South Dakota Water Research Institute
Annual Technical Report
FY 2016

Introduction

South Dakota Water Resources Institute’s (SDWRI) programs are administered through the College of Agricultural and Biological Sciences at South Dakota State University (SDSU). Dr. Van Kelley has served as the Director for the Institute since August 1, 2000. Dr. Kelley is also the head of the Agricultural and Biological Systems Engineering Department. In addition to the Director, the Institute’s programs are administered and executed by a staff consisting of an Assistant Professor and an Environmental Research Coordinator. During FY2016, the SDWRI financially supported, through its base funding or through externally funded projects, four MS students and two undergraduate research assistants.

The annual base grant from the United States Geological Survey (USGS) and a South Dakota legislative appropriation form the core of the SDWRI budget. The core budget is supplemented by research grants from state and federal agencies as well as private organizations and industry interested in specific water-related issues.

The mission of the South Dakota Water Resources Institute is to address the current and future water resource needs of the people, industry, and the environment, through research, education, and service. To accomplish this mission, SDWRI provides leadership by coordinating research and training at South Dakota State University and other public educational institutions and agencies across the state in the broad area of water resources. Graduate research training, technology transfer, and information transfer are services which are provided through the Institute.

This report is a summary of the activities conducted by the SDWRI during the period March 1, 2016 through February 28, 2017.

Research Program Introduction

Water is one of the most important resources in South Dakota. Together with the state's largest industry, agriculture, it will play an important role in the economic future of the state.

During FY 2016, the South Dakota Water Resources Institute (SDWRI) used its 104B Grant Program funds to conduct research of local, state, regional, and national importance addressing a variety of water problems in the state and the upper Midwest region.

The WRI 104B External Review Panel reviewed 12 grant applications, and 4 projects were funded that addressed research priorities that had a good chance of success, and would increase our scientific knowledge. The projects were titled: - Evaluating the Potential of Duckweed for Nutrient Capture and Use in Midwest Livestock Production. PI’s: E. Cortus, L. Wei, D. Casper, J. Walker, South Dakota State University. Controlling Harmful Algal Blooms in Eutrophic Lakes by Combined Phosphorus Precipitation and Sediment Capping (year 2). PI’s: K. Min, G. Hua, South Dakota State University. Evaluating Nutrient Best Management Practices to Conserve Water Quality (Year 2). PI’s: L. Ahiablame, S. Kumar, J. Kjaersgaard, South Dakota State University. Evaluating E. coli particle attachment and the impact on transport during high flows. PI’s: R. McDaniel, B. Blakely, South Dakota State University.

Furthermore, the project listed below was funded through a USGS 104G grant: - Hydrologic Life Cycle Impact of Mountain Pine Bark Beetle Infestations. PI: J. Stone. South Dakota School of Mines and Technology.

Progress and completion reports for these projects are enclosed on the following pages.

Hydrologic Life Cycle Impact of Mountain Pine Bark Beetle Infestations

Basic Information

Title:	Hydrologic Life Cycle Impact of Mountain Pine Bark Beetle Infestations
Project Number:	2015SD248G
USGS Grant Number:	2015SD248G
Start Date:	9/1/2015
End Date:	8/30/2018
Funding Source:	104G
Congressional District:	1
Research Category:	Water Quality
Focus Category:	Water Quality, Water Use, Surface Water
Descriptors:	None
Principal Investigators:	James Stone, Scott J Kenner, Heidi Leah Sieverding

Publications

There are no publications.

Introduction

This project is assessing the dissolved organic carbon (DOC) runoff from mountain pine beetle (MPB) impacted catchments within the ponderosa pine forest of the Black Hills of South Dakota. This project primarily involves field work measuring runoff water quality and soil changes due to MPB as well as hydrologic modeling.

Research Program

Problem

Across the Western US, both large and small population centers are situated in the foothills and mountains at the heart of the MPB epidemic. These cities are also heavily dependent on storage of surface water for drinking water resources; smaller urban areas lack the leverage, capital and resources that larger municipalities have. DOC exports from MPB-impacted forest ecosystems contain precursor compounds that can react during drinking water purification (in treatment facilities) with disinfectants such as chlorine to form highly toxic and regulated disinfection by-products (DBPs).

Research Objectives

The project objectives include the following:

- Better elucidate how and why the timing of organic matter and carbon loading occurs during various stages of MPB mortality stages in ponderosa pine forests;
- Determine whether the expected increase in watershed runoff efficiencies during MPB stages may results in increased metal, carbon, and nutrient loading may occur;
- Determine changes in the 'embodied energy' of drinking water supplies using life cycle assessment (LCA) modeling due to MPB water resources impairment; and
- Provide a watershed assessment that integrates changes in organic and carbon loading, drinking water environmental footprints (embodied energy), and ponderosa pine forestry MPB management options that addresses 'triple bottom line' alternatives for forestry managers and municipalities.

Methodology

Five hydrologic sub-basins based on hydrologic unit code (HUC) cataloging unit level (12-digit) representing each of the four phases of MPB infestation (green, red, gray, toppled) with similar geology have been identified within the upper Rapid Creek watershed. Due to the pervasiveness of the infestation, it was not possible to identify un-impacted areas in the all of the general watershed zones outlined in the proposal. After assessing the sub-basins during sample site selection, it was also discovered that the sub-basins with significant active acid mine drainage (iron bogs) and karst impacts created unique, basin-level situations that could not be replicated in different zones or sub-basins and did not represent the entire watershed. So, in order to create a realistic representation of overall watershed interactions, these unique areas were avoided for intensive sub-basin sampling and dynamics modeling. These unique areas were characterized by a water pH of 5 or lower and lack of or intermittent surface flow (sink areas). Watersheds with similar degrees of impact were selected through aerial photographic analysis, screening with USGS-EROS' new land cover mapping tool LCMap, GIS analysis (soil, geology, tree stand density and age distribution, harvest and MPB infection history), and physical site visits. The sub-basins which meet these constraints generally do not have year-round access.

As part of the site selection process, LCMap change detection was contrasted with USFS MPB infestation records. LCMap analyzes the reflectance value of all images in the Landsat archive on a per pixel basis. This tool then detects the annual reflectance pattern for each of the bands and when there is a statistically significant change in the reflectance. Based on the comparison with USFS MPB annually mapped impact areas, this tool can effectively determine the month of red phase onset and subsequent forest response.

Baseflow water quality samples prior to annual high spring flow events have been collected and are being analyzed. These baseline samples were analyzed by USFS at the Rocky Mountain Research Station (Fort Collins CO) and analyzed at SDSM&T using total organic carbon (TOC) standard operating procedures (SOP) developed under the USGS 104B grant by recent MS graduate, Erik Vik.

A sampling protocol defining the number and volume of samples collected, analyses to be conducted on each volume and detailing labeling system, storage and disposal has been developed. New MS student funding on the project, Jesse Punsal, has been trained on SDSM&T's TOC analyzer. Jesse will be trained on SDSM&T's AquaLog to conduct organic carbon characterization analyses this summer. As part of his training, he will be developing a SOP for the instrument.

Significance

Development of SOPs, sampling protocols, and the study site selection process has provided valuable learning experiences for students. Active participation of the students in this process ensures that sample collection will be properly collected and processed.

Data is currently being collected and being used to determine the timing of DOC loading due to MPB and associated runoff efficiencies. Once data is processed and incorporated into models, watershed assessments and impacts to water resources and associated LCA impact and energy consumption will be evaluated.

Principal Findings

Preliminary assessments conducted through preceding USGS 104B grant detected a pattern of DOC release roughly coinciding with MPB stages. A statistically significant correlation between runoff and DOC migration for three and five to six years after the red phase, presumed to correlate with the decay of needles and wood respectively, has been made for most of the upper Rapid Creek basin. A peer review manuscript has been written and submitted to peer-reviewed publication summarizing this finding (Timing of Organic Carbon Release from Mountain Pine Beetle Impacted Ponderosa Pine Forests: Erik S. Vik, Heidi L. Sieverding, Jesse J. Punsal, Scott J. Kenner, Lisa A. Kunza, and James J. Stone to *Water Environment Research*). Due to the discovery of this correlation and the current, widespread nature of the infection - sub-basin hydrologic monitoring plan has been slightly altered to further investigate if this correlation is present consistently in the watershed.

It has been determined that the new USGS-EROS LCmap tool can be used detect the current and historical (back to 1984) spatial and temporal spread of MPB mortality and document recovery.

Information Transfer Program

The project is in its first year. As part of the information transfer to the public, four related presentations and a paper was submitted to peer-review (listed under prior projects) on preliminary results.

Students rehearsed their presentations at the SDSM&T Student Research Symposium (http://www.sdsmt.edu/Research/Student-Research-Symposium/) with a smaller audiences prior to the Hydrology Conference.

Western South Dakota Hydrology Conference (http://sd.water.usgs.gov/WSDconf/) had approximately 300 attendees from the region. Presentations included:

- Oral Presentation: Sensitivity of Black Hills Hydrology to Land-use Change Using WRF-Hydro. Lucas Barrett and William Capehart
- Oral Presentation: Simulation of the effects of deforestation on headwater streams in the Black Hills, western South Dakota. Brian Freed, Galen Hoogestraat, and Scott Kenner
- Oral Presentation: Geochemical impacts of mountain pine beetles on Rapid Creek, SD. Jesse Punsal, Erik Vik, Heidi Sieverding, Scott Kenner, Lisa Kunza, and James Stone
- Poster Presentation: Modeling the hydrological impact with land cover change over time. Patrick Shaw and Scott Kenner

Student Support

Jesse Punsal and Patrick Shaw, M.S. graduate student in Civil Engineering started graduate research assistanceships on the project during January 2016. Patrick is also a volunteer at the USGS South Dakota Water Science Research Center. During the first six months they have been working on the project, they have made significant strides and have learned several new instruments and analysis tools. Jesse traveled to Fort Collins, CO and received instrument and sample collection training from Chuck Rhoades with the USFS in March 2016. In April 2016, Patrick Shaw and Heidi Sieverding traveled to Garretson, SD and received training on the new USGS-EROS' Landsat-based land cover change detection tool.

Preliminary work on the project was conducted with the support of USGS CESU-funded students, Brian Freed and Lucas Barrett as well as USGS 104B-funded student, Erik Vik. Their work is being continued and expanded by current students.

Notable Awards and Achievements

N/A

Publications from Prior Projects

Timing of Organic Carbon Release from Mountain Pine Beetle Impacted Ponderosa Pine Forests: Erik S. Vik, Heidi L. Sieverding, Jesse J. Punsal, Scott J. Kenner, Lisa A. Kunza, and James J. Stone submitted to *Water Environment Research.*

SDSMT M.S. Thesis: Erik Vik - Potential organic carbon exports within the upper Rapid Creek watershed due to mountain pine beetle infestation

SDSMT M.S. Thesis: Brian Freed - Hydrologic Impacts of the Mountain Pine Beetle in Headwater Streams in the Black Hills of Western South Dakota.

Evaluating the Potential of Duckweed for Nutrient Capture and Use in Midwest Livestock Production

Basic Information

Title:	Evaluating the Potential of Duckweed for Nutrient Capture and Use in Midwest Livestock Production
Project Number:	2016SD258B
Start Date:	3/1/2016
End Date:	2/28/2017
Funding Source:	104B
Congressional District:	SD-001
Research Category:	Water Quality
Focus Category:	Treatment, Water Quality, Nutrients
Descriptors:	None
Principal Investigators:	Erin Cortus, Lin Wei, David Casper, Julie Walker

Publications

There are no publications.

Annual Report

Evaluating the Potential of Duckweed for Nutrient Capture and Use in Midwest Livestock Production

Prepared for:
South Dakota Water Resource Institute
South Dakota State University

Prepared by:
Erin Cortus
Associate Professor and Environmental Quality Engineer
Agricultural and Biosystems Engineering, South Dakota State University

Project Partners:

Lin Wei, Assistant Professor
Mustafa Alsowij, Graduate Research Assistant
Tom Dubois-Moureau, Visiting Apprentice
Joseph Darrington, Assistant Professor

Agricultural and Biosystems Engineering, South Dakota State University

Julie Walker, Associate Professor

Animal Science, South Dakota State University

David Casper, Assistant Professor

Dairy Science, South Dakota State University *(formerly)*

Roger Foote, Project Coordinator

Upper Big Sioux Watershed Project

April 27, 2017

Executive Summary

There is a continuous need to identify, evaluate and optimize cost-effective means for cleaning impaired water. By producing a valuable by-product in the process of cleaning water, there are opportunities to alter the cost-benefit ratio of new technologies and enhance adoption of practices. Duckweed is recognized as a small, floating aquatic plant with a propensity to grow under a relatively wide range of physical and chemical conditions, while removing nutrients and metals from the supporting water. Given the potential environmental and economic benefits, this project was designed to promote further understanding of the opportunities and challenges for duckweed growth systems in South Dakota and the Midwest region.

This project took advantage of a multi-disciplinary team, centered on lab-scale experiments and team-level design meetings. The project scope was limited to primarily bench and lab-scale work to address three objectives. The objectives were: (1) identify the range of suitable conditions for duckweed growth, and limitations imposed by Northern Great Plains climate; (2) evaluate the digestibility of duckweed inclusion in silage; and (3) develop a baseline model for processing duckweed into useable formats for livestock feed.

