^{-1} linear m^{-1} (0.072 g d^{-1} linear ft^{-1}) and 1.18 g d^{-1} (0.360 g d^{-1} linear ft^{-1}) linear m^{-1} for the trapezoidal and two-stage ditches, respectively.

Due to a lack of information on N removal in floodplains (benches) during low-flow conditions, the annual number of days of overbank flow plays a major role in the estimation of annual floodplain nitrate N removal rates. Roley et al. (in press) reported 29 and 132 annual days of overbank flow in 2008 and 2009, respectively, at their ditch site in Indiana. These data show that there may be a wide variation in the number of days of overbank flow each year. The analysis given here will focus on three scenarios: 10 days, 30 days, and 100 days (annually) of overbank flow, and the associated N removal costs will reflect those values.

5.4.2 Hypothetical Ditch Analysis

To provide useful and realistic results, sample ditch dimensions will be combined with the data discussed previously to present a hypothetical ditch for calculation of N removal costs. The *hypothetical* ditch dimensions and characteristics illustrate potential economic benefits of N removal in two-stage ditch systems. Characteristics are purely intended as a demonstration of how accounting for additional N removal in a two-stage system may positively impact the economic analysis presented previously. The ditch dimensions and N removal characteristics are defined in Table 23.

Table 23. Dimensions, Nitrate N Removal Rates, and Cleanout Cost of A Hypothetical Ditch for Trapezoidal and Two-Stage Conditions.

Dimension/Characteristic	Conventional Ditch (Before Two-Stage Construction	Two-Stage Channel
Floodplain Width, ft (m)	4.00 (1.22)(1)	20.0 (6.10)(1)
Overall Ditch Width, ft (m)	42.6 (13.0)(2)	58.6 (17.9)(2)
Reach Floodplain Nitrate N Removal, g d^{-1} linear ft^{-1} (g d^{-1} linear m^{-1})	0.072 (0.24)(1)	0.360 (1.18)(1)
Marginal Nitrate N Removal due to Two-Stage Channel, g d^{-1} linear ft^{-1} (g d^{-1} linear m^{-1})	0.29 (0.96)(1)	
Conventional Ditch Cleanout Cost, \$ linear ft^{-1} (\$ linear m^{-1})	3.00 (9.84)(2)	-
Two-Stage Ditch Construction Cost, \$ linear ft^{-1} (\$ linear m^{-1})	-	10.00 (32.80)(2)
Discount Rate i	3%, 5%, and 7%(2)	
Cleanout Interval (y)	Varies as part of analysis(2)	
Two-Stage Channel Construction Cost, \$ linear ft^{-1} (\$ linear m^{-1})	5, 10, and 15 (16.4, 32.8, and 49.2)(2)	
Annual Days of Overbank Flow (d)	10, 30, and 100(2)	
Adjacent Land Value, \$ $acre^{-1}$ (\$ ha^{-1})	5,000 (12,500)(2)	

(1) estimates based on values presented by Roley et al. (2008), Tank et al. (2009), and Roley et al. (in press) as previously discussed. (2) values based on literature cited in this report, previously discussed values in this report, or on experience with the Mullenbach ditch site.

Due to the large number of variables included in this analysis, results are presented in tabular form (Table 24), rather than in a series of graphs. The values presented in Table 23 may be combined with Equation (29) (where the S_{ABE} value can be determined using Figure 34 through Figure 36) to determine the break-even N removal cost for a number of scenarios to aid in future two-stage ditch design projects. Certain discount rate/two-stage ditch construction cost combinations are not represented in the results of this analysis due to the fact that those scenarios corresponded to ditch settings that were economically feasible without subsidies.

Table 24. Break-Even Nitrogen Removal Costs for Two-Stage Ditches Constructed in Various Settings.

Discount Rate	Cleanout Interval, years	Two-Stage Channel Construction Cost, \$ linear ft^{-1} (\$ linear m^{-1})	Annual Days of Overbank Flow, days	Break-Even Nitrogen Removal Cost (\$ kg N^{-1} removed)
5%	12	5 (16.40)	100	\$0.18
3%	12	5 (16.40)	30	\$0.60
7%	10	5 (16.40)	100	\$1.81
3%	12	10 (32.80)	100	\$2.42
7%	5	10 (32.80)	100	\$3.39
3%	11	10 (32.80)	30	\$5.30
3%	7	15 (49.20)	30	\$5.52
5%	7	10 (32.80)	30	\$8.92
7%	12	5 (16.40)	30	\$11.72
5%	6	15 (49.20)	30	\$29.73
5%	11	15 (49.20)	10	\$168.78

Table 24 shows that there are certain situations (mostly corresponding to lower discount rates, more annual days of overbank flow, and lower two-stage ditch construction costs) where the break-even N removal costs in two-stage ditches are rather low (\$3 to \$4 kg^{-1} of additional N removed or less). Conversely, there are many situations where the break-even price of N removal is very high. These cases correspond to situations with high two-stage ditch construction costs, fewer annual days of overbank flow, and generally high discount rates. The cases corresponding to low N removal costs are those situations where the NPV of two-stage ditch costs was only slightly higher than that of the existing trapezoidal ditch system.

For purposes of comparison, literature values were obtained for N removal costs from various proposed projects and BMPs, and are presented in Table 25. Edge-of-field N-loss reduction costs reported by The National Science and Technology Council Committee on Environment and Natural Resources (CENR) (2000) were as low as \$0.88 kg^{-1} N removed, and show an exponential increase in removal cost as load reduction targets increase. CENR (2000) also showed that a 20% reduction in N loads could be achieved at a cost of just \$0.69 kg^{-1} N removed through reduced fertilizer use alone. N removal cost associated with wetland restoration in the Mississippi River basin would be \$6.06 kg^{-1} N removed for the first one million acres restored. It is important to point out that this is simply an average N removal cost for the most cost-effectively restored one million acres, and that there are likely many areas where wetlands could reduce N loading for far less than \$6.06 kg^{-1}. Several values from this study are of the same magnitude as many of the other costs presented here.

The results presented here, although approximations, suggest that some two-stage ditch projects may be able to compete with other BMPs and large-scale practices. Interest rates play a large part in determining the economic feasibility of a two-stage drainage ditch. However, even at higher interest rates there are likely ditches (such as those with short cleanout intervals or low land adjacent land values) that may provide the right economic conditions for cost savings and nutrient reductions through the implementation of a two-stage drainage ditch.

Table 25. Comparison of Nitrogen Removal Cost for Various Treatments and BMPs.

Nitrogen Reduction Method	Net Cost Within Agricultural Sector ($ kg^{-1} N removed)
Edge-of-field N-loss reductions[1]	
20% load reduction	0.88
30% load reduction	1.90
40% load reduction	3.37
50% load reduction	5.20
60% load reduction	7.48
Through reductions in fertilizer use alone [1]	
20% load reduction	0.69
45% load reduction	2.85
500% fertilizer tax	14.54
Wetland restoration in the Mississippi basin[1]	
1,000,000 acres	6.06
5,000,000 acres	8.90
10,000,000 acres	10.57
18,000,000 acres	11.93
Other methods	
Riparian buffers (27,000,000 acres)[1]	26.03
River diversion to coastal wetlands[1]	~6
Wastewater nitrogen removal[1]	~40
407,000 acres of wetland restoration in the Illinois River basin[2]	0.60
Hennepin Levee District floodplain restoration, Illinois[2]	2.87

(1) taken from CENR (2000), (2) taken from TWI (2001)

6 Summary and Conclusions

A two-stage agricultural drainage ditch was designed, and then constructed at the site of an existing trapezoidal ditch in southern Minnesota, USA during the autumn of 2009. The two-stage drainage was dimensioned to mimic the hydraulic geometry found in natural streams and stable ditches in the surrounding area, with the hope of creating a more fundamentally stable ditch than the existing trapezoidal drainage ditch. The two-stage ditch system has the potential to:

1) improve water quality through increased nutrient assimilation and processing;

2) enhance biological function by providing a more natural stream through better sediment sorting, deeper water at low flow levels, and improved water quality; and

3) significantly reduce maintenance costs by creating a self-sustaining ditch where net sediment deposition is negligible, seepage forces on outer banks are removed by dewatering, and outer ditch banks are not exposed to water during periods of low flow.

Field monitoring studies were conducted following ditch construction. Geomorphic aspects of the field study have focused on stability through the analysis of repeat cross-section surveys at seven points throughout the project reach. Longitudinal channel thalweg profile surveys have been conducted to gain insight into the fluvial interactions with the channel substrate, which shapes the channel and can determine biological function based on the characteristics of pool-riffle sequences.

Geomorphic monitoring data show that the channel is making small adjustments, and that there has been a large development of pool-riffle sequences throughout the ditch reach. Continued cross-section monitoring shows little change to the ditch benches and outer banks, while there have been some changes seen in the low-flow channel geometry.

Water quality and quantity monitoring has focused on flow rates and nitrate N concentration measurements in outlet water from subsurface drainage tiles, channel water, and groundwater seepage samples along the ditch benches. Water stable isotope sampling and analysis have also been carried out to help determine the magnitudes of source waters that enter the ditch system.

Volumetric-discharge and water-quality monitoring was done during the period from April through October 2010. Issues with data collection equipment, installed nitrate N sensors, and accurate discharge measurements led to a limited data set for the year 2010. An analysis of data from August 3rd, 2010 showed that nitrate N removal within the ditch system was likely between 10 and 15 percent of that input to the system.

Economic analyses have focused on developing relationships and curves for use in determining the economic viability of future two-stage ditch projects. Results suggest that two-stage ditches can compete with other BMPs and restoration projects for nitrogen removal cost, and may be economically preferable to trapezoidal ditches, even without accounting for possible subsidies.