Tasks performed include a literature review (Objective 1), mini-silo tests to evaluate silage conditions and opportunities (Objective 2); and a group-based brainstorming activity and model development (Objective 3).

Experimental and modeling work was conducted by graduate research assistants, with the supervision of an advisory team including animal scientists, engineers and an industry partner from the Upper Big Sioux Watershed Project.

The outcomes are:

(1) Ideal conditions for duckweed growth are possible in both natural and constructed water bodies, but the ideal temperature conditions are limited by changing seasons in South Dakota. An existing model provides a reasonable starting point for predicting duckweed growth in this region.
(2) Fermentation and digestibility analyses of ensiled duckweed allow for future consideration in feed rations for ruminants.
(3) Process flows and realistic end uses of duckweed enable future experimentation and optimization experimentation.

Background

Non-point source pollution, including excess nutrients like nitrogen and phosphorus, is a common problem for water bodies across the state of South Dakota and region. Implementation of best management practices for reducing pollution load from upland sources is the ideal means for pollution prevention, but relies on adoption by multiple landowners and managers. Once nutrients enter a water body, there are few methods for cleaning the water outside of natural processes. Therefore, new methods for cleaning water either at the upland source/discharge point, or in the cumulative sink, are needed.

Duckweed is recognized as a small, floating aquatic plant with a propensity to grow under a relatively wide range of physical and chemical conditions, while removing nutrients and metals from the supporting water. Literature demonstrates duckweed's potential as a feedstuff for swine (Rojas et al., 2014), fish and shrimp (Landesman et al., 2002), broilers (Olorunfemi, 2006; Mwale and Gwaze, 2013) and other domestic animals, poultry and fish (Leng et al., 1995). Duckweed properties can be further influenced through treatments like drying or ensiling. Other potential value-added products from duckweed are paper and oil (Bell, 2011, Biotech Waste Management).

Literature has also documented the survivability of duckweed (Leng et al., 1995), the use of duckweed for removing nutrients from water (Mohedano et al., 2012), and as a water quality indicator (Chaudhary and Sharma, 2014). Mohedano et al. (2012) reported a very high nitrogen removal rate from piggery effluent.

The overall hypothesis is that the growth, collection and utilization of duckweed results in a self-sustaining, economically viable, nutrient cycle that can be used in tandem with manure management on livestock production systems, or as an add-on system to nutrient-rich water sinks in the Midwest Region (Figure 1).

The hypothesized system is described in four basic steps. First, the Growth System provides the seedstock, growth media and fertilizer (including the nutrient-rich water) in optimized environmental conditions. The second step is the Harvest System, wherein equipment and labor requirements need to be matched to duckweed production and utilization rates. After harvest, the Processing System takes the raw, high water content duckweed material through the required unit operation processes (i.e. drying, ensiling, grinding, etc.) to produce the desired product. The final component is called Utilization, and this proposal will focus on the use of duckweed as livestock feedstuff. Feed trials and the design of a harvest system are not proposed in this proposal.

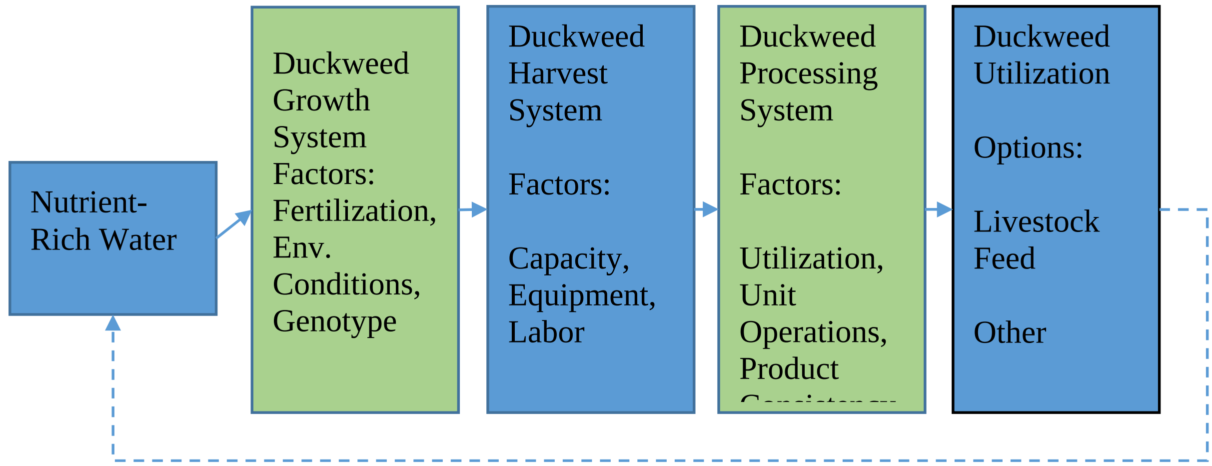


Figure 1. Flow of nutrients (eg. Nitrogen and Phosphorus) through a duckweed production and utilization system. Required linkages are shown with solid lines and potential linkages are shown with dashed lines. System components addressed through project objectives are shown in green.

We limited the project scope to bench and lab-scale work to address the two components shown in green in Figure 1, with the concept of generating baseline data to support larger proposals for feed trials, implementation and demonstration sites, optimization studies, etc. The specific objectives were: (1) identify the range of suitable conditions for duckweed growth, and limitations imposed by Northern Great Plains climate; (2) evaluate the digestibility of duckweed inclusion in silage; and (3) develop a baseline model for processing duckweed into useable formats for livestock feed.

Results and Discussion

Objective 1: Identify the range of suitable conditions for duckweed growth, and limitations imposed by Northern Great Plains climate

The reviewed literature highlighted relationships in duckweed growth with nitrogen content of the water source (Landesman et al., 2005), air temperature and solar radiation (Leng et al., 1995), water flow rates and ammonia concentration (Leng et al., 1995), and plant density (or harvest frequency) (Frederica et al., 2006). Nitrogen content of the water source also affects the protein content of duckweed (Landesman et al., 2005). Landesman et al. (2005) suggest an optimal nitrogen content of 10 mg/L for biomass and protein, but concentrations up to 35mg/L should be acceptable.

A literature review revealed a growth prediction equation based on Lemna obscura growth in a greenhouse in Texas (Landesman et al., 2005). The prediction model relates duckweed growth to air temperature, solar radiation, and nitrogen concentration (Eq 1).

$$Y = \left((1 - e^{-AX})(B - CX) + D\right)\left(1 - \left|1 - \frac{T}{T_o}\right|\right)\left(1 - \left|1 - \frac{R}{R_o}\right|\right) \quad \text{(Eq. 1)}$$

Where:
Y = Wet mass growth rate (g/day)
X = Total nitrogen concentration (mg/L)

A, *B*, *C*, *D* = Coefficients of the model (A = 0.308, B = 7.18, C = 0.201 and D = 7.01)
T = Observed temperature (°C)
T_o = Optimal temperature for duckweed growth (26°C)
R = Observed solar radiation (W/m2)
R_o = Optimal solar radiation for duckweed growth (138 W/m2)

Landesman et al. (2005) validated the coefficients in Eq. 1 for greenhouse growth conditions in Texas.

We applied the growth model to growth conditions inside a local facility. The concept for this project arose from the Phosphorus Removal Facility (PRF) on Lake Kampeska in Watertown, SD. The PRF was originally designed for algae growth, and includes a bioreactor whose surface is 271 m^2 and illuminated by 3500 LED to remove phosphorus from lake water cycled through the facility. The bioreactor has additional nitrogen added (1.6 mg/L) to support biomass growth. In the past year, there was periodic uninhibited growth of duckweed in the algae growth chamber. Collaborator and PRF Manager Roger Foote monitored and recorded the environmental conditions and rate of duckweed growth (mass removed) at the PRF from September 2015 to June 2016. Data included input water stream physical and chemical composition, PRF growth chamber physical conditions, output water stream composition, and duckweed harvest volumes. Table 1 demonstrates variable duckweed production rates. The moisture content and this wet weight of duckweed can also be highly variable based on if and how duckweed is removed from the water.

Table 1. Wet weight of duckweed harvested from the PRF in 2015.

Collection Period	**Wet Weight Harvested (lbs)**	**Daily Production Rate (lb/d)***
25Aug to 29Sept	100	2.86
29Sept to 13Oct	100	7.14
13Oct to 17Nov	100	2.86

* Calculated

Figure 2 compares the growth estimates by Eq. 1 to the measured duckweed production (Table 1).

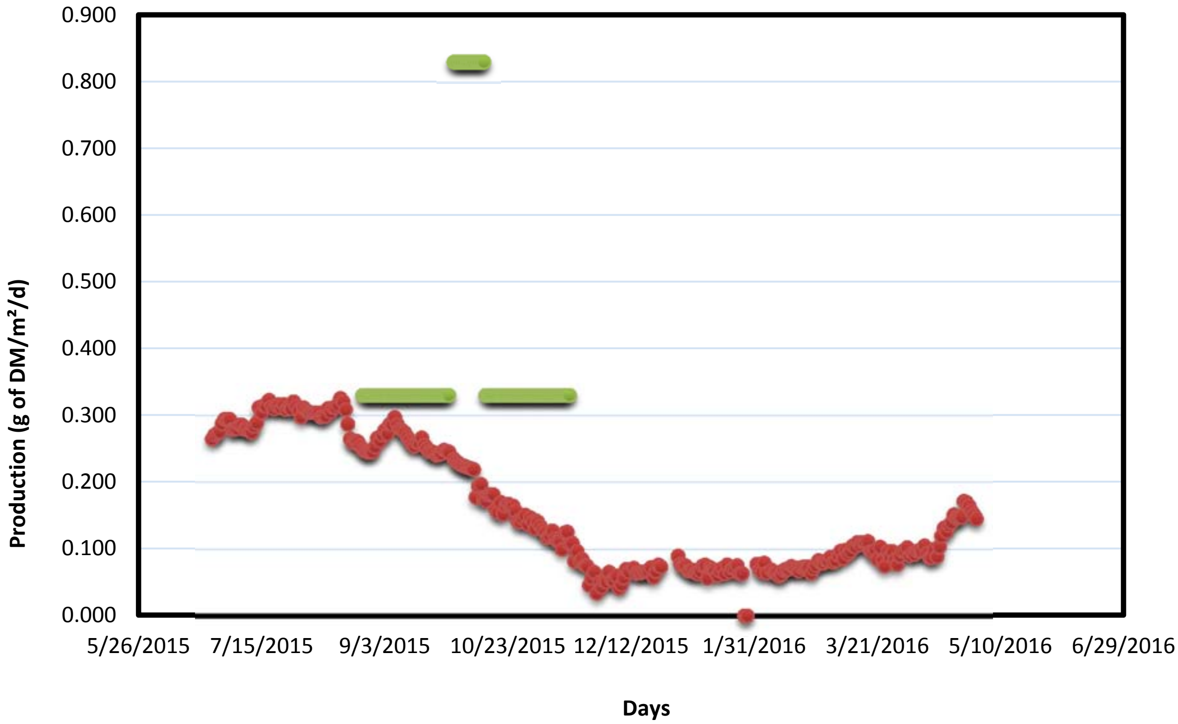


Figure 2. Estimated (red) duckweed production using the Landesman et al. (2005) model, and observed (green) duckweed production at the Phosphorus Removal Facility from June 2015 to June 2016.

During warm weather conditions (early September 2015), the model and observed production rates were in a similar range. Duckweed production remained high longer than estimated by the Landesman et al. (2015) model, which may be related to the air and water temperature influence on growth. Equation 1 is a reasonable starting point to model duckweed production in this region, with future research to fine-tune coefficients and optimal conditions.

Interestingly, during the summer of 2016, duckweed growth was prevalent in natural water systems in South Dakota (Figure 3).

Figure 3. Duckweed growth on the Big Sioux River in August 2016.

Outcomes: Per existing literature, duckweed growth is optimized when nitrogen content is 10 mg/L, water temperature is 26°C and average solar radiation levels are 138 W/m^2. These conditions are possible in both natural and constructed water bodies, but the ideal temperature conditions are limited by changing seasons in South Dakota. An existing model provides a reasonable starting point for predicting duckweed growth in this region.

Objective 2: Evaluate the digestibility of duckweed inclusion in silage

Duckweed production within the bioreactor did not produce sufficient duckweed material in the project time period, so we collected duckweed from the surface of the Big Sioux River in August and September of 2016 and dried the material. The duckweed was rewetted to 35% moisture content at the time of ensiling. Mini-silos were vacuum-sealed polyethylene bags with 200 g of wet weight silage material (Cherney et al. 2004). At the end of the silage period, we transferred the samples to sealed bags and placed the samples in a freezer prior to analysis. Dairyland Laboratories (Arcadia, WI) performed fermentation quality (volatile fatty acid profile), digestibility (neutral detergent fiber, NDF) and fat (ether extract) analyses of the samples.

The experimental design was a randomized block design. The treatment design was a 2 x 2 factorial design considering silage time (60 days, 90 days) and inoculant use (inoculant, no inoculant) as fixed factors. Duckweed was collected in two batches, and batch was considered a random block effect. There were two replicates of each treatment combination in Batch 1, and four to five replicates of each treatment combination in Batch 2.

The statistical analysis approach was a mixed model of the fixed (silage time, inoculant use) and random factors, and interaction of the fixed factors. Results are presented in Table 2 and 3. A 90-day silage time significantly ($p<0.001$) decreased crude protein content and increased ammonia content compared to 60-day silage time. There were no significant effects of silage time, inoculant use or interaction of these treatments on the other digestibility characteristics of the ensiled duckweed. A 90-day silage time significantly increased butyric and iso-butyric compared to 60-day silage time, whereas inoculant use significantly increased lactic acid and ethanol content of the ensiled duckweed. Succonic acid was interesting occurrence, and typically not detected in silage analyses (Personal communication with N. Wininger, Dairyland Laboratories, January 24, 2017).

Table 2. Digestibility and fat content of ensiled duckweed based on silage time and inoculant use.

Factors	Level or p value	Dry Matter (DM), %	aNDF, %DM	aNDFom, %DM	Fat (EE), %DM	Crude Protein (CP), %DM	Ammonia as CP, %DM
Silage Time (days)	60	32.59± 1.61	34.14 ± 1.42	30.78 ± 1.60	2.85 ± 0.32	**16.86 ± 2.25**	**8.28 ± 3.72**
	90	32.95± 2.87	34.69 ± 2.16	31.17 ± 2.23	2.79 ± 0.38	**15.95 ± 2.29**	**8.54 ± 4.18**
	p	0.71	0.27	0.55	0.77	**<0.001**	**<0.001**
Inoculant Use	I	32.66 ± 1.75	34.41 ± 1.78	30.90 ± 1.70	2.85 ± 0.30	16.44 ± 2.22	8.39 ± 3.94
	NI	32.86 ± 2.71	34.38 ± 1.86	31.02 ± 2.13	2.79 ± 0.39	16.44 ±2.40	8.41 ± 3.94
	p	0.65	0.92	0.14	0.56	0.964	0.873
Interaction	60 x I	32.68 ± 1.69	34.00 ± 1.19	30.56 ± 1.19	2.87 ± 0.24	16.78 ± 2.31	8.28 ± 3.83
	60 x NI	32.50 ± 1.66	34.29 ± 1.71	30.99 ± 2.01	2.82 ± 0.40	16.94 ± 2.36	8.28 ± 3.91
	90 x I	32.63 ± 1.98	34.89 ± 2.31	31.31 ± 2.21	2.82± 0.38	16.03 ± 2.24	8.51 ± 4.42
	90 x NI	33.28 ± 3.73	34.49 ± 2.19	31.04 ± 2.45	2.77 ± 0.42	15.87 ±2.54	8.57 ± 4.35
	p	0.36	0.34	1.33	0.96	0.314	0.862

Table 3. Fermentation quality of ensiled duckweed based on silage time and inoculant use, expressed as percent of dry matter.

Factors	Level or p value	Lactic Acid	Acetic Acid	Propionic Acid	Butyric Acid	Iso-Butyric Acid	Ethanol	Succinic acid
Silage Time (days)	60	0.17 ± 0.17	4.57 ± 1.58	0.67 ± 0.36	**1.41 ± 0.84**	**0.30 ± 0.15**	0.32 ± 0.07	2.70 ± 1.13
	90	0.18 ± 0.24	4.31 ± 1.5	0.75 ± 0.48	**1.70 ± 1.06**	**0.36 ± 0.19**	0.28 ± 0.07	2.38 ±1.09
	p	0.898	0.433	0.077	**<0.001**	**0.017**	0.171	0.155
Inoculant Use	I	**0.22 ± 0.26**	4.46 ± 1.61	0.72 ± 0.43	1.55 ± 0.95	0.33 ± 0.16	**0.33 ± 0.06**	2.65± 1.15
	NI	**0.12 ± 0.10**	4.44 ± 1.50	0.69 ± 0.42	1.54 ± 0.96	0.32 ± 0.18	**0.27 ± 0.07**	2.45 ± 1.08
	p	**0.037**	0.903	0.716	0.974	0.778	**0.012**	0.168
Interaction	60 x I	0.21 ± 0.22	4.63 ± 1.72	0.69 ± 0.41	1.43 ± 0.86	0.32 ± 0.16	0.35 ± 0.07	2.72 ± 1.23
	60 x NI	0.12 ± 0.10	4.51 ± 1.57	0.64 ± 0.34	1.40 ± 0.89	0.28 ± 0.15	0.28 ± 0.07	2.67 ±1.11
	90 x I	0.24± 0.32	4.26± 1.6	0.74± 0.48	1.68 ± 1.11	0.34± 0.18	0.31± 0.05	2.57± 1.17
	90 x NI	0.12 ± 0.12	4.35 ± 1.55	0.75 ± 0.53	1.71 ± 1.1	0.37 ± 0.21	0.25 ± 0.08	2.18 ± 1.07
	p	0.740	0.480	0.633	0.760	0.129	0.955	0.256

Outcomes: Fermentation and digestibility analyses of ensiled duckweed allow for future consideration in feed rations for ruminants.

<u>Objective 3: Develop a baseline model for processing duckweed into useable formats for livestock feed</u>

The first step on the baseline model development process was a brainstorming session with the project team members. The goal of the meeting was to identify priority processing scenarios for additional analysis

Figure 4 demonstrates the process flow diagrams resulting from the brainstorming meeting.

Figure 4. Priority process flows resulting from a brainstorming meeting regarding potential end uses of duckweed.

Two end use options were selected for future consideration: (1) animal feed or feed supplement; and (2) bioproducts, like nanocellulose. For both end use options, dry mass, water and energy flow analyses can be described using steady-state balances, shown in Equations 2 to 4, respectively.

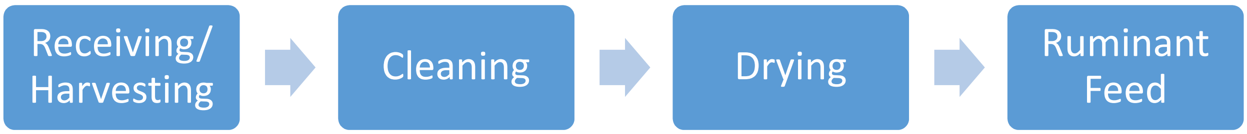
$D_{final} = D_{initial} - \sum_{p=1}^{f} D_p$ (Eq. 2)

$W_{final} = W_{initial} - \sum_{p=1}^{f} W_p$ (Eq. 3)

$E_{final} = E_{initial} - \sum_{p=1}^{f} E_p$ (Eq. 4)

Where:
D = dry matter (kg)
W = water (kg)
E = Energy (J)
p = process
f = number of processes

Option 1: Processing duckweed for feed or feed supplement
There are 4 units included in this model: 1) receipt or harvest of raw duckweed; 2) impurity removal; and 3) drying to produce ruminant (or non-ruminant) feed.

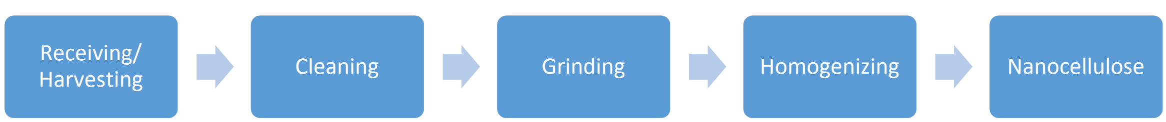


Raw duckweed can contain some impurities such as plastics, leaves, branches, or small stones, from both the storage location or acquired during harvesting and transportation. A cleaning unit is important to ensure duckweed is free from impurities to avoid contamination as part of animal feed. The cleaning process will result in dry matter loss, water loss (particularly as part of the impurities) and energy use, regardless to the mechanical or manual matter by which the duckweed is cleaned.

Harvested duckweed generally has very high moisture content, upwards of 95%. A drying unit in the model is necessary to reduce the moisture content of duckweed for storage or feed use, to below 35%. There potential drying methods are sun-drying or air-drying. Drying will result in water and minimal dry matter loss, and consume solar and/or electrical energy in the process.

Other requirements that will influence the processes are the desired moisture content and particle size for feed purposes.

Option 2: Processing duckweed for bioproducts (e.g. Nanocellulose)
Duckweed cell walls contain cellulose that can be extracted and upgraded into nanocellulose. This nanocellulose can be used for many applications such as food packaging or biomedical materials. The particle size of nanocellulose ranges from $0.3 - 0.5\ \mu m$, but the particle size of raw duckweed is generally between 1-10 mm, which is 20,000 times bigger than that of nanocellulose. In order to reduce the particle size of raw duckweed to the size range of nanocellulose, two stages of size reduction are required. Therefore, a processing model to convert duckweed into nanocellulose includes at least five operation units: 1) receipt or harvest of raw duckweed; 2) impurity removal; 3) grinding; and 4) homogenizing to produce nanocellulose.



Similar to option 1, cleaning is important for a quality end product, and will result in water and dry matter loss and energy use. After cleaning, duckweed will go through two steps of size reduction. The size of raw duckweed is reduced to a range of $1 - 4\mu m$ in the grinding unit using a hammer mill. After grinding, the duckweed can be homogenized to reduce duckweed particle size to less than 1 μm. There is water and dry matter loss during duckweed grinding and homogenizing, respectively. A large amount of energy will be consumed in these processes.

Outcomes: Process flows and realistic end uses of duckweed enable future experimentation and optimization experimentation.

Additional Activities and Outcomes

Two post-graduate students were engaged in this project. One of the students will present this work at the American Society of Agricultural and Biological Engineers Annual International Meeting in July 2017.

Baseline data is now available for future proposals to environmental, engineering, animal science and/or bioprocessing funding programs related to the larger goal demonstrated in Figure 1.

The project enabled interdisciplinary participation and cooperation of both faculty and students.

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Evaluating Nutrient Best Management Practices to Conserve Water Quality (Year 2)

Basic Information

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Evaluating Nutrient Best Management Practices to Conserve Water Quality

Final Report
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Prepared By:
Laurent Ahiablame, Assistant Professor
Department of Agricultural and Biosystems Engineering, South Dakota State University,
Brookings, SD 57007

Shikha Singh and Sandeep Kumar
Respectively Graduate Research Assistant and Assistant Professor
Department of Plant Science, South Dakota State University,
Brookings, SD 57007

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Executive Summary

The extreme winter conditions prevailing in the state of South Dakota make it difficult for the livestock producers to manage the manure generated at the farm. The South Dakota Department of Environmental and Natural Resources does not recommend manure application in the state during the winter months when the ground is frozen. Thus, producers are left with the options such as storing the manure over a longer period until summer or spreading on snow or frozen ground. Storing manure for longer duration leads to increased risks of concentrated spills into the streams. Thus, it is important to develop management strategies for manure to reduce negative impacts to the environment. The present study was conducted to test the hypothesis that manure spread near the outlets of the watersheds would lead to an increased loss of nutrients as compared to the manure spread away from the watershed outlets. A paired watershed study was established near Colman, South Dakota, in which two watersheds were used as treatment watersheds while one was used as control. The watersheds were named as north (NW), south (SW) and east (CW) watersheds; north and south were treatment watersheds while east was the control. The North watershed received manure application on 50% area close to its outlet while south watershed received manure 50% of its area away from its outlet. At the East watershed and the areas in the north and south watershed that did not receive any manure, inorganic fertilizer was applied to meet the nutrient needs for the crop growth. Surface runoff was measured from the three watersheds, and runoff samples were collected from 2013 to 2015 to assess the impacts of manure application on water quantity and quality. Soil samples were also extracted from the three watersheds to measure the physical and chemical properties as impacted by the manure treatment. In addition, soil erosion was estimated using the Revised Universal Soil Loss Equation 2 (RUSLE2) model. Results from this study showed that soil quality, organic matter and water infiltration improved in the landscape positions that received the manure application. Manure improved the infiltration capacity of the soil and also improved the nutrient status of the soil. Runoff data did not show any particular trend among the three watersheds, rather, it varied according to the precipitation pattern and the topography of the watershed. The runoff depth was not statistically significant across the three watersheds. The north watershed showed the highest loss of nutrients into the streams while the south watershed showed the lowest. The east watershed also showed high nutrient losses which may be due to high solubility of the inorganic fertilizers. Soil erosion results showed that topography (LS factor in RUSLE) played the most important role in determining the soil erosion. Our soil erosion estimation results were coherent with the results obtained for the total suspended solids. Thus, it can be concluded that manure treatment in the south watershed showed best results in terms of reduced water quality impairment and soil erosion as nutrient concentrations in the surface runoff samples were significantly higher from the NW as compared to the other watersheds. Results from this study would provide an insight to the producers about managing manure during winter months. In addition, monitoring water quantity and quality for longer duration is strongly encouraged to assess the impacts of manure on soils and water.

1. Introduction

Manure, an organic substance obtained from animal waste, is a rich source of plant nutrients (Gruhn et al., 2016; Wijnja, 2016). Nutrient contents in manure vary based on the animal species (Sommer and Hutchings, 2001). For example, cattle manure contains 76% dry matter, 34% organic matter, 1.9% nitrogen, 0.6% phosphorus, and 1.4% potassium (Cruz, 1997; Sommer and Hutchings, 2001). Appropriate and recurring manure application can increase soil organic matter (SOM) content, which in turn increases plant nutrient availability, promotes plant growth, and facilitates nutrient cycling (Abawi and Widmer, 2000). Manure contributes to soil fertility (Lupwayi et al., 2014) and when properly managed in the fields, manure contributes to economical gain through increased crop productivity, and environmental benefits through improved soil resilience to variations in climate, cropping system, and management (Kongoli and Bland, 2002). However, application of manure at inappropriate landscape position, time and amount can impair the quality of receiving water bodies. During high intensity precipitation events, surface runoff may lead to manure washing off into nearby water bodies. Manure and nutrients originating from manure entering surface waters from agricultural fields may result in nutrient enrichment, eutrophication, and pathogen enrichment, rendering the water unsafe for recreational activities or drinking purposes and create hypoxic conditions for the aquatic ecosystem (Haack et al., 2015). Thus, to minimize the risk of movement of manure into surface waters, several states in the United States have established minimum setback distances between the point of manure application and waterways (Haack et al., 2015).