The economic and water quality/quantity analyses conducted thus far are only approximations, but data collection is ongoing and will continue to provide insights on this complex system. Additional work is needed to further measure water quality benefits of the Mullenbach two-stage ditch. A priority should be placed on accurately determining water sources and pathways through the ditch system, in order to quantify the respective fluxes of surface water, subsurface drainage, and groundwater into the ditch. A robust nitrate N balance could be carried out once a sufficient method for determining the volumetric water fluxes is in place.

Performance of innovative two-stage ditch channel features presented in this paper (linear treatment structure, seepage trench, and the rock trench side inlet) has not thus far been measured. Further study is necessary to assess the potential of these features to improve drainage ditch stability and performance.

Establishing definitive results surrounding the economic costs and benefits of two-stage ditch systems should also be a priority. To this end, obtaining an accurate measure of nitrate N removal, as well as water quality and habitat benefits, will be important in allowing policy makers and engineers to properly gauge the efficacy of two-stage ditches as a BMP. Long-term studies are also needed to determine if two-stage ditches can deliver predicted reductions in channel maintenance and cleanout.

7 References

Alexander, R.B., R.A. Smith, and G.E. Schwarz. 2000. Effect of Stream Channel Size on the Delivery of Nitrogen to the Gulf of Mexico. Nature 403(6771): 758-761.

Alexander, R.B., R.A. Smith, G.E. Schwarz, E.W. Boyer, J.V. Nolan, and J.W. Brakebill. 2008. Differences in Phosphorus and Nitrogen Delivery to the Gulf of Mexico from the Mississippi River Basin. Environmental Science and Technology 42: 822-830.

Appelboom, T.W., and J.L. Fouss. 2006. Methods for Removing Nitrate Nitrogen from Agricultural Drainage Waters: A Review and Assessment. Presented at the 2006 ASABE Annual International Meeting. ASABE Paper #062328. American Society of Agricultural and Biological Engineers. St. Joseph, Mich., USA.

Beauchamp, K.H. 1987. A history of drainage and drainage methods, in Pavelis, G.A., ed., Farm drainage in the United States - History, status, and prospects. Miscellaneous Publication no.1455: p13-29. Economic Research Service, U.S. Department of Agriculture. Washington, D.C., USA.

Blann, K.L., J.L. Anderson, G.R. Sands, and B. Vondracek. 2009. Effects of agricultural drainage on aquatic ecosystem: a review. Critical Reviews in Environmental Science and Technology 39(11): 909-1001.

Blue Earth County Minnesota. 2008. Blue Earth County Water Plan 2008-2013. Priority Concern: County Drainage Ditches. Retrieved on July 2, 2011 at: http://www.co.blue-earth.mn.us/documents/waterplan_files/ditches.pdf

CENR [The National Science and Technology Council Committee on Environment and Natural Resources]. 2000. An Integrated Assessment: Hypoxia in the Northern Gulf of Mexico. National Science and Technology Council Committee on Environment and Natural Resources, Washington, D.C., USA.

U.S. Census Bureau [U.S. Department of Commerce Census Bureau]. 1984. 1982 Census of Agriculture. U.S. Government Printing Office, Washington, D.C., USA.

Christner, W.T., J. Magner, E.S. Verry, and K.N. Brooks. 2004. Natural channel design for agricultural ditches in SW Minnesota. Proceedings of Self-sustaining Solutions for Streams, Wetlands, and Watersheds. American Society of Agricultural Engineers. St. Joseph, Mich., USA.

Clark, I., and P. Fritz. 1997. Environmental Isotopes in Hydrogeology. Boca Raton, FL: CRC Press.

Craig, H. 1961. Isotopic Variation in Meteoric Waters. Science 133(3465): 1702-1703.

David, M.B., L.E. Drinkwater, and G.F. McIsaac. 2010. Sources of Nitrate Yields in the Mississippi River Basin. Journal of Environmental Quality 39: 1657-1667.

Eaton, A.D., L.S. Clesceri, E.W. Rice, and A.E. Greenberg. 2005. Standard methods for the examination of water and wastewater. Baltimore, MD: Port City Press, 2005.

Fouss, J.L., and R.C. Reeve. 1987. Advances in Drainage Technology: 1955-85, in Pavelis, G.A., ed., Farm drainage in the United States - History, status, and prospects. Miscellaneous Publication no.1455: p13-29. Economic Research Service, U.S. Department of Agriculture. Washington, D.C., USA.

Goolsby, D.A., W.A. Battaglin, B.T. Aulenbach, and R.P. Hooper. 2000. Nitrogen Flux and Sources in the Mississippi River Basin. Science of the Total Environment 248(2-3): 75-86.

Hansen, B.J., B.N. Wilson, J. Magner, and J. Nieber. 2006. Geomorphic characteristics of drainage ditches in southern Minnesota. Presented at the 2006 ASABE Annual International Meeting. ASABE Paper #062319. American Socity of Agricultural and Biological Engineers. St. Joseph, Mich., USA.

Herzon, I., and J. Helenius. 2008. Agricultural drainage ditches, their biological importance and functioning. Biological Conservation 141: 1171-1183.

Jayakaran, A.D., and A.D. Ward. 2007. Geometry of inset channels and the sediment composition of fluvial benches in agricultural drainage systems in Ohio. Journal of Soil and Water Conservation 62(4): 296-307.

Kallio, R., A. Ward, J. D'Ambrosio, and J.D. Witter. 2010. A Decade Later: The Establishment, Channel Evolution, and Stability of Innovative Two-Stage Agricultural Ditches in the Midwest Region of the United States. Presented at the 9th International Drainage Symposium. Paper #IDS-CSBE-100209. American Society of Agricultural and Biological Engineers. St. Joseph, Mich., USA.

Karr, J.R. 1981. Assessment of Biotic Integrity Using Fish Communities. Fisheries 6(6): 21-27.

Karr, J.R., K.D. Fausch, P.L. Angermeier, P.R. Yant, and I.J. Schlosser. 1986. Assessing Biological Integriety in Running Waters: A Method and Its Rationale. Illinois Natural History Survey Special Publication No. 5. Champaign, Ill., USA.

King, K.W., P.C. Smiley, Jr., and N.R. Fausey. 2009. Hydrology of channelized and natural headwater streams. Hydrological Sciences 54(5): 929-948.

Landwehr, K., and B.L. Rhoads. 2003. Depositional response of a headwater stream to channelization, East Central Illinois, USA. River Research and Applications 19: 77-100.

Magner, J.A., G. Johnson, and T. Larson. 1993. The Minnesota River Basin: Environmental Impacts of Basin-Wide Drainage, in Volume 5 of Industrial and Agricultural Impacts of the Hydrologic Environment, Eckstein and Zaporozec, ed. p.147-162. Alexandria, VA: Water Environment Federation.

Magner, J.A., and C.P. Regan. 1994. Tools and Techniques for the Assessment of Ground Water/Surface Water Interactions in Glacial Hydrogeologic Settings. Ground Water Management Series (National Ground Water Assocation (NGWA)) 18: 685-698.

Magner, J.A., and K.N. Brooks. 2007. Stratified Regional Hydraulic Geometry Curves: A Water Quality Management Tool. Hydrologic Science and Technology 23: 159-172.

Magner, J.A., and S. Alexander. 2008. Drainage and nutrient attenuation in a riparian interception-wetland: southern Minnesota, USA. Environmental Geology 54: 1367-1376.

Magner, J.A., B. Hansen, T. Sundby, G. Kramer, B. Wilson, and J. Nieber. 2011. Channel evolution of Des Moines Lobe till drainage ditches in southern Minnesota (USA). (in review).

Mecklenburg, D. 1998. Stream Morphology - Modules. Ohio Department of Natural Resources. Retrieved July 8, 2011 at: http://www.dnr.state.oh.us/soilandwater/water/streammorphology/default/tabid/9188/Default.aspx

Mecklenburg, D., and A. Ward. 2004. STREAM modules: Spreadsheet tools for river evaluation, assessment, and monitoring. Proceedings of the ASAE Specialty Conference: Self-Sustaining Solutions for Streams, Watershed, and Wetlands. p.312-322. American Socity of Agricultural Engineers. St. Joseph, Mich., USA.

Mitsch, W.J., J.W. Day, Jr., J.W. Gilliam, P.M. Groffman, D.L. Hey, G.W. Randall, N. Wang. 2001. Reducing Nitrogen Loading to the Gulf of Mexico from the Mississippi River Basin: Strategies to Counter a Persistent Ecological Problem. BioScience 51(5): 373-388.

MnDOT [Minnesota Department of Transportation]. 2005. Mn/DOT Standard Specifications for Construction. Retrieved on July 8, 2011 at: http://www.dot.state.mn.us/pre-letting/spec/2005/3835-3973.pdf#ecb

Omernik, J.M. 1979. Non-Point Source Stream Nutrient Level Relationships: A Nationwide Study. Corvallis, OR: U.S. Environmental Protection Agency.

Osterman, L.E., R.Z. Poore, P.W. Swarzenski, and R.E. Turner. 2005. Reconstructing a 180 yr record of natural and anthropogenic induced low-oxygen conditions from Louisiana continental shelf sediments. Geology 33: 329-332.

Peterson, J., B.N. Wilson, and G. Kramer. 2010. Two-Stage Ditch Assessment using the CONCEPTS Model. Presented at the 2010 ASABE Annual International Meeting. ASABE Paper No. 1009158. American Society of Agricultural and Biological Engineers. St. Joseph, Mich., USA.

Powell, G.E., A.D. Ward, D.E. Mecklenburg, and A.D. Jayakaran. 2007a. Two-stage channel systems: Part 1, a practical approach for sizing agricultural ditches. Journal of Soil and Water Conservation 62(4): 277-286.