The concept of maintaining a setback distance could be of help in all the seasons, especially during winter months when the soil is frozen and covered with snow. Solid manure can be applied to these frozen soils only if the slope is less than 4% (USDA, 2012), however, this varies from state to state. A setback distance of 91 meters for water conveyance systems, and 305 meters for lakes, rivers and perennial streams is recommended (USDA, 2012). Some states such as South Dakota do not recommend manure spreading on frozen soils (South Dakota, DENR, 2008; USDA, 2012) because of extreme winter conditions. In these situations, proper guidelines about manure application and management practices in managing agricultural waste during winter to minimize the risk of water quality degradation, are strongly needed. Spreading manure on soils during extreme winter conditions in the Upper Midwestern states have advantages and disadvantages. The advantages include that lower soil compaction occurs while driving heavy machinery on the frozen soil for manure application, less manure storage space is required and the risk of concentrated spills into streams is reduced, while the major disadvantage is that manure can be washed off into nearby streams during snowmelt events. Manure application on frozen soils can also lead to ammonia volatilization as frozen soils do not foster infiltration and typically does not allow for mechanical incorporation into the soil (Hayashi et al., 2003); thereby, reducing nutrient content in the soil profile.

Not applying manure during the winter period and storing it may lead to risks of concentrated spills into streams and rivers. In addition, it may be difficult for farmers to store manure during winter months in the upper Midwest states, with a large number of cattle operations. In South Dakota, there are substantial animal farms that raise beef cattle, hogs, lambs, sheep for wool production, therefore, a huge amount of agricultural waste is generated from these farms. Managing agricultural waste on these farms during winter months may be challenging. A range of various factors that include type and application method of manure, soil type, slope, ground cover and precipitation impact runoff from the agricultural fields that receive manure application (Gilley and Risse, 2000; Tomer et al., 2016). Long-term application of manure improves soil properties and water infiltration, and tends to reduce runoff and soil loss (Ahmed et al., 2013; O'Flynn et al., 2013). The benefits of manure application may not be fully realized until several years after application, and long-term experiments under field conditions are required to determine the impacts of manure application on runoff (Gilley and Risse, 2000). It has been reported that monitoring of runoff that occurs from natural precipitation events from a long-term field scale plots is potentially an efficient way to identify the effect of manure on runoff water quantity (Gilley and Risse, 2000). Runoff water carries nutrients, fertilizers, chemicals, pesticides, and various harmful bacteria which may impair the water quality. These pollutants, carried by runoff water, are known as non-point source (NPS) pollutants. Agricultural fields are the major contributor of NPS pollution to streams and rivers in North America (Berka et al., 2001; Kellogg et al., 1994; USEPA, 1996). Intensive agricultural practices release considerable amounts of NPS pollutants such as nitrogen and phosphorus into the streams (Monaghan et al., 2005). Therefore, best management practices need to be implemented to control the NPS pollutants from the agricultural fields that are receiving excessive amount of manure.

Understanding the relationships between runoff quantity and quality, and the type, timing, rates, and methods of manure application can help in developing best manure management practices to improve water quality. Assessing ways for appropriate winter manure spreading can have positive impacts on soil and water quality. The present study was based on the hypothesis that application of manure at the higher landscape (upslope) position will lead to less water quality problems compared to manure application at the lower landscape (downslope) position.

2. Objectives

The objective of this study was to evaluate the impacts of different field manure application on runoff quantity and quality. The study has the following three objectives:

To evaluate the response of soil nutrients and selected properties (soil pH, electrical conductivity, soil organic carbon, available phosphorus, soil total nitrogen, water retention, bulk density, water infiltration rate) to winter manure application.

To evaluate the response of water quality to winter manure application during winter months.

To estimate the soil erosion using RUSLE (Revised Universal Soil Loss Equation) model and simulate the impacts of various management practices on soil erosion.

3. Materials and Methods

3.1. Study Watersheds

The experiment was conducted at field scale in Egan Township, Moody County, South Dakota. Three different watersheds (Fig. 3.1) named North Watershed (NW), South Watershed (SW) and East Watershed (CW) were established on Egan – Ethan complex and Wentworth – Egan complex, as determined from the Web Soil Survey. The area of the NW is 2.71 hectare, SW is 4.13 hectare, and control watershed (CW) has an area of 2.75 hectare. The three watersheds are present in the same field and have been managed with similar management practices and crops (corn-soybean rotation). The design of the watershed treatments was paired watershed design (Clausen and Spooner, 1993). The NW and the SW were treated with manure, and the treatments that were applied were: manure was spread on the NW to the one half of the watershed located lowest in the terrain while on the SW, manure was spread to the one half of the watershed located highest in the terrain. CW was left without any manure treatment (but received inorganic fertilizers in order to meet the requirements of the crops for their growth) and considered as the control watershed. This treatment was selected to test the hypothesis that the treatment on the SW should have less nutrient and sediment loss as compared to the NW as the distance between the manure treatment and the sampling point of the runoff water, i.e. the outlet of the watershed was more in the SW more time for the water to infiltrate and thereby, reducing nutrient and sediment loss. Manure was spread on the watersheds using a truck spreader (Fig. 3.2) and the uniformity of application was checked using a cross track calibration in 2012. The manure was sufficient to meet the nitrogen demands and nitrogen fertilizer was not as necessary.

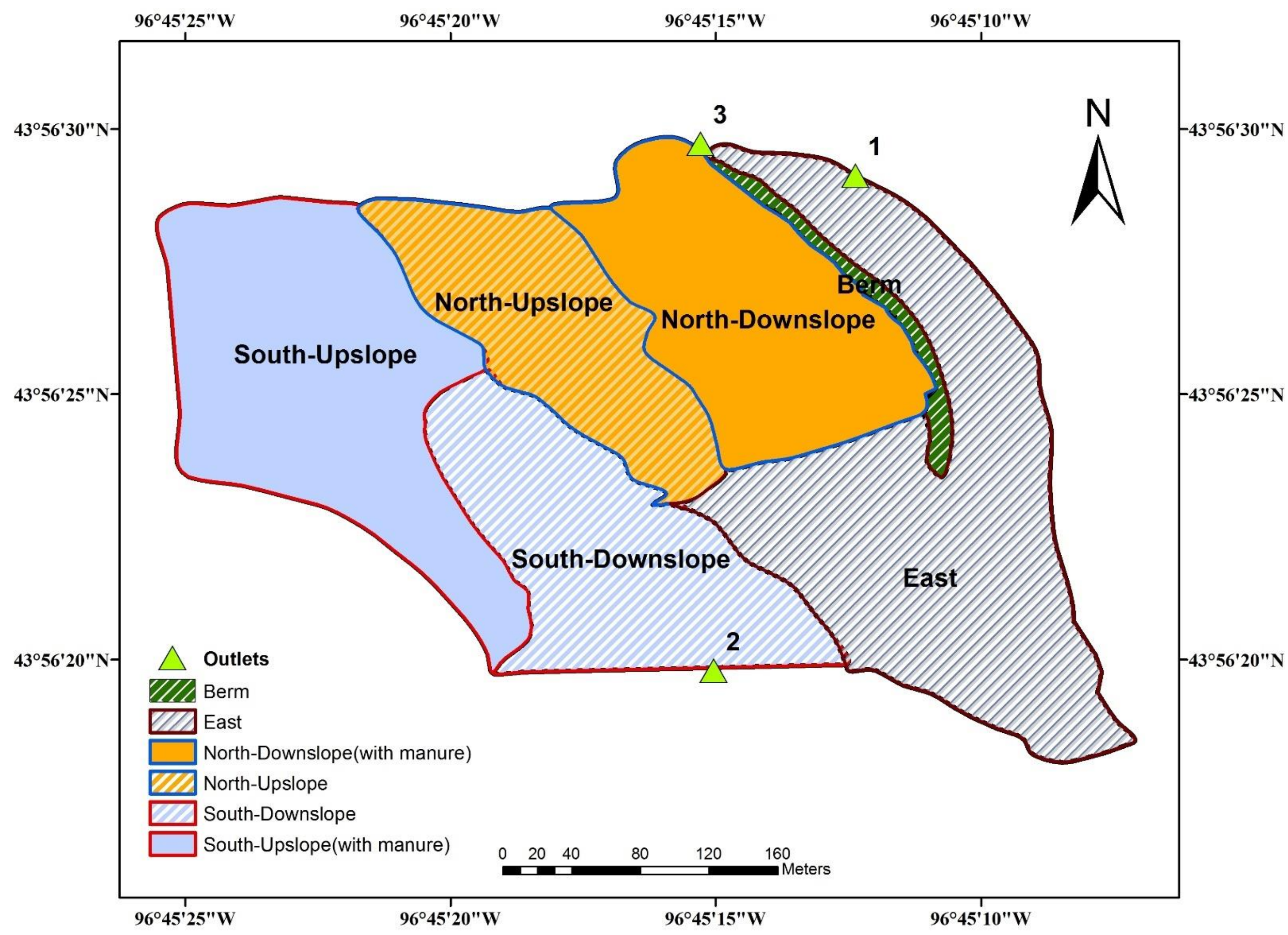

Figure 3.1 The study watersheds at Colman, South Dakota

3.2. Climate Data

The climate data, on a daily temporal scale were obtained from the National Climatic Data Center, National Oceanic and Atmospheric Administration (NCDC NOAA) website from the Flandreau, SD station which is located 16 km away from the study site. To check variation in the rainfall during the study period, annual average rainfall was estimated for the last 20 years and the long term average rainfall was compared to average at this station.

3.3. Soil Quality

Soil auger samples were collected in summer of 2015 from six different landscape positions, viz., NW upslope (no manure treatment), NW downslope (manure treatment), SW upslope (manure treatment), SW downslope (no manure treatment), CW upslope (control) and CW downslope (control) up to 4 depths (0 – 10, 10 – 20, 20 – 30 and 30 – 40 cm). From each landscape position samples were collected in 4 replications (Figure 3.3). A total of 96 samples were collected from the site and were packed in plastic zip lock bags and transported to the laboratory. The collected samples were air dried, crushed and sieved through a 2 mm sieve. The prepared samples were used to analyze soil organic matter, total nitrogen, Olsen phosphorus, pH and electrical conductivity. Water infiltration rate was also measured for all the six landscape positions with the double ring method (20 cm height, and 30 and 20 cm diameter for the inter and the outer rings) using the ponded head method (Reynolds et al., 2002). Infiltration measurements were done in three replicates at each landscape position. Core samples were collected from 2 depths (0-10 cm and 10-20 cm) from all the landscape positions in two replicates. The cores were of 5 cm diameter and 5 cm length. The samples were sealed in plastic zip lock bags, transported to the lab and were analyzed immediately. Bulk density was analyzed using the core method (Grossman and Reinsch, 2002) for both the depths. Total nitrogen and carbon were determined using the TruSpec CHN Analyzer (LECO Corporation, St. Joseph, MI) and the inorganic carbon was determined using the hydrochloric acid method (Stetson et al., 2012). The difference between the total carbon and the inorganic carbon was soil organic carbon (Stetson et al., 2012). Soil available phosphorus was determined using the Olsen method (Olsen, 1954). Electrical conductivity (EC) and pH were measured using Orion star pH and EC meter using 1:1 and 1:2 soil: water ratio, respectively. Soil water retention (SWR) was determined using tension and pressure plate extractors (Klute and Dirksen, 1986). Intact soil cores were saturated using capillarity for more than 24 hours before determining the SWR at 0, -0.4, -1.0, -2.5, -5.0, -10.0 and -30 kPa matric potentials.