Powell, G.E., A.D. Ward, D.E. Mecklenburg, J. Draper, and W. Word. 2007b. Two-stage channel systems: Part 2, case studies. Journal of Soil and Water Conservation 62(4): 286-296.

Powell, K.L., and V. Bouchard. 2010. Is denitrification enhanced by the development of natural fluvial morphology in agricultural headwater ditches? Journal of the North American Benthological Society 29(2): 761-772.

Rabalais, N.N., R.E. Turner, D. Justic, Q. Dortch, W.J. Wiseman, Jr., and B.K.S. Gupta. 1996. Nutrient Changes in the Mississippi River and System Responses on the Adjacent Continental Shelf. Estuaries 19(2B): 386-407.

Rabalais, N.N., R.E. Turner, W.J. Wiseman, Jr., and Q. Dortch. 1998. Consequences of the 1993 Mississippi River flood in the Gulf of Mexico. Regulated Rivers: Research and Management 14: 161-177.

Rabalais, N.N., R.E. Turner, and W.J. Wiseman, Jr. 2001. Hypoxia in the Gulf of Mexico. Journal of Environmental Quality 30: 320-329.

Rabalais, N.N., R.J. Díaz, L.A. Levin, R.E. Turner, D. Gilbert, and J. Zhang. 2010. Dynamics and distribution of natural and human-caused hypoxia. Biogeosciences 7: 585-619.

Randall, G.W., and D.J. Mulla. 2001. Nitrate Nitrogen in Surface Waters as Influenced by Climatic Conditions and Agricultural Practices. Journal of Environmental Quality 30: 337-344.

Reddy, K.R., R.H. Kadlec, E. Flaig, and P.M. Gale. 1999. Phosphorus Retention in Streams and Wetlands: A Review. Critical Reviews in Environmental Science and Technology 29(1): 83-146.

Rhoads, B.L., and K.D. Massey. 2010. Flow structure and channel change in a sinuous grass-lined stream within an agricultural drainage ditch: implications for ditch stability and aquatic habitat. River Research and Application. published online: doi: 10.1002/rra.1430

RiverMorph, LLC. 2010. RIVERMorph: Stream Restoration Software v4.3. RiverMorph, LLC., Louisville, Kent., USA.

Robins, C.R., and R.W. Crawford. 1954. A Short Accurate Method for Estimating the Volume of Stream Flow. The Journal of Wildlife Management 18(3): 366-369.

Roley, S.S., J.L. Tank, L.T. Johnson, J.J. Beaulieu, and M.L. Stephen. 2008. Two-stage ditch restoration: a potential strategy for nitrogen removal from agricultural streams. Presented at the 56th Annual Meeting of the North American Benthological Society. Abstract retrieved on July 8, 2011 at: http://nabs.confex.com/nabs/2008/techprogram/P3146.HTM

Roley, S.S., J.L. Tank, M.L. Stephen, L.T. Johnson, J.J. Beaulieu, and J.D. Witter. 2011. Floodplain restoration enhances denitrification and reach-scale nitrogen removal in an agricultural stream. Ecological Applications. (in press).

Rosgen, D.L. 2006. Watershed Assessment of River Stability and Sediment Supply (WARSSS). Pagosa Springs, Colorado: Wildland Hydrology.

Shields, F.D., Jr., R.R. Copeland, P.C. Klingeman, M.W. Doyle, and A. Simon. 2003. Design for Stream Restoration. Journal of Hydraulic Engineering 129(8): 575-584.

Simpkins, W.W. 1995. Isotopic composition of precipitation in central Iowa. Journal of Hydrology 172: 185-207.

Skaggs, R.W., M.A. Youssef, G.M. Chescheir, and J.W. Gilliam. 2005. Effect of Drainage Intensity on Nitrogen Losses From Drained Lands. Transactions of the American Society of Agricultural Engineers 48(6): 2169-2177.

Smedema, L.K., W.F. Vlotman, and D.W. Rycroft. 2004. Modern Land Drainage - Planning Design and Management of Agricultural Drainage Systems. Leiden, The Netherlands: A.A. Balkema Publishers.

Smiley, P.C., Jr., R.B. Gillespie, K.W. King, and C. Huang. 2008. Contribution of habitat and water quality to the integrity of fish communities in agricultural drainage ditches. Journal of Soil and Water Conservation 63(6): 218A-219A.

Smith, D.R., and E.A. Pappas. 2007. Effect of ditch dredging on the fate of nutrients in deep drainage ditches of the Midwestern United States. Journal of Soil and Water Conservation 62(4): 252-261.

SonTek/YSI. SonTek FlowTracker Handheld ADV brochure. Retrieved on July 8, 2011 at: http://www.sontek.com/pdf/brochures/flowtracker_S01-05-0411.pdf

Taff, S.J. 1998. Managing Minnesota's Drainage System. Minnesota Agricultural Economist 1998: 1-7.

Tank, J.L., S.S. Roley, and M. Stephen. 2009. The Two-Stage Ditch and Nutrient Dynamics/The Two-Stage Ditch and Sediment Dynamics. A factsheet produced by The Nature Conservancy and The University of Notre Dame Center for Aquatic Conservation. Retrieved on July 8, 2011 at: http://www.nature.org/ourinitiatives/regions/northamerica/unitedstates/indiana/howwework/tank_lab_fact_sheet-1.pdf

The Ohio State University Extension Service. 1998. Ohioline Bulletin Number 871-98. Retrieved on July 8, 2011 at: http://ohioline.osu.edu/b871/b871_3.html

The Wetlands Initiative. 2001. Using Illinois River Floodplains for Nitrate Removal: TWI's Hennepin Levee District Example. The Wetlands Initiative. Chicago, Ill., USA.

Turner, R.E., and N.N. Rabalais. 1991. Changes in Mississippi River Water Quality This Century. BioScience 41(3): 140-147.

U.S. EPA-SAB [U.S. Environmental Protection Agency Science Advisory Board]. 2007. Hypoxia in the Northern Gulf of Mexico: An Update by the EPA Science Advisory Board. EPA-SAB-08-004. U.S. Environmental Protection Agency, Washington, D.C., USA.

UMN-RAL [University of Minnesota Research Analytical Laboratory]. 2011. Water Analysis and Methods. Retrieved July 8, 2011 at: http://ral.cfans.umn.edu/water.htm

USDA [U.S. Department of Agriculture]. 1987. Farm Drainage in the United States: History, Status, and Prospects. G.A. Pavelis, ed. Miscellaneous Publication No. 1455. Economic Research Service. Washington, D.C., USA.

USDA-NRCS [U.S. Department of Agriculture Natural Resources Conservation Service]. 2007. National Engineering Handbook. Part 654: Stream Restoration Design. U.S. Department of Agriculture. Washington, D.C., USA.

van Schilfgaarde, J. 1971. Drainage Yesterday, Today, and Tomorrow. in Proceedings of the American Society of Agricultural Engineers National Drainage Symposium. American Society of Agricultural Engineers. St. Joseph, Mich., USA.

Ward, A.D., and S.W. Trimble. 2004. Environmental Hydrology. Second Edition. Boca Raton, FL: CRC Press, 2004.

Ward, A.D., D. Mecklenburg, G.E. Powell, L.C. Brown, and A.C. Jayakaran. 2004. Designing Two-Stage Agricultural Drainage Ditches. in The Proceedings of the Eighth International Drainage Symposium. Presented at the 8th International Drainage Symposium. ASAE Publication #701P0304. American Society of Agricultural Engineers. St. Joseph, Mich., USA.

Wilson, B.N. 2001. Evaluation of Hydrologic Models Using Statistical Methods. Presented at the 2001 ASAE Annual International Meeting. ASAE Paper #01-012207. American Society of Agricultural Engineers. St. Joseph, Mich., USA.

Yang, C.T. 1971. On River Meanders. Journal of Hydrology 13: 231-253.

Zedler, J.B. 2003. Wetlands at your service: reducing impacts of agriculture at the watershed scale. Frontiers in Ecology and the Evironment 1(2): 65-72.

Appendix A Mullenbach Ditch Site Pictures

A.1 Pre-construction

Figure 37. Looking North from Mower County Road 6: A View Upstream from the Downstream End of the Project Reach. March 31st, 2009. Photo by Brad Hansen.

Figure 38. A Section of the Reach with a Continuous Sand Bed. March 31st, 2009. Photo by Brad Hansen.

Figure 39. A View of Sand Deposits and a Bench Near 52+00. Photo Taken Looking Upstream. March 31st, 2009. Photo by Brad Hansen.

Figure 40. Large Sand and Gravel Deposits on the Left Bank (Right in the Photo) at the Bend Near 50+00. Photo Taken Looking Upstream. March 31st, 2009. Photo by Brad Hansen.

Figure 41. Bank Undercutting and Overhang with a Bench Feature at the 46+54 Cross-Section. March 31st, 2009. Photo by Brad Hansen.

Figure 42. The Side Inlet on the Left Bank at 52+00. March 31st, 2009. Photo by Brad Hansen.

A.2 During Construction

Figure 43. Beginning of Excavation at the Upstream (North) End of the Ditch Reach. Two Backhoes are Excavating the Ditch Benches while the Other Machine is Leveling the Bench and Bank Slope. September 23rd, 2009.

Figure 44. A View of the Completed 2-Stage Channel Form at the Upstream (North) End of the Ditch. Photo Taken Looking Downstream from 120th Street. October 1st, 2009. Photo by Brad Hansen.

Figure 45. Excavation Continues Through the Bend Near 12+00. Photo Taken Looking Upstream. September 28th, 2009. Photo by Brad Hansen.

Figure 46. Erosion Control Mat Covering the Ditch Benches and Banks at the Upstream End. Photo Taken Looking Upstream from Approximately 10+00. September 28th, 2009. Photo by Brad Hansen.

Figure 47. Photo Showing the Side Inlets at 58+28 (Foreground) and 57+00 Following Construction. Note that This Photo was Taken Before Riprap Armoring Was Placed at the Outlets of These Structures. Photo Taken Looking Upstream. November 2nd, 2009.

Figure 48. A View of the Linear Treatment System Along the Left Bank at 28+00. The Tile Outlet is Approximately 200 ft Upstream (61 m) (in Background), while the Outlet from the Linear Treatment System to the Channel is in the Foreground. Photo Taken Looking Upstream. November 4th, 2009. Photo by Brad Hansen.

Figure 49. A Tile Outlet Discharging Onto the Ditch Bench After Construction. The Riprap Armoring Protects the Bench from the Erosive Power of the Tile's Discharge. November 4th, 2009. Photo by Brad Hansen.

Figure 50. The Rock Trench Side Inlet Intake Structure. Note the Mounded Pea Gravel, which Routes Water Downward into the Rock Trench. The Auxiliary Inlet is to the Left of the Mounded Rock, and Includes a Trash Rack to Prevent Plugging of the System. November 4th, 2009. Photo by Brad Hansen.

Figure 51. A View of the Channel Looking Downstream from the Field Road Crossing. November 4th, 2009. Photo by Brad Hansen.

Figure 52. A View of the Upstream End of the Ditch, Taken Looking Upstream from 6+00. The Vegetation Emerging Through the Erosion Control Mat Shows the Approximate Extent to which Vegetation Emerged in 2009. November 4th, 2009. Photo by Brad Hansen.

Figure 53. The Excavated Seepage Trench in the Right Bank at 22+00 Before Being Filled. November 4th, 2009. Photo by Brad Hansen.

Figure 54. The First Section of Seepage Trench Outlet Pipe (12" Plastic) Exiting the Seepage Trench Toward the Channel. November 4th, 2009. Photo by Brad Hansen.

Figure 55. The Seepage Trench Being Filled with Pea Gravel (Note the Liner to Maintain the Gravel Within the Trench). November 4th, 2009. Photo by Brad Hansen.

Figure 56. A View of the Longitudinal Drainage Pipe (8" plastic) Installed at the Bottom of the Seepage Trench to Collect Water and Carry it to the Seepage Trench Outlet Pipe. November 4th, 2009. Photo by Brad Hansen.

A.3 Post-construction

Figure 57. View of the Double-Box Culver Beneath Mower County Highway 6 (Looking Downstream). March 24th, 2010.

Figure 58. View of a Riffle Section Near Station 18+00. March 24th, 2010.

Figure 59. A View of the Culvert Beneath 120th Street (Looking Upstream). March 24th, 2010.

Figure 60. A View of the Upstream Section of Ditch (Taken from 1+00 Looking Downstream). March 24th, 2010.

Figure 61. The Tile Outlet at 5+88, with Vegetation Emerging on the Bench and Bank. May 2010.

Figure 62. A View of Emerging Vegetation in the Low-Flow Channel and on the Benches and Banks. Taken from 61+00 Looking Upstream. May 2010.

Figure 63. View of the Mullenbach Two-Stage Drainage Ditch During a High-Flow Event on September 23rd, 2010. Picture Taken Looking North from Just North of Mower County Highway 6.

Figure 64. A View of the Field Road Crossing, Looking West (Streamflow is from Right to Left). October 21st, 2010. Photo by Brad Hansen.

Figure 65. A View of the Channel Looking Downstream from 120th Street. October 21st, 2010. Photo by Brad Hansen.

Figure 66. A View of Vegetation within the Ditch While Conducting a Longitudinal Survey. October 21st, 2010. Photo by Brad Hansen.

Figure 67. A View of the Channel Looking Downstream from 26+00. The Tile Outlets at 28+00 (Left and Right Banks) Are in the Picture. The Left Bench Includes a Linear Treatment Structure. April 12th, 2011.

Figure 68. A View Looking Upstream from 24+00. April 12th, 2011.


Figure 69. A Tile Outlet and Riprap Reinforcement. April 12th, 2011.

Appendix B Ditch Seed Mixture

Table 26. Seed Mixture Applied at the Mullenbach Two-Stage Ditch Site Following Construction.

Species Common Name	Scientific Name	Seeding Rate (PLS lbs/Acre)
	Grasses	
Oats	Avena sativa	25
Fringed Brome	Bromus ciliatus	1
Canada Wild Rye	Elymus canadensis	2.5
Slender Wheat Grass	Elymus trachycaulus	2.5
Virginia Wild Rye	Elymus virginicus	2
Fowl Bluegrass	Poa Palustris	2
Switchgrass	Panicum virgatum	3
Big Bluestem	Andropogon geraidii	2
Indiangrass	Sorghastrum nutans	1
Western Wheat Grass	Elytrigia smithii	1
Little Bluestem	Schizachyrium scoparium	2
	Forbs	
Purple Prairie Clover	Dalea purpureum	0.09
Showy Tic-trefoil	Desmodium canadense	0.09
Early Sunflower	Heliopsis helianthoides	0.08
Wild Bergamot	Monarda fistulosa	0.07
Black-eyed susan	Rudbeckia hirta	0.1
Blue Vervain	Verbena hastata	0.07
Swamp Milkweed	Ascelpias incarnata	0.07
Golden Alexanders	Zizia aurea	0.15

Appendix C Surveyed Ditch Cross-Sections

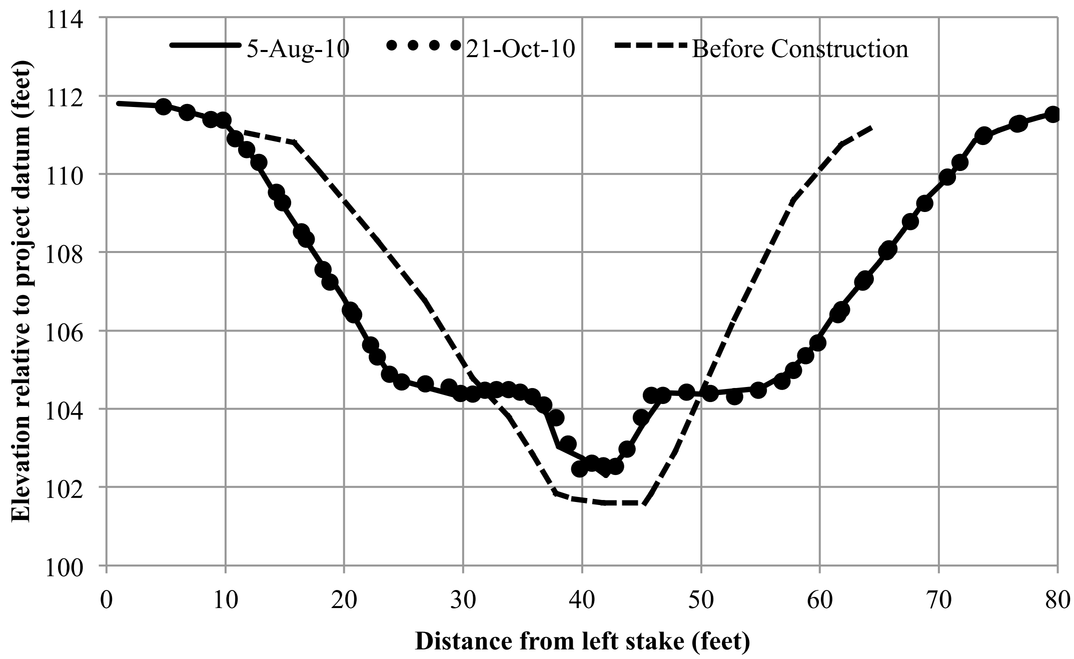

Figure 70. Channel Cross-Section Surveys Taken at the 5+16 Cross-Section.

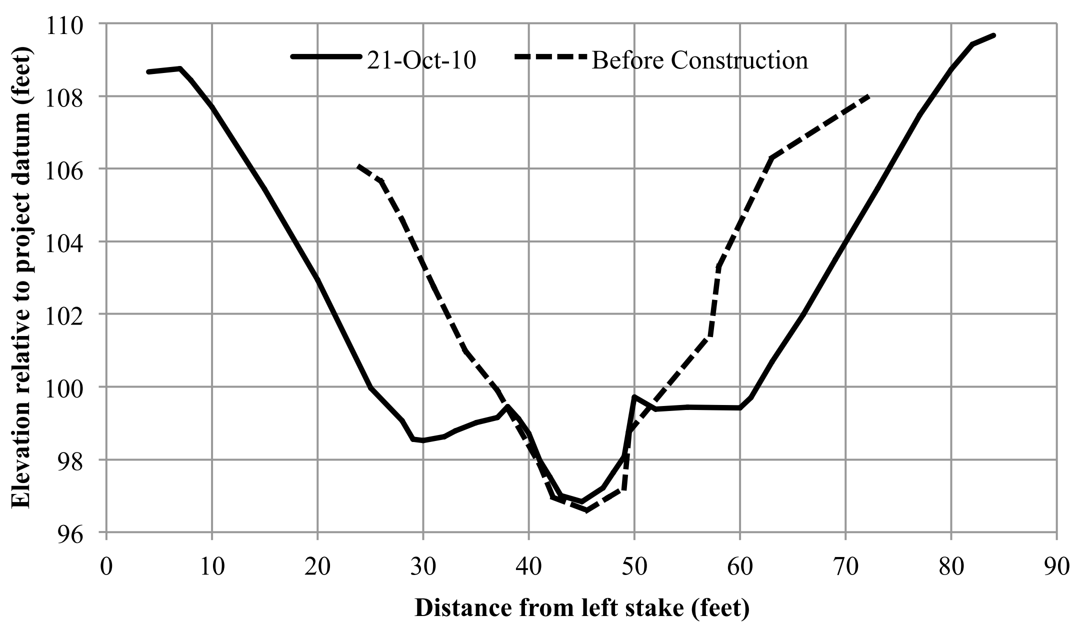

Figure 71. Channel Cross-Section Surveys Taken at the 28+25 Cross-Section.