Figure 3.2 Manure spread using a spreader in 2016

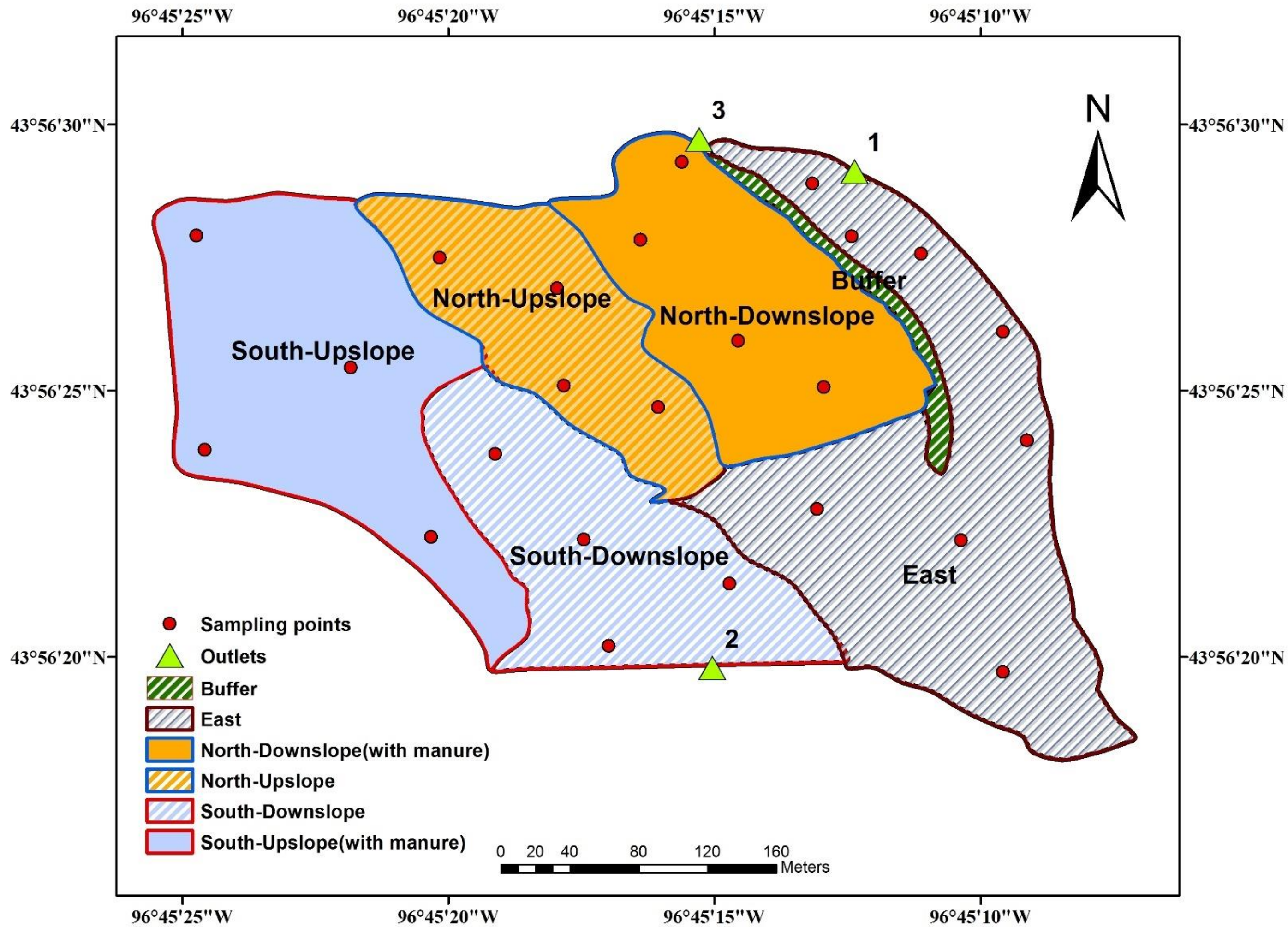

Figure 3.3 Study area map showing the soil sampling points

3.4. Surface Runoff Quantity

The H-flumes were installed at the outlet of each watershed to monitor the surface runoff from each watershed outlet. The peak flow was recorded with the help of H-flume and the depth of the water flowing through the flume was recorded by ultrasonic depth sensor (SR50A) (Campbell Scientific, Logan, UT). Due to extreme weather conditions in the state of South Dakota, monitoring of runoff was difficult because the water used to freeze before it came out of the flume which may have caused erroneous readings of the runoff. In addition to that, it was observed that sometimes the depth of water exceeded the normal depth of 18 inches, which could be due to some rodents or bovines or some other animals sitting onto them or standing next to the sensor which triggered the ultrasonic depth sensor to record the depth wrongly. To avoid this problem, a digital camera (Moultrie M80 GameSpy) (Moultrie, Birmingham, AL) was added near the outlet of each flume location. This addition helped to find out if the runoff was occurring and if the water froze within the flume. The power to the sampler was provided by a 12 V battery and the battery was recharged with the help of 10 W solar panel. Runoff for this study was monitored from 2011 to 2016. Runoff depth was converted into flow rate (cubic feet per second) using the exponential flow equation mentioned below (Brakensiek et al., 1979):

If the depth (x) was less than or equal to 0.75 ft, the flow equation used was:

$$\text{Flow rate in cfs} = 0.6933*(x^3) + 1.3559*(x^2) + 0.0568*(x) - 0.0003 \tag{3.1}$$

If the depth (x) was greater than 0.75 ft (upto 1.5 ft), the flow equation used was:

$$\text{Flow rate in cfs} = 0.2851*(x^3) + 2.6216*(x^2) - 1.2703*(x) + 0.4597 \tag{3.2}$$

$$\text{Flow in } m^3\ s^{-1} = \text{flow in cfs} * 0.0283$$

3.5. Surface Water Quality

Water samples were collected by Teledyne ISCO automatic samplers (model number 6712) (ISCO, Lincoln, NE) during the years 2013 and 2014 and using Campbell Scientific (Campbell Scientific, Logan, UT) automatic samplers during 2015 and 2016. These samples were collected during each runoff event and placed in a cooler and transported to the South Dakota Agricultural Laboratories in Brookings, South Dakota. The samples were analyzed for total Kjeldahl nitrogen, nitrate nitrogen, ammonium nitrogen, total dissolved phosphorous, total phosphorus and total suspended solids. The samples were tested by the following standard methods. The total Kjeldahl nitrogen was estimated using the method provided by EPA 351.3 (Colorimetric, Titrimetric, Potentiometric method); ammonia nitrogen was tested by EPA 350.2 (Colorimetric, Titrimetric, Potentiometric Distillation Procedure); nitrate nitrogen by SM 4110B (Ion chromatography with chemical suppression of eluent conductivity); total phosphorous by SM 4500PE (Ascorbic Acid method); total dissolved phosphorous by SM 4500B&E (Sample digestion and Ascorbic acid method); total soluble solids by SM 2540D (Total solids dried at 103-105^{0}C) (Federation and Association, 2005). After getting all the results for each sample, the representative concentration of nutrients was calculated for each storm event for all the three watersheds. There were several storm events where numerous samples were collected and to represent the representative concentration of each pollutant in the sample, flow weighted mean concentration method was used in which mass load was calculated first (equation 3.3), followed by flow weighted mean concentration (equation 3.4) (Smith et al., 2003).

$$\text{Mass load} = \sum c\ q\ t \qquad (3.3)$$

where, c = sample concentration (mg m^{-3})

q = instantaneous streamflow ($m^3 s^{-1}$)

t = time interval (s)

$$\text{Flow weighted mean concentration} = \frac{\text{Total mass load}}{\text{Total stream flow volume}} \qquad (3.4)$$

3.6. Statistical Analysis

Statistical analysis of water quality for the three watersheds was performed using SAS 9.3 software (SAS Institute, 2012). The distributions of the data sets was tested for composite normality using the Kolmogrov-Smirnov test and histogram method. Parallel line analysis was used for comparing the mean differences of each pair of water quality parameters (nitrate nitrogen, ammonia nitrogen, total Kjeldahl nitrogen and total dissolved phosphorus) under different watersheds as the data are time correlated values and interdependent. A log transformation was used when residuals were not normally distributed. In addition, an estimate for the least significant difference (Duncan's LSD) among treatments was obtained using the 'Mixed procedure' in SAS (2007) for the soil data to assess the treatments impact on measured parameters (soil pH, EC, available P, soil organic carbon, total nitrogen, soil water retention and infiltration rates). Statistical differences were declared significant at $\alpha = 0.05$ level.

3.7. RUSLE2 Model

The RUSLE equation was used to predict the annual soil loss from the three watersheds. RUSLE2, being an empirical model, helped in estimating the soil loss after letting us apply the manure management as mentioned in the study site description. RUSLE has six parameters which help in measuring the annual soil loss.

$$A = R*K*LS*C*P \qquad (3.5)$$

where, A is the average annual soil loss (t ac^{-1} yr^{-1}), R is the rainfall erosivity factor, K is the soil erodibility factor, LS is the topographical factor which includes slope length and slope steepness, C is the vegetation cover and P is the conservation practices. LS, C and P factors are dimensionless.

For calculating the R factor for the equation in RUSLE2, as soon as we enter the location of the study area, it automatically takes up the R factor for that place. For out study area, the R factor picked by the model was 100.

Soil erodibility factor (K) is the susceptibility of the soil to erosion (Wischmeier and Smith, 1978). The K value depends on the soil properties (soil texture, soil structure, soil organic matter content, etc) (Wischmeier and Smith, 1978). For the K value as well, when we entered the soil series for the study area, it automatically took up the K factor for the soil and for our study area it took the alue as 0.26.

Slope length and slope steepness were determined using the digital elevation model using the 10m by 10m resolution in ArcGIS and LS factor was calculated using the following equation (Morgan, 2006):

$$LS = \left[\frac{Q_a\, M}{22.13}\right]^n \times (0.065 + 0.045 \times s + 0.0065 \times s^2 \qquad (3.6)$$

where, where Q_a denotes flow accumulation grid; s is grid slope in percentage; M is grid size; n has constant value of 0.2-0.5.

C factor which represents the soil cover, soil biomass, and soil disturbing activities on erosion was determined using the RUSLE guide tables (Morgan, 1995). For just soybeans, the C factor value was 0.39 while for the landscape positions which had manure treatment and the soybeans with them, the C factor was 0.34 (Gabriels et al., 2003) as C factor takes into account the surface soil cover which intercepts the runoff and reduces the soil erosion and in this present study, manure was applied (was not incorporated) so as check its impact on soil erosion and runoff.

P factor denotes the support practices and range between 0 to 1 (Renard et al., 1997). P is normally assumed to be unity. P value is decreased when we apply practices such as strip cropping, terracing and contouring. At the end, the soil erosion was calculated by multiplying all factors together. After calculating the soil loss, we created scenarios of increasing the residue cover on the watersheds so that they could trap the sediments and prevent soil erosion. Similarly, we tested how the installation of terraces as support practice to reduce the erosion. We chose these two practices as this would be easy for the producers to employ and does not require any additional expense.

4. Results and Discussion

4.1. Climate and Soil Data

The amount of precipitation the study site received was 6% and 12% below the long-term mean 669 mm in 2013 and 2014, respectively. In the year 2015, the annual precipitation was 25% higher than the long-term average. The climatogram for the study area has been shown in Figure 4.1.

The soil data for the 2 depths (0 – 10 cm and 10 – 20 cm) is shown in Table 4.1 and 4.2, respectively. Soil pH for the 0 – 10 cm depth of north upslope and downslope were slightly acidic with the values of 5.8 and 6.3, respectively. However, soil pH for south upslope and downslope were 5.4 and 5.1, and 4.7 and 4.8 for east upslope and downslope, respectively. There were no significant differences observed in pH values between the landscape positions for each watershed, however it was observed that manure applied landscape positions had numerically higher pH values indicating that manure application increased the soil pH. For the 10 – 20 cm depth (Table 4.2), pH for all the landscape positions did not show any statistical significance. However, it was seen that manure treated landscape positions (north downslope and south upslope) showed an increased pH numerically, though statistically non-significant. A study conducted in China by Whalen et al. (2000) reported that application of manure in soils lead to an immediate and persistent effect on soil pH and showed an increase in the pH value, although the differences were not significant. Another study conducted by Walker et al. (2004) in Southwestern Spain reported that application of manure showed a considerable increase in the soil pH.

It was observed that there was a significant buildup in the 0 - 10 cm depth of soil nutrients in the areas manure was applied (Table 4.1, Figure 4.2). The north downslope of the NW had higher total nitrogen (2.4 g kg^{-1}) compared to the north upslope (2.0 g kg^{-1}), although the difference was not statistically significant. At the SW, the south upslope had significantly higher (25% more) nitrogen content than the south downslope. There were no differences observed in the nitrogen content between upslope and downslope positions of the control watershed (i.e. CW). There were some statistical differences observed in the watersheds when comparing upslope and downslope (averaged across the watersheds) which may be due to the soybean crop grown during the sampling period (summer 2015) which helps in nitrogen fixation. For the second depth (10 – 20 cm), it was observed that there were no significant differences among the landscape positions of each watershed (Table 4.2), however, the manure treated landscape positions showed numerically high value as compared to the untreated ones. A long term (17 years) study conducted in China by He et al. (2015), concluded that manure significantly increased the nitrogen content in the soil. Similar results were reported by Mancinelli et al. (2013).

For soil available phosphorus in the 0 – 10 cm depth, there was a significant higher build up in the north downslope which was treated with manure than the north upslope, except in the SW (Table 4.1). The available P content was about 61% higher at the north upslope compared to the north downslope. At the SW, the manured south upslope had numerically higher available P content than the south downslope, but statistically insignificant (Table 4.1). This may be due to manure applied to these specific landscape positions. In the CW, the upslope was significantly lower than downslope which may be due to erosional losses from the upslope landscape position to downslope. In addition to that, it may be that the inorganic fertilizers (being soluble in water) may get dissolved in the runoff water and move towards the downslope which may lead to a greater concentration in the downslope. Manure application leads to an increase in available P content in the soil. For the second depth (Table 4.2), there was no significant difference observed in the watersheds except the NW. For the NW, soils in north upslope had significantly higher phosphorus content as compared to the north downslope. A study conducted by Tadesse et al. (2013) reported that application of manure led to a considerable increase in the available P (11.9 to 38.1 mg L^{-1}). Xue et al. (2013) also showed that application of manure on soils in North China Plains led to a considerable increase in the labile and non-labile P pools in the soil. Similar observations were reported by Waldrip et al. (2012), who indicated that organic dairy manure treated soils had higher P than the ones treated with inorganic fertilizers ($p<0.05$).