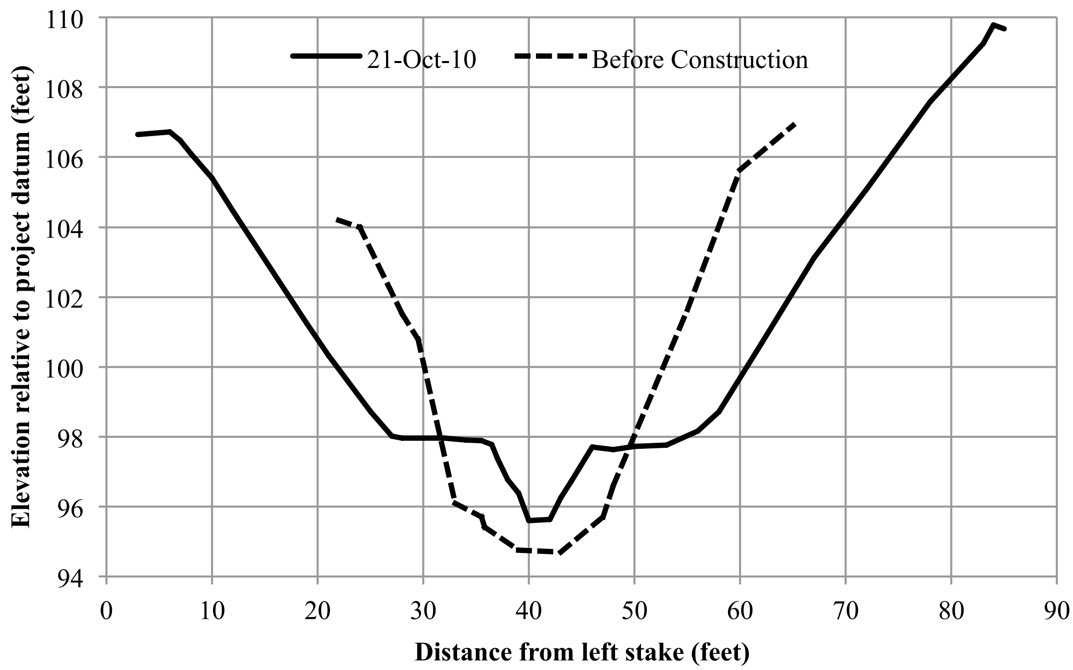

Figure 72. Channel Cross-Section Surveys Taken at the 40+04 Cross-Section.

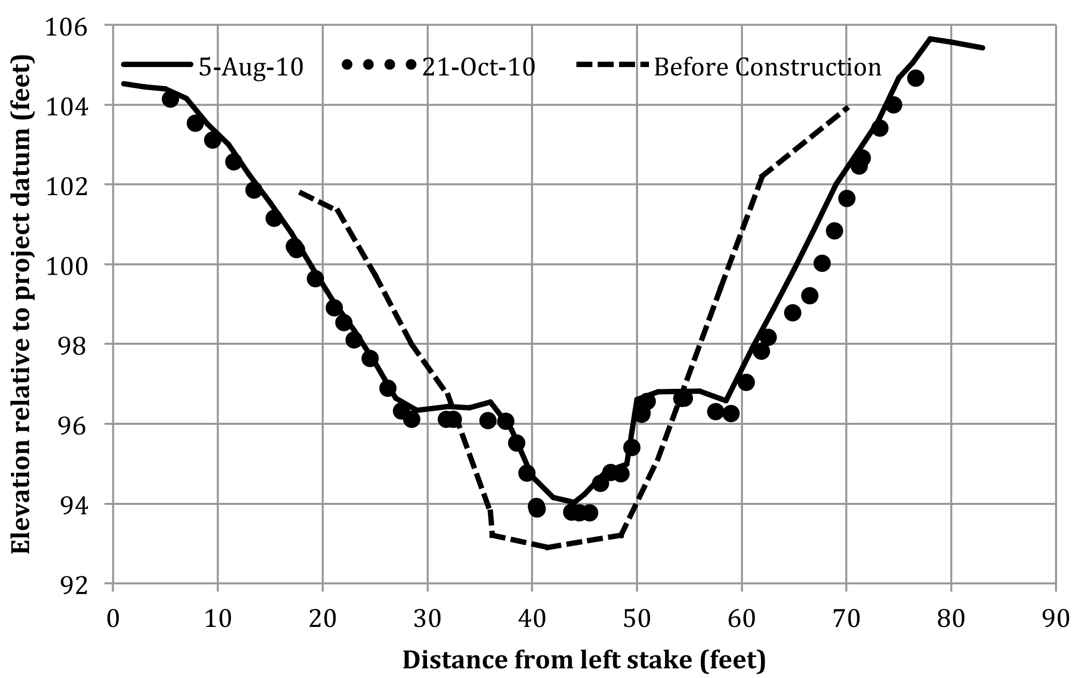

Figure 73. Channel Cross-Section Surveys Taken at the 54+83 Cross-Section.

Appendix D Stable Isotope Laboratory Methodology

The following methodology was provided by Natalie Schultz, a graduate student in the Water Isotope Laboratory in the Department of Soil, Water, and Climate at the University of Minnesota. The Water Isotope Laboratory was used for all water stable isotope analysis conducted as part of this study.

Isotope Analysis Procedure for Liquid Water Samples Using LGR Liquid Water Analyzer

revised 5/10/2011

All liquid water samples are analyzed for their isotopic composition using a laser spectroscopy system (Liquid Water Analyzer, DLT-100, Los Gatos Research, Inc.) coupled to an autosampler (HT-300A, HTA) for simultaneous measurements of D/H and $^{18}O/^{16}O$. Manufacturer specifications give a precision of ±1.0‰ for δD and ±0.25‰ for $\delta^{18}O$. The autosampler is interfaced to the analyzer and programmed using LabView 8.2 and embedded on a National Instruments cFP-2120 controller. The Liquid Water Analyzer uses off-axis integrated cavity output spectroscopy (OA-ICOS) with the laser wavelength tuned over the absorption spectra of the water species of interest. It utilizes highly reflective mirrors installed within the cavity to extend the average optical path length for light to ≈3000 meters. This extended optical path length increases the absorption, which allows the use of relatively economical diode lasers that can be operated at room temperature.

The autosampler holds a 1.0µL SGE Analytical Science microliter syringe and is programmed to inject 0.8µL syringe 6 times per sample per analyzer manufacturer recommendations. The first 2 injections of each sample are ignored due to known memory effects from the previous sample. A heated (70˚C) injection port immediately vaporizes injected water samples and transfers the vapor to the analyzer via 1/8 inch Teflon tubing. The sample is transferred into the cavity where absorption of the isotopic species is measured. Evacuation of the optical cavity occurs after each sample injection to ensure no residual water vapor remains.

The analyzer measures the concentrations of the individual isotopologues and reports them in absolute ratios. The reported absolute isotope ratios are not immediately relative to VSMOW. It is necessary to include pre-calibrated internal laboratory standards within and throughout the autoruns to calibrate the unknown standards to VSMOW. The standard operating procedure given by Los Gatos Research, Inc. is followed for autorun configuration and data processing. Using this procedure, a maximum capacity autorun of 45 total samples can hold 30 unknown samples with the rest being internal standards for unknown sample calibration (3 standards, each analyzed 5 times) Standards for each autorun are selected based on the expected isotopic composition of the unknown samples and should bracket the range of unknown samples.

This procedure begins each run with a pre-calibrated standard and then places a standard after every two unknown samples to correct for instrumental drift that can randomly occur in long autoruns. Linear calibration equations are calculated using each set of standards throughout the run and used to correct unknown samples. The standard deviation of the water standards throughout a typical autorun (45 total samples (15 standards, 30 unknowns taking approx. 18.7 hours) is better than 0.8‰ for D/H and 0.3‰ for $^{18}O/^{16}O$ and is typically around 0.40‰ for D/H and 0.15‰ for $^{18}O/^{16}O$.

The isotopic composition of water samples was shown to be very sensitive to the amount of water injected and measured within the optical cavity. Therefore, it is important to monitor the injection volume and reject samples outside an acceptable range. Samples are rejected if they vary by more than 3% per LGR protocol, although this occurrence is rare. Sample quality is ensured by maintaining the operating equipment (i.e. frequent septum changes to avoid leakage and syringe washing to reduce salt precipitate buildup) to produce consistent injection volumes. The LGR post-processing software allows for easy analysis of the accuracy of the internal standards, and thus the quality of the unknown sample data. All measured unknown water samples are calibrated to the known internal water standards and then reported in delta (δ) notation relative to VSMOW.

Appendix E Water Quality Data

As detailed in the Section 4.4.2, laboratory analysis of various parameters was performed at three separate laboratories: nutrient and other water quality analyses at the University of Minnesota Research Analytical Laboratory (UMN-RAL); nutrient and other water quality analysis at Pace Analytical (Pace); and water stable isotope analysis performed in the Department of Soil, Water, and Climate at the University of Minnesota (UMN-SWC).

E.1 October 28th, 2009

Table 27. October 28, 2009 Water Stable Isotope Analysis Results (UMN-SWC).

Location	δ^2H	δ^2H st. dev.	$\delta^{18}O$	$\delta^{18}O$ st. dev.
In channel (north end)	-51.1116	0.1189	-8.2506	0.0468
4+53 tile (left)	-51.9441	0.3105	-8.2442	0.2383
59+00 tile (left)	-52.0224	0.5147	-8.5790	0.2550
In channel (south end)	-52.0446	0.2220	-8.6803	0.0715
Left bank seepage (61+00)	-53.0491	0.1843	-8.7650	0.1052

Table 28. October 28, 2009 Water Sample Analysis (UMN-RAL).

Location	Nitrate/Nitrite-N (mg L^{-1})	Total Phosphorus (mg L^{-1})	pH	Specific Conductance (µmhos/cm)
In channel (north end)	20.6	0.12	7.9	597
In channel (north end) - duplicate	20.8	0.12	8.0	590
4+53 tile (left)	10.4	0.06	7.6	619
59+00 tile (left)	25.1	0.11	7.6	608
In channel (south end)	19.9	0.12	8.0	612
Left bank seepage (61+00)	28.3	0.09	8.2	819
Left bank seepage (61+00) - duplicate	28.5	0.11	8.2	837

E.2 November 20th, 2009

Table 29. November 20, 2009 Summary of Field Data Collected with YSI Sonde.

Location	Temperature (˚C)	Dissolved Oxygen (mg L^{-1})	pH	Specific Conductance (µS/cm)
Seepage trench	9.01	14.32	8.38	680
20+70 tile right	8.59	12.92	8.16	717
In channel (20+70)	6.08	20.37	8.10	587
18+00 tile right	8.63	14.55	8.09	854
In channel (north end)	6.64	15.67	8.02	574
4+53 tile left	8.23	14.63	8.01	577
28+00 tile left	8.49	14.55	8.07	456
28+00 tile right	7.67	14.14	7.75	686
32+25 tile left	8.03	14.31	7.85	427
34+80 tile left	8.13	14.25	7.51	599
In channel (50 feet downstream of field crossing)	7.36	14.92	7.85	578
45+45 tile left	7.92	14.77	8.10	699
47+50 (approx.) – outlet water from linear treatment structure	10.51	23.29	8.22	659
59+00 tile left	8.36	13.74	7.59	563
Seepage – right bank – south end	10.5	8	7.65	762
Seepage – left bank – south end	10.4	10	7.74	756
In channel (south end)	8.75	14.39	8.19	544
62+00 tile (left)	8.01	15.08	8.15	810
62+05 tile (right)	7.09	11.92	7.56	819

Table 30. November 20, 2009 Isotope Analysis Results (UMN-SWC).

Location	δ^2H	δ^2H st. dev.	δ^{18}O	δ^{18}O st. dev.
Seepage trench	-48.0898	0.2543	-7.2608	0.1055
20+70 tile right	-53.2616	0.5967	-7.9535	0.1839
In channel (20+70)	-53.2461	0.1286	-8.1105	0.0104
18+00 tile right	-48.8408	0.2107	-8.0295	0.0263
In channel (north end)	-51.1611	0.2235	-7.5994	0.2742
4+53 tile left	-52.2742	0.2318	-8.2276	0.2740
28+00 tile left	-50.7554	0.0500	-7.5826	0.0771
28+00 tile right	-49.6966	0.2343	-7.6071	0.1879
32+25 tile left	-50.4264	0.3732	-7.7297	0.1827
34+80 tile left	-53.7769	0.1304	-8.1663	0.2151
In channel (50 feet downstream of field crossing)	-51.7619	0.0241	-7.8075	0.1742
45+45 tile left	-48.2993	0.4107	-7.4259	0.0856
47+50 (approx.) – outlet water	-48.0912	0.3065	-7.2423	0.1879

Location	δ^2H	δ^2H st. dev.	$\delta^{18}O$	$\delta^{18}O$ st. dev.
from linear treatment structure				
59+00 tile left	-52.2602	0.1496	-7.8293	0.0956
Seepage – right bank – south end	-53.1493	0.2181	-8.0189	0.0586
Seepage – left bank – south end	-51.0214	0.1641	-8.1354	0.0799
In channel (south end)	-52.8759	0.0345	-8.0772	0.2114
62+00 tile (left)	-54.0558	0.3016	-8.1010	0.2681
62+05 tile (right)	-51.2205	0.0995	-7.7774	0.1549
MDA-SFR (MDA reference ditch)	-50.6073	0.2870	-7.6217	0.2639

Table 31. November 20, 2009 Water Sample Analysis (UMN-RAL).

Location	Nitrate/Nitrite-N (mg L^{-1})	Total Phosphorus (mg L^{-1})
Seepage trench	7.0	0.18
Seepage trench - duplicate	7.1	0.15
20+70 tile right	17.3	0.11
In channel (20+70)	17.2	0.14
18+00 tile right	26.8	0.10
In channel (north end)	21.3	0.13
4+53 tile left	10.2	0.09
28+00 tile left	12.0	0.08
28+00 tile right	28.0	0.11
32+25 tile left	9.6	0.10
34+80 tile left	11.7	0.11
In channel (50 feet downstream of field crossing)	20.3	0.15
45+45 tile left	33.4	0.09
47+50 (approx.) – outlet water from linear treatment structure	33.3	0.08
59+00 tile left	23.1	0.11
Seepage – right bank – south end	10.4	0.09
Seepage – left bank – south end	30.3	0.10
In channel (south end)	16.7	0.13
62+00 tile (left)	39.1	0.10
62+00 tile (left) - duplicate	39.3	0.11
62+05 tile (right)	22.8	0.25
MDA-SFR (MDA reference ditch)	11.6	0.10

E.3 March 24th, 2010

Table 32. March 24, 2010 Isotope Analysis Results (UMN-SWC).

Location	δ^2H	δ^2H st. dev.	$\delta^{18}O$	$\delta^{18}O$ st. dev.
In channel (north end)	-72.9383	0.4718	-10.6746	0.0655
3+96 tile (left)	-64.5331	0.5184	-9.5466	0.0515
4+53 tile (left)	-63.3150	0.6639	-9.3879	0.0660
5+88 tile (left)	-78.7140	0.1580	-11.5119	0.1126
32+25 tile (left)	-64.4382	0.4080	-9.7213	0.0623
45+45 tile (left)	-58.3009	0.7331	-9.1503	0.1191
In channel (south end)	-69.5239	0.2067	-10.1585	0.1120

Table 33. March 24, 2010 Water Sample Analysis (UMN-RAL).

Location	Nitrate/Nitrite-N (mg L^{-1})	Total Phosphorus (mg L^{-1})	pH	Specific Conductance (µmhos/cm)
In channel (north end)	16.8	0.13	7.7	453
In channel (north end) - duplicate	17.0	0.10	7.8	453
3+96 tile (left)	25.6	0.14	7.1	545
4+53 tile (left)	11.6	0.04	7.4	475
5+88 tile (left)	2.1	0.38	7.4	412
32+25 tile (left)	8.5	0.06	7.1	348
45+45 tile (left)	27.9	0.04	7.7	619
In channel (south end)	16.0	0.10	7.9	478

E.4 April 22nd, 2010

Table 34. April 22, 2010 Summary of Field Measurements of Drainage Tile Outlet Discharge and Nitrate N Concentration (HACH Nitrate N Probe).

Location	Nitrate N (mg L^{-1})	Measured Discharge (L min^{-1})
4+53 tile east	10.0	60
18+00 tile right	29.1	33
20+70 tile right	16.0	3
28+00 tile left (sample from linear wetland)	11.4(1)	0
28+00 tile right	29.1	30
32+25 tile left	16.0	1.5
32+90 tile right	N/A(2)	<<1
34+80 tile left	9.1	10
35+25 rock trench outlet	N/A(2)	<<1
45+45 tile left	31.3	18
48+20 tile right	20.8	2
59+00 tile left	22.4	45
62+05 tile right	20.5	7.5

(1) Nitrate N concentration measured in a sample taken from standing water in the linear treatment structure as there was no discharge from the linear treatment structure. (2) It was not possible to obtain a sample for nitrate N concentration measurement due to extremely low discharge (essentially a drip from the pipe end)

Table 35. April 22, 2010 Isotope Analysis Results (UMN-SWC).

Location	δ^2H	δ^2H st. dev.	$\delta^{18}O$	$\delta^{18}O$ st. dev.
In channel (north end)	-59.4774	0.4562	-8.3727	0.1598
In channel (south end)	-57.4875	0.8497	-8.2787	0.3213
Seepage near south end	-51.5095	0.8219	-7.7920	0.1574
4+53 tile (left)	-60.0222	0.9303	-9.1647	0.1451

Table 36. April 22, 2010 Water Sample Analysis (UMN-RAL).

Location	Nitrate/Nitrite-N (mg L^{-1})	Total Phosphorus (mg L^{-1})	pH	Specific Conductance (µmhos/cm)
In channel (north end)	15.4	0.06	8.2	540
In channel (north end) - duplicate	15.4	0.06	8.2	547
In channel (south end)	12.4	0.04	8.4	492
Seepage near south end	29.1	0.07	8.1	602
4+53 tile (left)	10.1	0.06	8.0	557

E.5 May 13th, 2010

Table 37. May 13, 2010 Isotope Analysis Results (UMN-SWC).

Location	δ^2H	δ^2H st. dev.	$\delta^{18}O$	$\delta^{18}O$ st. dev.
45+45 linear treatment structure (outlet from tile into linear treatment structure)	-56.8655	0.7404	-8.4703	0.3183
45+45 linear treatment structure (outlet from linear treatment structure into channel)	-53.0129	0.1868	-8.3473	0.1071
Surface runoff (south end of ditch)	-37.7103	0.6889	-6.0273	0.0351
Surface runoff (through side inlet near south end of ditch)	-53.9184	0.1754	-8.8186	0.0182

Table 38. May 13, 2010 Water Sample Analysis (UMN-RAL).