Soil organic carbon for the 0 – 10 cm depth, was significantly higher in the landscape positions treated with manure compared to untreated landscape position within a watershed (Table 4.1, Figure 4.3). North downslope had 42% higher SOC concentration than north upslope, while south upslope had 37% higher SOC content than the south downslope. For the CW, the downslope was numerically higher than the upslope although it was not statistically significant. For the second depth (Table 4.2), the landscape positions did not show any statistical differences in the SOC except the NW, in which soils in north downslope showed statistically higher SOC than the north upslope. Addition of manure increased the organic matter content of the soils. Similar results were also documented by Eghball et al. (2004) in Nebraska, USA where they reported that total C and total N increased after application of manure. Haynes and Naidu (1998) reported that addition of manure to soils leads to an increase in carbon content which ultimately increased soil microbial biomass and enzyme activity. An increase in microbial activity eventually lead to improvement in various soil properties such as water infiltration, porosity, and water holding capacity (Celik et al., 2004; Eghball et al., 2004; Liu et al., 2010). Similar results were reported by Xie et al. (2014). The results obtained in the present study were consistent with the published literature discussed above.

Electrical conductivity (EC) results did not show any significant differences in the landscape positions within a watershed. However, for the 0 - 10 cm depth (Table 4.1), it was noticed that manure treated landscape positions had numerically higher electrical conductivity than the untreated parts due to the accumulation of soluble salts in the soil (Turner, 2004). A similar trend was observed in the 10 – 20 cm depth (Table 4.2) of the soil profile with no significant differences observed in the landscape positions within the watersheds.

Soil bulk density results (Fig 4.4) also showed that manure application led to a decreased bulk density in the soils. North downslope and the south upslope positions had a lower bulk density value as compared to the north downslope and south upslope, respectively. This may be due to the fact that manure leads to soil aggregation which helped in reduction lowering the bulk density of soils. However, there were no significant differences observed in bulk density values within the watersheds.
The landscape positions treated with manure had higher infiltration rates (Table 4.3) compared to areas not receiving manure although the differences were not always statistically significant. The only statistically significant difference observed in the infiltration rates were in the NW, where north downslope showed increased infiltration rate (14%) compared to the north upslope. In the SW, infiltration rate in south upslope was numerically higher but there was no statistical difference. This may be due to the manure application which helped in the improvement of infiltration rates. For the CW, no statistical significance was observed. Manure improved soil organic matter which ultimately, increased soil porosity and hence improved water infiltration (Gholami et al., 2013). A study conducted by Peng et al. (2016) in China compared four different treatments (inorganic Nitrogen, Phosphorus and Potassium (NPK) fertilizer, NPK plus rice straw mulch, NPK plus rice straw derived biochar and NPK plus swine manure) and reported that NPK plus rice straw-derived biochar and NPK plus swine manure increased infiltration capacities compared to the other treatments due to increases in organic matter and improved soil properties.

4.2. Surface Runoff

Surface runoff from the watersheds differed greatly due to the different treatments among the watersheds, their slope and orientation. However, it was observed that the runoff patterns were not the same in all the study watersheds for all 3 years of the monitoring period. In 2013 (Fig 4.5), SW had a total flow (21 mm) followed by CW (14 mm) and then NW (12 mm) collected during all the precipitation events, which was comparatively lower than the runoff measured during the previous years of the study **(**Adapted from Nathan Brandenburg MS thesis**)**. This may be because of the 6% less precipitation during this year as compared to the long term mean of 669 mm. No statistical difference was observed between the runoff depths across the three watersheds. When statistically compared, NW and CW had no differences ($P<0.68$) and SW and CW had a P value of 0.53. The year 2014 received only one runoff event (Fig 4.6) throughout the year and the observed trend remained similar to the previous years were NW had a runoff of 15 mm, while SW and CW had runoff of 5 and 0.22 mm runoff, respectively. There was only a small amount of runoff in 2014 as it was the driest year of the study. The year 2015 (Fig 4.7) had very high intensity rainfall events with maximum of 20.3 cm (8-inch) rainfall received in a single storm event. The NW produced 23.5 mm of runoff, SW had 21 mm and CW had 20 mm. No statistical significance was observed between the runoff depths of NW and CW ($P<0.62$) and SW and CW ($P<0.87$). It was observed that for all the years, there was no significant difference in the runoff depths measured for the three watersheds. However, there were 5, 1 and 5 runoff sampling events in 2013, 2014 and 2015, respectively in the SW. All the study watersheds did not produce the same number of runoff events due to the difference in their treatments, orientation, slope, topography, etc.

The reduced runoff in 2015 compared to that of previous sampling years was partially due to differences in rainfall pattern and increased infiltration capacity of the soils that have gradually improved during the 5 years of experiment. Chinkuyu et al. (2002) studied manure application on runoff from a Nicollet loamy soil under a corn-soybean system and noted greater runoff in soils with fertilizer application than that of soils which received manure application. A similar field study was conducted by Kongoli and Bland (2002) at Agricultural Research Station, Madison, Wisconsin, in which dairy barn bedding with chopped corn stalks was used in an alfalfa crop. The authors reported that manure application slowed down snow melt in proportion to the application amount, as manure acts as an insulating layer and delays snow melt. This resulted in greater infiltration of water into the soil which reduces surface runoff. Based on studies conducted across various environmental and geographical settings, the runoff loss was less in the fields treated with manure compared to fields without manure (e.g., Gilley and Risse, 2000; Long et al., 1975; Vories et al., 1999; Wood et al., 1999).

4.3. Water Quality

Nitrate Nitrogen: Nitrogen concentration was measured as nitrate nitrogen, for 2013, 2014 and 2015, and the data is shown in Fig. 4.8, 4.9, 4.10, respectively. In 2013 (Fig 4.8), nitrate concentration varied among the three watersheds. For comparison, the maximum nitrate nitrogen concentration in drinking water is 10 mg L^{-1} (US EPA 2012a). Thus, there were generally no nitrate concentration exceedances in the runoff water but for a few exceptions in the year 2015. In 2013, the trend was SW having the significantly less nitrate concentration followed by CW, and NW ($P<0.015$). On March 9, 2013, SW had 3.9 mg L^{-1} of nitrates in the runoff while CW and NW had 4.6 mg L^{-1} and 4.75 mg L^{-1}, respectively (Fig. 4.8). Likewise, On March 14, 26, 27 and 28, 2013, nitrate concentrations in the runoff samples were higher in NW and CW than that of the SW (Fig 4.8). The trend showed that NW had more concentration than the SW. This may be attributed to the fact that CW had no manure treatment due to which runoff was higher which carried higher nutrients loss. In addition, the inorganic fertilizers may have a higher solubility of ions that increased the nutrient loss in the CW. Similarly, the NW had manure application near the outlet, thus reducing the pathway length for transporting manure to the field edge. In 2014 (Fig 4.9), precipitation was 12% below the long-term average. Thus, there was very little runoff collected at the outlet of the study watersheds. On June 16, 2014, there was considerable runoff in the NW and SW during which time one sample was collected from the NW and the SW, while no runoff sample was collected from CW. Based on these events, the NW had higher nitrate concentration in runoff compared to SW, which could not be statistically tested due to limited number of data points. This was because in the NW, manure application was near the outlet of the watershed which led to more nitrate concentration in the runoff leaving the watersheds. In 2015 (Fig 4.10), nitrate concentration values were low for June 6 and 7, 2015 but the trend remained as expected with SW having the least nutrient loss compared to that of other watersheds. On June 19 and 20, nitrate concentrations were 10.5 and 43% lower in SW compared to that in NW, respectively, on these days, whereas, no runoff was collected from CW on these sampling days. On July 6, 2015, the trend remained similar with SW samples having lower (4 mg L^{-1}) concentrations compared to NW (5 mg L^{-1}) and CW (5 mg L^{-1}). Due to the limited number of samples collected from the CW prevented us from applying statistical method in order to know the significant differences in the nitrate concentration of the runoff samples. The nitrate load across the three watersheds and for the three years (2013, 2014 and 2015) has been shown in Fig 4.11. The load was statistically similar across the three watersheds and for all the three years of study. According to Chinkuyu et al. (2002), the nitrate nitrogen was higher in manured soils with high application amount than manured soil with low application and fertilizer applied soil. Similar trend was observed by these researchers for the phosphate loss. Zhang et al. (1996) reported that fertilizer application to soils lead to an increase in the nitrate nitrogen content in the drinking water (300 mg L^{-1}).

<u>Ammonium Nitrogen</u>: Manure is a rich source of ammonium nitrogen and organic nitrogen (Hooda et al., 2000). Runoff samples in this study were also tested for ammonium nitrogen. In 2013 (Fig 4.12), the trend was NW and CW had higher ammonium loss than the SW and the results were statistically different. Ammonia loss from NW was significantly higher than that of CW ($P<0.02$), while there was no statistically significant difference in ammonia loss between SW and CW ($P<0.99$). This can be explained by the fact that this was the third year of the experiment, and manure application near the outlet over the years gradually increased ammonium build up in that part. It may also be attributed to the fact that since manure rate was within to the crop

need requirement but the rainfall that occurred during that year was less, compared to the previous years may have increased the concentration of ammonia in runoff samples. Thus, the combination of manure application near the outlet and reduced rainfall led to the increased concentrations of ammonia in samples collected from the NW. A similar trend was observed in the year 2014 (Fig 4.13) with the NW having higher concentrations than the SW but statistical analysis was not performed due to just one sampling event in 2014. In 2015 (Fig 4.14), ammonium loss was low, below 1 mg L^{-1} in all three watersheds. This may be attributed to the fact that manure application in 2015 occurred late March due to absence of the snow cover, leading to easy volatilization of ammonia. On June 6 and 7, 2015, the CW had the highest ammonia loss, while NW and SW were roughly the same throughout the year. The ammonium nitrogen load across the three watersheds has been shown in Fig 4.15 and no differences were observed across the three watersheds and for all the three years of study.

<u>Total Kjeldahl Nitrogen (TKN)</u>: When runoff occurs it may carry nitrogen in organic form with it into the water bodies and cause water quality threats. The trend of TKN loss was similar to the other nutrients, however, in 2013 TKN was not significantly different between NW and CW ($P<0.04$), while it was statistically similar between SW and CW ($P<0.72$) (Fig. 4.16). The low TKN in SW may be due to manure treatment on the upper 50% terrain in this watershed and the distance or nutrients to travel to the outlet was more from the manure application point. The NW had higher concentrations because the manure treatment was on the lower 50% terrain, near the outlet. Thus, when runoff occurred, nutrients in manure were quickly carried to the outlet of the watershed impaired the water quality. The CW, which was the control and had higher concentration of TKN in the runoff samples. This may be because this watershed had no manure in it, therefore, the runoff was more and carried all the fertilizers with it. For the years 2014 (Fig 4.17) and 2015 (Fig 4.18), trend in TKN coincided with that of previous years but there were limited number of sampling events in the CW, and statistical analysis was not performed. The Kjeldahl nitrogen load across the three watersheds has been shown in Fig 4.19. No statistical difference was observed across the three watersheds and the three years of study.

<u>*Total Dissolved Phosphorus:*</u> Phosphorus loss from the watersheds were measured as total dissolved phosphorus (TDP) which contains all soluble organic and inorganic forms of phosphorus present in the water sample after filtering. As expected, the TDP was lower in SW and the trend was the same for all the three watersheds. In 2013 (Fig 4.20), the TDP loss from the watersheds showed no statistical differences observed between SW and CW ($P<0.99$) but TDP from NW was statistically higher than from CW ($P<0.04$). Again, this may be due to the combination of reduced rainfall and manure application near the outlet in the NW. However, for 2014 (Fig 4.21) and 2015 (Fig 4.22), the trend

was similar to the previous years' trends with SW samples having the least concentration of TDP. Statistical analysis was not performed for these years due to the limited number of sampling events. Vadas et al. (2004), concluded the similar findings. The dissolved phosphorus load has been shown in Fig 4.23 and no statistical differences were observed across the three watersheds.

Total Phosphorus: The total phosphorus concentration in the surface runoff samples were followed a similar trend as the other nutrients. Surface runoff samples collected from the NW had a higher phosphorus loss than the CW ($P<0.035$) and no differences were observed between the SW and CW ($P<0.93$). SW water samples had the least phosphorus content as compared to the other two watersheds. On March 9, 2013 (Fig 4.24), the phosphorus loss was low as compared to the other days of sample collection. As we move ahead, we see there is an increase in the phosphorus loss from all the watersheds. This may be due to thawing of snow taking place as we move ahead. There were no significant differences observed between the samples of SW and CW while NW samples had significantly high concentration of TP than CW. In 2014 (Fig 4.25), just one sampling event took place in which no sample was collected from the CW. This may be due to topography of the CW. However, the total phosphorus loss from the NW and the CW, there was not much difference. We could not test the values statistically due to less number of data points in the whole year. Again, in 2015 (Fig 4.26), it was observed that NW water samples had higher phosphorus content as compared to the SW and CW. This may be again to the fact that manure application in NW near the outlet led to a higher loss of phosphorus. Again, we could not test the values statistically due to less number of data points available. Sharpley et al. (1994) reported that addition of manure to soils may lead to an increase in the phosphorus loss into the streams, thus, management of agricultural phosphorus is very important so as to reduce the loss into the streams and reduce the environmental concern that would arise as a result of phosphorus accumulation in the water. The total phosphorus load has been shown in Fig 4.27. No statistical difference was observed across the three watersheds.