Location	Nitrate/Nitrite-N (mg L^{-1})	Total Phosphorus (mg L^{-1})	pH	Specific Conductance (µmhos/cm)
28+00 linear treatment structure (outlet from tile into linear treatment structure)	15.7 / 16.0[(1)]	0.16	7.6	716
28+00 linear treatment structure (outlet from linear treatment structure into channel)	14.6	0.09	7.9	680
45+45 linear treatment structure (outlet from tile into linear treatment structure)	29.6	0.07	8.0	608
45+45 linear treatment structure (outlet from linear treatment structure into channel)	25.1	0.10	8.0	520
Surface runoff (south end of ditch)	22.0	1.54	7.7	548
Surface runoff (through side inlet near south end of ditch)	6.9	0.23 / 0.31[(1)]	7.6 / 7.7[(1)]	378 / 377

(1) denotes duplicate results

E.6 June 14th, 2010

Table 39. June 14, 2010 Summary of Field Measurements of Drainage Tile Outlet Discharge.

Location	Measured Discharge (L min^{-1})(1)
32+25 tile left	74.9
34+80 tile left	87.0
45+45 tile left	295
48+20 tile right	278
59+00 tile left	404
62+00 tile left	21.6
62+05 tile right	24.8

(1) not all running tile lines were measured on this day

E.7 June 16th, 2010

Table 40. June 16, 2010 Summary of Field Measurements of Nitrate N Concentration (HACH Nitrate N Probe) and Drainage Tile Outlet Discharge.

Location	Nitrate N (mg L^{-1})	Measured Discharge (L min^{-1})
3+96 tile left	28.3	590
4+53 tile left	26.1	976
5+88 tile left	9.4	41.7
18+00 tile right	25.3	38.2
20+70 tile right	9	8.82
22+00 seepage trench	5.2	1
28+00 tile left (outlet from tile into linear treatment structure)	32.6	Not possible to measure
28+00 tile right	46.3	451
32+25 tile left	26.7	295
32+90 tile right	17.2	2
34+80 tile left	32.1	600
35+25 rock trench side inlet	not measured	1
45+45 linear treatment structure (outlet from tile into linear treatment structure)	33.7	750
45+45 linear treatment structure (outlet from linear treatment structure into channel)	34.0	Not possible to measure
48+20 tile right	33.3	663
51+27 tile left	29.7	36.3
59+00 tile left	38.1	1534
62+00 tile left	47.5	134
62+05 tile right	38.5	102
10+00 in channel	28.6	-
20+00 in channel	19.3	-
27+50 in channel	28.6	-
Field road crossing in channel (35+45)	29.7	-
45+00 in channel	29.3	-
52+00 in channel	29.2	-

E.8 June 24th, 2010

Table 41. June 24, 2010 Summary of Field Data Collected with YSI Sonde.

Location	Temperature (˚C)	Dissolved Oxygen (mg L^{-1})	pH	Specific Conductance (μS/cm)
In channel (south end)	18.1	9.23	7.48	499
62+05 tile right	14.8	2.82	6.97	911
62+00 tile left	15.3	7.05	7.17	902
47+50 in channel	20.2	8.97	7.45	672
45+45 tile outlet	16.7	6.48	6.95	665
30+00 in channel	20.5	9.3	7.02	732
28+00 tile left	16.7	7.58	6.93	715
5+88 tile left	17.7	6.28	6.91	522
3+96 tile left	16.7	7.13	6.91	541
In channel (north end)	18.6	8.51	7.2	475
MDA-SFR (MDA reference ditch)	19	9.21	7.62	513

Table 42. June 24, 2010 Isotope Analysis Results (UMN-SWC).

Location	δ^2H	δ^2H st. dev.	$\delta^{18}O$	$\delta^{18}O$ st. dev.
In channel (south end)	-54.2782	0.2643	-8.1912	0.1002
62+05 tile right	-58.3078	0.1404	-8.7748	0.0320
62+00 tile left	-55.4870	0.4484	-8.6114	0.0655
47+50 in channel	-58.6899	0.1392	-8.9030	0.0356
45+45 tile outlet	-57.0435	0.2907	-8.6006	0.0684
30+00 in channel	-56.4409	0.2497	-8.3786	0.0421
28+00 tile left	-56.6777	0.3973	-8.2958	0.1004
5+88 tile left	-65.0868	0.5632	-9.8578	0.0856
3+96 tile left	-56.6584	0.5139	-8.5556	0.0892
In channel (north end)	-54.3530	0.4055	-8.2402	0.0431
MDA-SFR (MDA reference ditch)	-59.1874	0.3772	-8.8143	0.0789

Table 43. June 24, 2010 Water Sample Analysis (UMN-RAL).

Location	Nitrate/ Nitrite-N (mg L^{-1})	Total Phosphorus (mg L^{-1})	Al (mg L^{-1})	B (mg L^{-1})	Ca (mg L^{-1})
In channel (south end)	24.1	0.18	0.147	<0.011	80.215
In channel (south end) - duplicate	24.3	0.18	0.159	<0.011	79.298
62+05 tile right	30.6	0.54	<0.082	<0.011	136.280
62+00 tile left	46.8	0.15	<0.082	<0.011	137.200
47+50 in channel	27.4	0.15	<0.082	<0.011	110.800

Location	Nitrate/ Nitrite-N (mg L^{-1})	Total Phosphorus (mg L^{-1})	Al (mg L^{-1})	B (mg L^{-1})	Ca (mg L^{-1})
45+45 tile outlet	30.4	0.14	0.088	<0.011	533.770
30+00 in channel	26.9	0.25	<0.082	0.014	101.810
28+00 tile left	27.5	0.35	<0.082	0.014	97.463
5+88 tile left	1.4	0.49	<0.082	<0.011	90.377
3+96 tile left	18.7	0.17	<0.082	<0.011	77.749
In channel (north end)	25.0	0.20	0.114	0.020	74.785
MDA-SFR (MDA ref. ditch)	16.8	0.14	<0.082	<0.011	100.880

Table 43, continued

Location	Cd (mg L^{-1})	Cr (mg L^{-1})	Cu (mg L^{-1})	Fe (mg L^{-1})	K (mg L^{-1})	Mg (mg L^{-1})
In channel (south end)	<0.011	<0.014	<0.017	0.096	2.423	23.185
In channel (south end) - duplicate	<0.011	<0.014	<0.017	0.124	2.436	22.879
62+05 tile right	<0.011	<0.014	<0.017	0.017	7.630	35.483
62+00 tile left	<0.011	<0.014	<0.017	<0.016	7.388	39.002
47+50 in channel	<0.011	<0.014	<0.017	<0.016	0.469	30.621
45+45 tile outlet	<0.011	0.033	<0.017	<0.016	2.224	146.980
30+00 in channel	<0.011	<0.014	<0.017	<0.016	5.045	30.856
28+00 tile left	<0.011	<0.014	<0.017	<0.016	4.925	30.007
5+88 tile left	<0.011	<0.014	<0.017	<0.016	2.690	26.560
3+96 tile left	<0.011	<0.014	<0.017	<0.016	0.776	21.416
In channel (north end)	<0.011	<0.014	<0.017	0.073	2.450	21.645
MDA-SFR (MDA reference ditch)	<0.011	<0.014	<0.017	0.040	1.018	24.921

Table 43, continued

Location	Mn (mg L^{-1})	Na (mg L^{-1})	Ni (mg L^{-1})	P (mg L^{-1})	Pb (mg L^{-1})	Zn (mg L^{-1})
In channel (south end)	0.008	10.265	<0.032	<0.370	<0.176	<0.014
In channel (south end) - duplicate	0.014	10.135	<0.032	<0.370	<0.176	<0.014
62+05 tile right	0.015	39.850	<0.032	0.444	<0.176	<0.014
62+00 tile left	<0.003	16.588	<0.032	<0.370	<0.176	<0.014
47+50 in channel	<0.003	9.864	<0.032	<0.370	<0.176	<0.014
45+45 tile outlet	<0.003	50.633	<0.032	<0.370	<0.176	<0.014
30+00 in channel	<0.003	19.513	<0.032	<0.370	<0.176	<0.014
28+00 tile left	<0.003	19.120	<0.032	<0.370	<0.176	<0.014
5+88 tile left	0.005	8.700	<0.032	0.406	<0.176	<0.014
3+96 tile left	<0.003	6.636	<0.032	<0.370	<0.176	<0.014
In channel (north end)	0.008	9.223	<0.032	<0.370	<0.176	<0.014
MDA-SFR (MDA reference ditch)	<0.003	4.837	<0.032	<0.370	<0.176	<0.014

E.9 July 14th, 2010

Table 44. July 14, 2010 Summary of In-Channel Water Temperature Measurements Performed with YSI Temperature Probe.

Location	Temperature (˚C)
0+00	22.4
2+00	22.7
4+00	23.1
6+00	22.8
8+00	23.1
10+00	23.2
12+00	23.0
14+00	23.1
16+00	23.1
17+65	22.8
18+00	23.0
20+00	23.1
22+00	23.5
24+00	23.6
26+00	23.7
28+00	23.8
30+00	23.7
32+00	24.2
34+00	24.6
35+50	25.0
36+00	25.2
38+00	25.3
40+00	25.6
42+00	25.8
44+00	25.9
46+00	25.7
48+00	25.8
50+00	24.8
51+00	24.2
52+00	24.0
53+00	23.7
54+00	22.9
55+00	22.1
56+00	22.2
57+00	22.5
58+00	22.7
59+00	23.0
60+00	23.3
61+00	23.3

E.10 August 3rd, 2010

Table 45. August 3, 2010 Summary of Field Measurements of Nitrate N Concentration (HACH Nitrate N Probe) and Drainage Tile Outlet Discharge.

Location	Nitrate N (mg L^{-1})	Measured Discharge (L min^{-1})
3+00 in channel	20.2	-
3+96 tile left	19.8	61.2
4+53 tile left	13.6	179
5+88 tile left	3.6	41.6
6+00 in channel	19.0	-
12+00 in channel	19.1	-
15+00 in channel	19.3	-
18+00 in channel	19.3	-
18+00 tile right	22.3	62.0
20+70 tile right	2.6	4.89
21+00 in channel	19.2	-
22+00 seepage trench	-	0
24+00 in channel	19.2	-
27+00 in channel	19.1	-
28+00 linear treatment structure (outlet from tile into linear treatment structure)	21.1	20
28+00 linear treatment structure (outlet from linear treatment structure into channel)	19.4	Not possible to measure
28+00 tile right	24.8	156
30+00 in channel	19.3	-
32+25 tile left	18.8	28.6
32+90 tile right	-	0
33+00 in channel	19.3	-
34+80 tile left	20.2	63.6
35+25 rock trench side inlet	-	0
35+45 field road crossing in channel	19.1	-
36+00 in channel	19.1	-
39+00 in channel	18.9	-
42+00 in channel	18.8	-
45+00 in channel	18.3	-
45+45 linear treatment structure (outlet from tile into linear treatment structure)	31.7	84.3
45+45 linear treatment structure (outlet from linear treatment structure into	31.7	Not possible to measure

Location	Nitrate N (mg L^{-1})	Measured Discharge (L min^{-1})
channel)		
48+00 in channel	19.0	-
48+20 tile right	30.1	65.6
51+00 in channel	18.4	-
51+27 tile left	-	0
54+00 in channel	17.9	-
57+00 in channel	17.6	-
59+00 tile left	25.4	189
60+00 in channel	17.5	-
62+00 tile left	42.8	18.9

Table 46. August 3, 2010 Isotope Analysis Results (UMN-SWC).

Location	δ^{2}H	δ^{2}H st. dev.	δ^{18}O	δ^{18}O st. dev.
3+00 in channel	-53.64	0.8735	-7.96	0.3627
3+96 tile left	-51.97	0.2412	-7.68	0.2998
4+53 tile left	-54.29	0.4330	-8.40	0.0690
5+88 tile left	-57.25	0.2058	-8.93	0.1406
6+00 in channel	-54.67	0.2260	-7.76	0.0690
9+00 in channel	-53.20	0.3087	-7.66	0.2263
12+00 in channel	-54.06	0.7323	-8.31	0.2429
15+00 in channel	-53.06	1.3232	-7.94	0.1065
18+00 in channel	-53.69	0.4781	-8.22	0.1636
18+00 tile right	-55.02	0.1986	-8.58	0.1092
20+70 tile right	-59.20	0.2828	-8.86	0.0894
21+00 in channel	-53.64	0.4516	-8.37	0.2532
24+00 in channel	-53.94	0.7725	-8.55	0.0864
27+00 in channel	-54.85	0.7055	-8.53	0.1270
28+00 linear treatment structure (outlet from tile into linear treatment structure)	-54.04	0.3040	-8.27	0.0812
28+00 linear treatment structure (outlet from linear treatment structure into channel)	-54.00	0.4553	-8.00	0.0563
28+00 tile right	-55.69	0.1789	-8.44	0.0716
30+00 in channel	-53.94	0.3980	-8.52	0.1107
32+25 tile left	-53.40	0.2313	-7.96	0.1097
33+00 in channel	-54.17	1.3685	-8.46	0.1767
34+80 tile left	-55.21	0.3777	-8.38	0.1677
35+45 field road crossing in channel	-54.22	0.1812	-8.56	0.0843
36+00 in channel	-54.04	0.6384	-8.52	0.0904
39+00 in channel	-53.35	0.9842	-7.89	0.1482
42+00 in channel	-53.24	0.7308	-7.69	0.1040
45+00 in channel	-53.28	1.2754	-7.85	0.1892
45+45 linear treatment structure (outlet from tile into linear	-54.88	0.1171	-8.52	0.1378

Location	δ^2H	δ^2H st. dev.	$\delta^{18}O$	$\delta^{18}O$ st. dev.
treatment structure)				
45+45 linear treatment structure (outlet from linear treatment structure into channel)	-55.32	0.1634	-8.33	0.1573
48+00 in channel	-53.05	1.3950	-7.48	0.2567
48+20 tile right	-58.55	0.6957	-8.74	0.1179
51+00 in channel	-56.07	0.6171	-8.64	0.2113
54+00 in channel	-55.85	0.4464	-8.56	0.2764
57+00 in channel	-55.76	0.1231	-8.76	0.1679
59+00 tile left	-59.00	0.4976	-8.95	0.1218
60+00 in channel	-55.08	0.8503	-8.54	0.1643
62+00 tile left	-55.38	1.1160	-8.64	0.2308
Left bank seepage (25 ft downstream of field road crossing)	-53.13	0.8464	-7.72	0.1372
29+00 right bank seepage	-43.94	0.2924	-6.30	0.0786
30+00 right bank seepage	-40.75	0.3831	-5.67	0.1159

E.11 August 4th, 2010

Table 47. August 4, 2010 Summary of Field Measurements of Nitrate N Concentration (HACH Nitrate N Probe) and Drainage Tile Outlet Discharge.

Location	Nitrate N (mg L^{-1})	Measured Discharge (L min^{-1})
3+96 tile left	20.0	76.6
4+53 tile left	13.7	154
5+88 tile left	3.7	33.8
6+00 in channel	19.0	-
12+00 in channel	19.1	-
18+00 in channel	19.3	-
18+00 tile right	22.8	56.5
20+70 tile right	2.4	3.85
22+00 seepage trench	-	0
24+00 in channel	19.2	-
28+00 linear treatment structure (outlet from tile into linear treatment structure)	20.9	20
28+00 linear treatment structure (outlet from linear treatment structure into channel)	19.0	Not possible to measure
28+00 tile right	22.7	110
30+00 in channel	19.3	-
32+25 tile left	18.8	23.2
32+90 tile right	-	0
34+80 tile left	19.2	50.1
35+25 rock trench side inlet	-	0
35+45 field road crossing in channel	19.1	-
42+00 in channel	18.8	-
45+45 linear treatment structure (outlet from tile into linear treatment structure)	31.2	72.3
45+45 linear treatment structure (outlet from linear treatment structure into channel)	31.3	Not possible to measure
48+00 in channel	19.0	-
48+20 tile right	30.1	34.8
51+27 tile left	-	0
54+00 in channel	17.9	-
59+00 tile left	25.8	83.7
62+00 tile left	39.8	9.70

Table 48. August 4, 2010 Summary of In-Channel Water Temperature Measurements Performed with YSI Temperature Probe.

Location	Temperature (˚C)
0+00	19.1
1+00	19
2+00	19
4+00	19
6+00	18.8
8+00	18.8
10+00	18.8
12+00	18.6
14+00	18.6
16+00	18.6
18+00	18.5
20+00	18.6
22+00	18.7
24+00	18.7
26+00	18.7
28+00	18.7
30+00	18.7
32+00	18.8
34+00	19.2
35+50	19.1
36+00	19.2
38+00	19.2
40+00	19.2
42+00	19.3
44+00	19.4
46+00	19.1
48+00	19.5
50+00	19.3
52+00	19
54+00	18.6
56+00	18.2
58+00	18.3
60+00	18.4
61+00	18.4

E.12 August 8th, 2010

Table 49. August 8, 2010 Summary of Field Measurements of Nitrate N Concentration (HACH Nitrate N Probe) and Drainage Tile Outlet Discharge.

Location	Nitrate N (mg L^{-1})	Measured Discharge (L min^{-1})
3+96 tile left	15.8	13.9
4+53 tile left	12.7	60.5
5+88 tile left	3.5	13.6
6+00 in channel	16.5	-
12+00 in channel	16.6	-
18+00 in channel	16.7	-
18+00 tile right	23.4	44.5
20+70 tile right	1.6	2.23
22+00 seepage trench	-	0
24+00 in channel	16.4	-
28+00 linear treatment structure (outlet from tile into linear treatment structure)	-	0
28+00 linear treatment structure (outlet from linear treatment structure into channel)	-	0
28+00 linear treatment structure (standing water sampled at approximately 28+50)	6.8	-
28+00 tile right	11.5	35.7
30+00 in channel	16.1	-
32+25 tile left	13.7	1.38
32+90 tile right	-	0
34+80 tile left	16.9	10.0
35+25 rock trench side inlet	-	0
35+45 field road crossing in channel	15.8	-
42+00 in channel	15.3	-
45+45 linear treatment structure (outlet from tile into linear treatment structure)	31.3	25.0
45+45 linear treatment structure (outlet from linear treatment structure into channel)	31.2	Not possible to measure
48+00 in channel	15.7	-
48+20 tile right	29.7	10
51+27 tile left	-	0
54+00 in channel	13.6	-
59+00 tile left	19.9	18.2
62+00 tile left	-	0

E.13 October 21st, 2010

Table 50. October 21, 2010 Water Sample Analysis (Pace).

Location	Nitrate/Nitrite-N (mg L^{-1})	Total Phosphorus (mg L^{-1})	Chloride (mg L^{-1})
In channel (south flume)	1.8	0.38	22.1
48+20 tile right	3.0	< 0.050	29.9
45+45 linear treatment structure (outlet from tile into linear treatment structure)	1.9	< 0.050	20.2
In channel (field road crossing)	2.3	< 0.050	28.7
28+00 linear treatment structure (outlet from tile into linear treatment structure)	0.84	< 0.050	23.3
18+00 tile right	1.3	< 0.050	43.8
4+53 tile left	< 0.10	< 0.050	34.2
In channel (north flume)	2.8	< 0.050	28.4