Total Suspended Solids: TSS content in the surface runoff samples followed the same trend as that of the other nutrients with the NW being the highest among the three watersheds. In 2013 (Fig 4.28), the samples were collected during march. On march 9, 2013, it was observed that TSS loss from the three watershed was the least as compared to the other days of sample collection. As we move forward, it was observed that TSS increased which may due to thawing which made the soil loose and susceptible to move. The trend for TSS showed that NW samples were significantly higher than the CW which may be due to the shape and slope of the CW, while there was no significant difference between SW and CW. In 2014 (Fig 4.29), only one sample was collected but SW samples showed an increase in the TSS as compared to NW (no sample collected from CW). This may be due to larger area of SW than NW. The effect of rainfall or precipitation could not be taken into account as 2014 was the driest period of the study. We could not test the data for statistical significance due to an insufficient number of data points. In 2015 (Fig 4.30), again NW samples showed highest loss of TSS than the SW and CW. CW samples had comparatively lower TSS content due to the topography of the CW. CW is a long and kind of a flat watershed which may stop the sediments from moving to the outlet. Another reason may be due to the sowing pattern in CW as the planting was done across the slope near the outlet which formed kind of a terrace and helped in trapping the sediments. The higher TSS in NW and SW was due to the very high intensity and frequent rainfall events. In addition to the topography of the watershed also helped in movement of sediments towards the outlet. It was observed that in 2013, when the samples were all collected during March, the TSS was lower as compared to 2014 and 2015, due to snowmelt while in 2014 and 2015 the samples were collected during June and July. In June and July, runoff occurred as a result of rainfall events which can accelerate erosion due to its impacts on the bare soil. Vegetation can reduce this impact but not completely. Thus, in 2014 and 2015, the TSS values were higher as compared to 2013. The sediment load across the three watersheds has been shown in Fig 4.31 and no differences were observed across the three watersheds.

4.4. Soil Erosion Estimation Using RUSLE

As mentioned in the materials and methods section, soil erosion can be determined using the RUSLE equation in the RUSLE2 model. The six factor (R, K, L, S, C and P) were determined using the methods described before. The calculated factors have been shown in Table 4.4. It was observed that the R factor remained same for all the watersheds due to the small size of the watersheds which helped the rainfall to be similar among all of them. The K factor was also roughly the same due to similar soil texture throughout the watersheds. The LS factor was the most important part of the equation as it was highly variable among the three watersheds. It was observed that the north downslope having a higher LS factor contributed more to the soil loss even though it was treated with manure. This is coherent with our findings of the total suspended solids in the runoff water. Similarly, south upslope having a lower LS factor contributed less to the soil erosion. This shows that soil erosion was greatly impacted by the topography of the watersheds. Since CW did not have any treatment, we calculated the erosion for the whole watershed together. It was seen that LS factor was quite high but that was for the entire watershed and when you see the watershed physically, it is a long watershed and looks to be flat due to the which the soil loss was not that high. In 2015, the crop was soybean and using the land use map and the RUSLE guide tables, the soybean crop C factor came out to be 0.39 and soybean with manure was 0.34. The P factor was unity for all the watersheds as no support practices were applied to the watersheds.

The estimated soil erosion results are shown in Figure 4.32 and 4.33. It was seen that for the NW, north upslope had lower soil loss as compared to the north downslope due to the lower LS factor. For the SW, south upslope showed lower soil loss as compared to south downslope due to lower LS factor. Thus, it can be concluded that for this study, LS factor played a major role in determining the soil loss across the watersheds.

After creating the scenarios for the model like applying an increased residue cover on the fields and terracing the watersheds, it was observed that as the residue cover increased it led to a huge decrease in the soil erosion. After applying 20% residue cover (Figure 4.34), it was seen that the soil erosion decreased by 44 – 50 % across all the landscape positions. Again after applying 40% residue cover, the soil erosion further decreased over a range of 54 – 64 % in all the watersheds. Thus, it can be concluded that application of residue cover on the soil surface will help in decreasing the soil erosion as they will trap and sediments and prevent them from running into the streams. Another support practice was added to the watersheds (terracing) and the soil erosion was reduced (Figure 4.35) to half of the original estimated erosion as the P factor got reduced to 0.5 after the application of terracing. After seeing all the results, it can be concluded that application of conservation and support practices may be helpful in reducing the soil erosion in the watersheds.

Table 4.1. Soil chemical properties measured at upslope and downslope landscape positions of North, South, and East (Control) Watersheds from depth 0 – 10 cm in 2015

Landscape positions		pH	Electrical Conductivity	Soil Organic Carbon	Available P	Total N
			---µS cm^{-1}---	---g kg^{-1}---	---mg kg^{-1}---	---g kg^{-1}---
North	Upslope (No Manure)	5.8^{a†}	192.6^{a}	17.1^{b}	4.08^{b}	2.00^{a}
	Downslope (Manure)	6.3^{a}	222.8^{a}	24.4^{a}	6.58^{a}	2.40^{a}
South	Upslope (Manure)	5.4^{a}	321.8^{a}	20.3^{a}	3.93^{a}	2.50^{a}
	Downslope (No Manure)	5.1^{a}	253.5^{a}	14.8^{b}	3.29^{a}	2.00^{b}
East	Upslope (No Manure)	4.7^{a}	124.6^{a}	9.00^{a}	0.82^{b}	1.70^{a}
	Downslope (No Manure)	4.8^{a}	155.1^{a}	9.50^{a}	2.60^{a}	1.70^{a}

†Similar letters indicate that there was no significant difference observed between the different landscape positions within the same watershed.

Table 4.2. Soil chemical properties measured at upslope and downslope landscape positions of North, South, and East (Control) Watersheds from depth 10 – 20 cm in 2015

Landscape positions		pH	Electrical Conductivity	SOC†	Available P	Total N
			---µS cm^{-1}---	---g kg^{-1}---	---mg kg^{-1}---	---g kg^{-1}---
North	Upslope (No Manure)	6.5^{a}	154.1^{a}	6.85^{b}	2.5^{b}	1.4^{a}
	Downslope (Manure)	6.8^{a}	191.9^{a}	12.0^{a}	3.8^{a}	1.5^{a}
South	Upslope (Manure)	6.3^{a}	229.5^{a}	15.4^{a}	3.2^{a}	1.8^{a}
	Downslope (No Manure)	5.7^{a}	162.6^{a}	14.6^{a}	2.3^{a}	1.6^{a}
East	Upslope (No Manure)	5.3^{a}	94.40^{a}	4.11^{a}	0.5^{a}	1.0^{a}
	Downslope (No Manure)	5.3^{a}	133.3^{a}	5.97^{a}	0.7^{a}	1.3^{a}

†Similar letters indicate that there was no significant difference observed between the different landscape positions within the same watershed.

Table 4.3. Soil infiltration rates measured at upslope and downslope landscape positions of North, South, and East (Control) Watersheds in 2015

Landscape positions		Infiltration Rate
		---mm hr^{-1}---
North	Upslope (No Manure)	144.4[b†]
	Downslope (Manure)	165.3[a]
South	Upslope (Manure)	195.5[a]
	Downslope (No Manure)	181.1[a]
East	Upslope (No Manure)	139.6[a]
	Downslope (No Manure)	144.8[a]
Slope	Upslope	159.8[a]
	Downslope	163.7[a]

†Similar letters indicate that there was no significant difference observed between the different landscape positions within the same watershed.

Table 4.4. The R, K, LS, C, P factors calculated for each watershed and the landscape positions

Watershed	Treatment	R Factor	K Factor	LS Factor	C Factor	P Factor
North	Upslope	100	0.26	0.81	0.39	1
	Downlsope	100	0.26	1.23	0.34	1
South	Upslope	100	0.26	0.47	0.34	1
	Downslope	100	0.26	0.99	0.39	1
East	Upslope	100	0.26	0.45	0.39	1
	Downslope	100	0.26	0.60	0.39	1

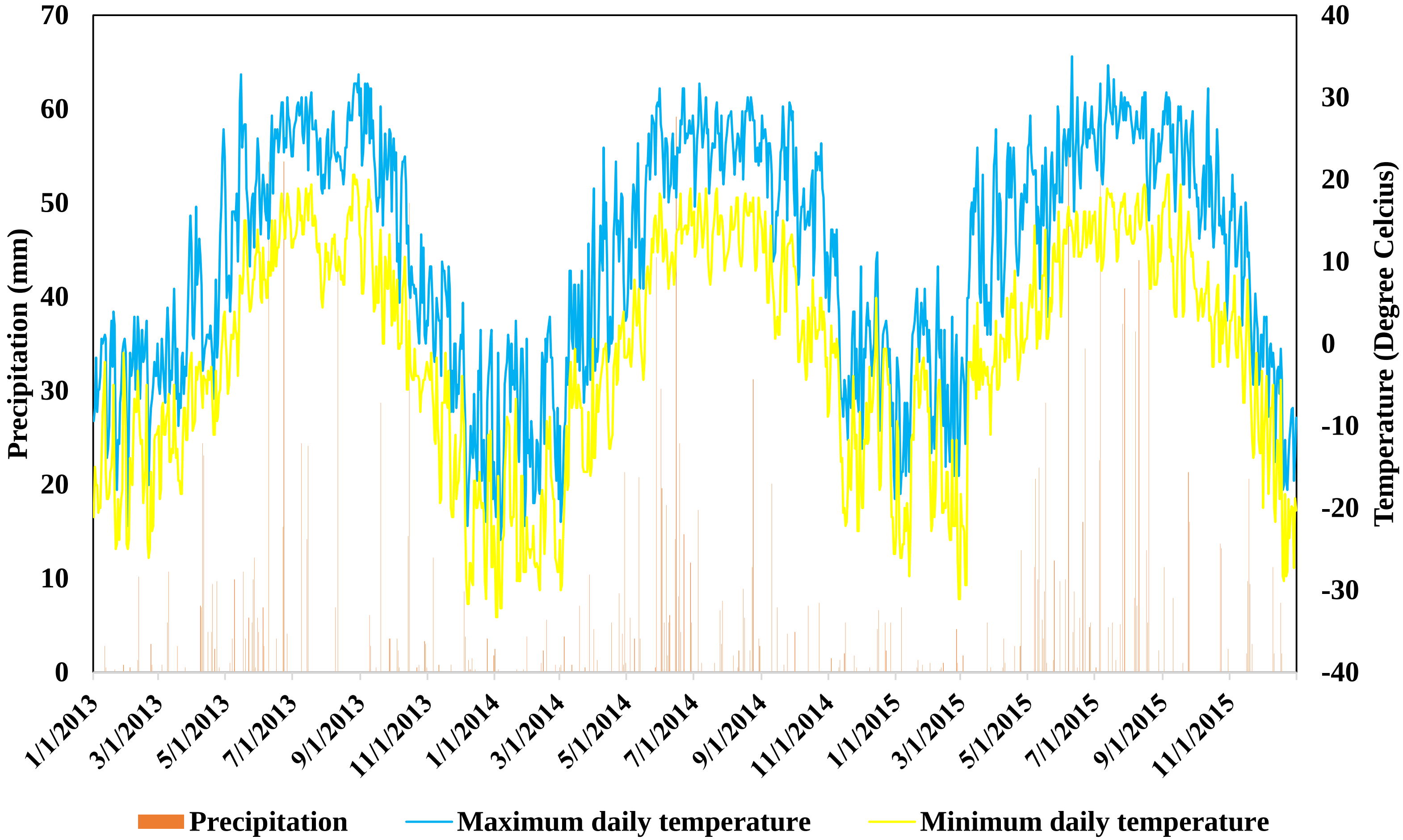

Figure 4.1 The climatogram for the study area for 2012-2015

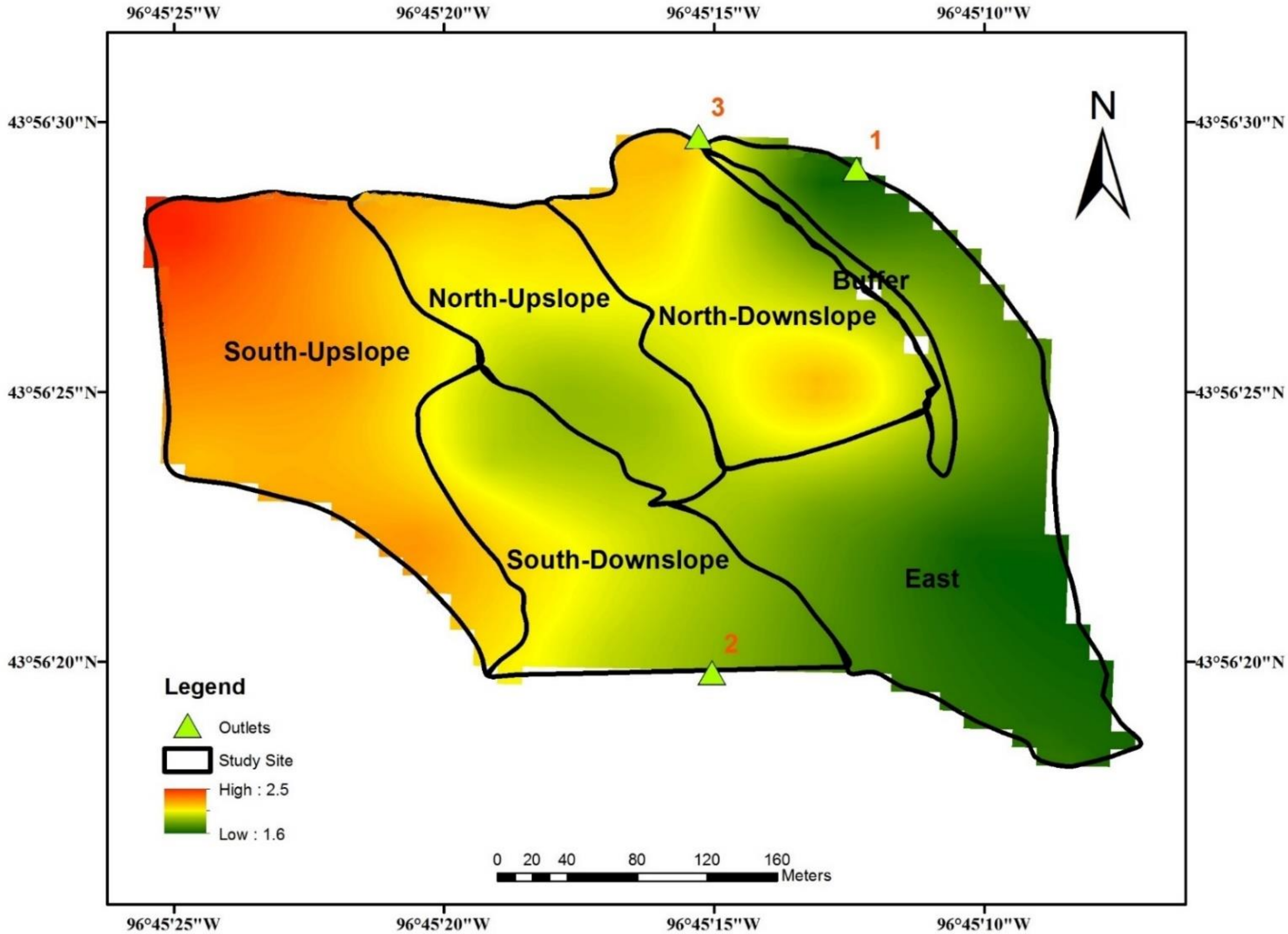

Figure 4.2 Spatial distribution of total nitrogen across the three watersheds

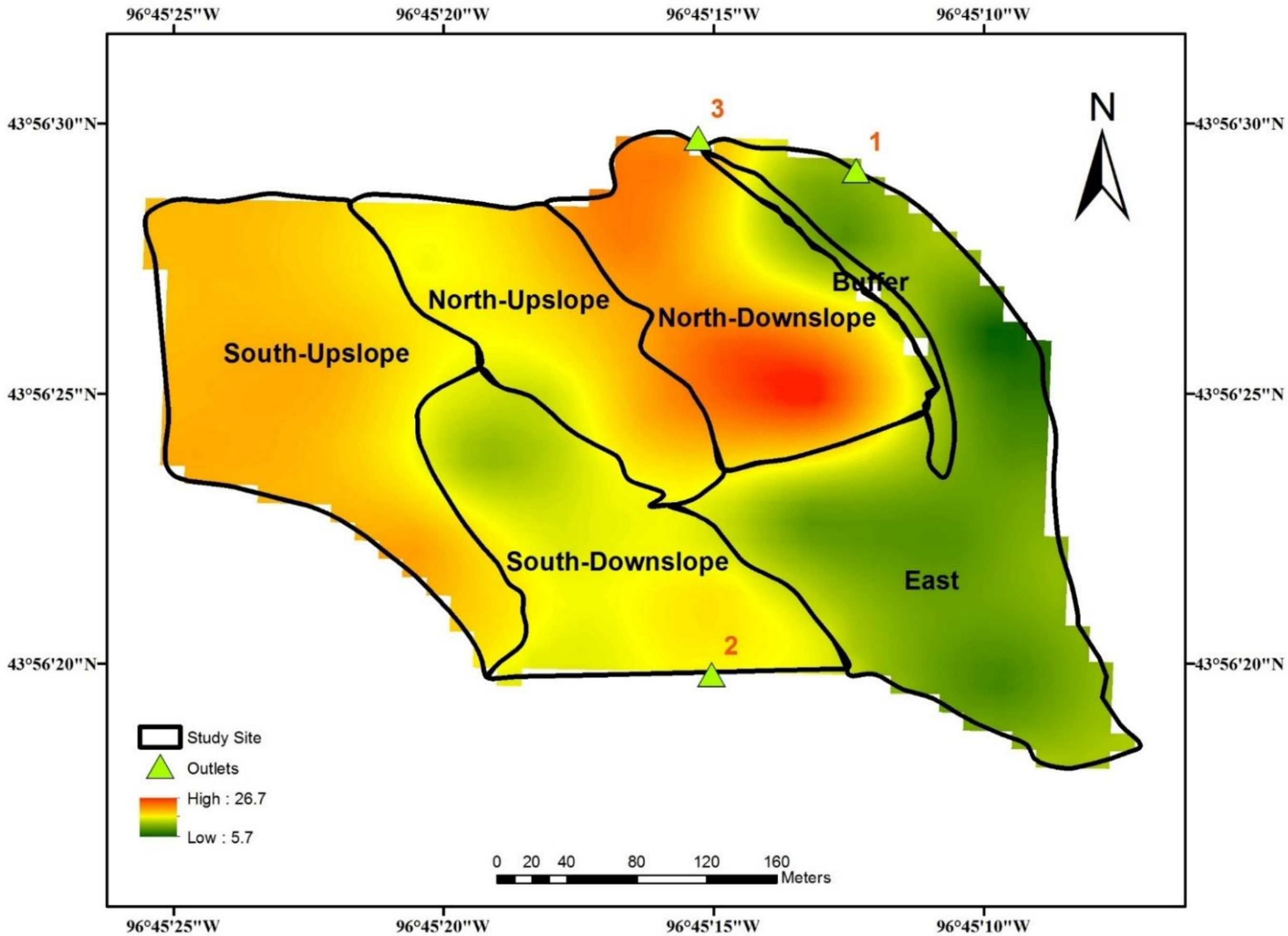

Figure 4.3 Spatial distribution of soil organic carbon across the three watersheds

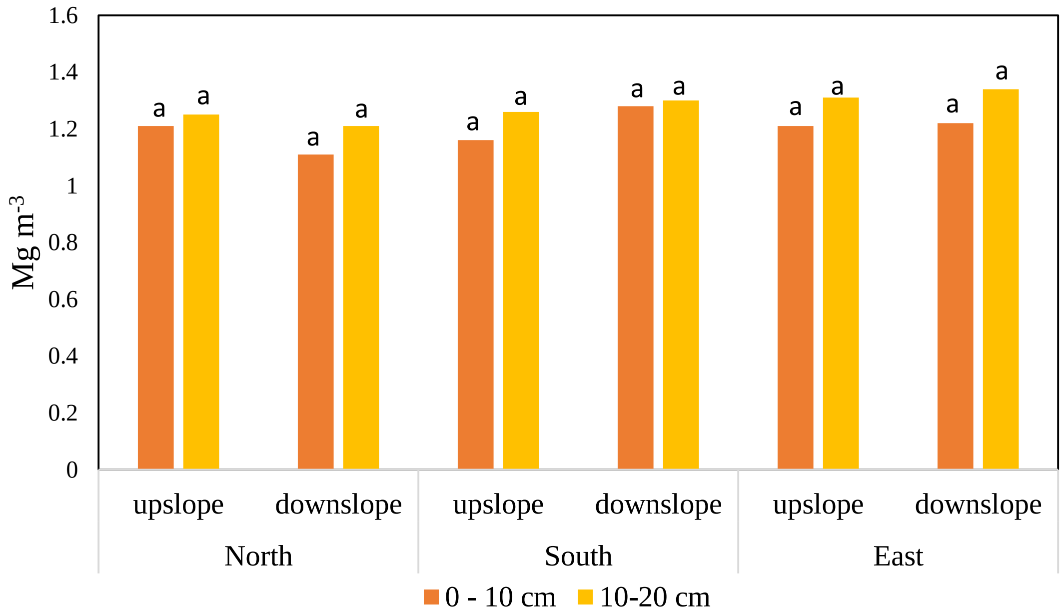

Figure 4.4 Soil bulk density measured at upslope and downslope landscape positions of North, South, and East (Control) Watersheds in 2015.

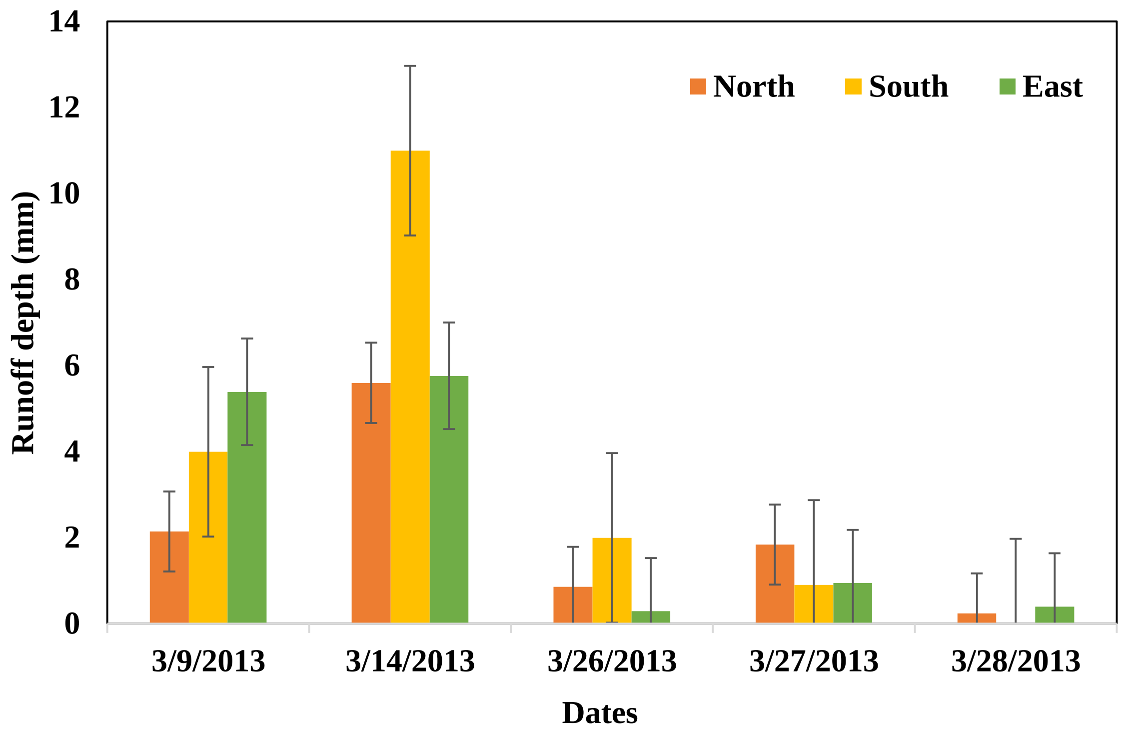

Figure 4.5 Runoff depths (mm) measured from the three watersheds during the storm events in 2013.

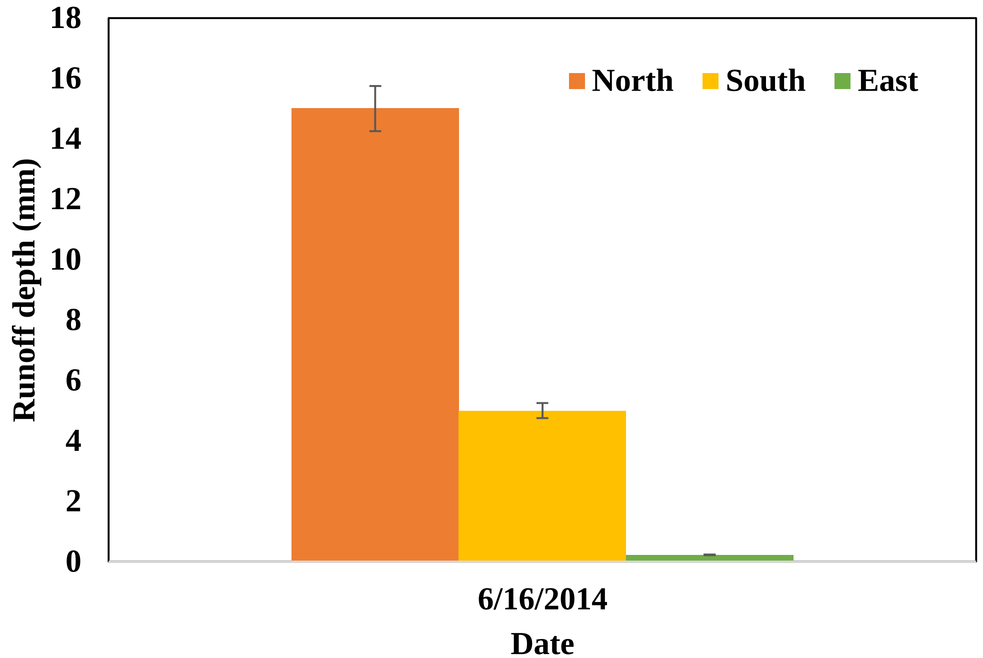

Figure 4.6 Runoff depths (mm) measured from the three watersheds during the storm event in 2014.

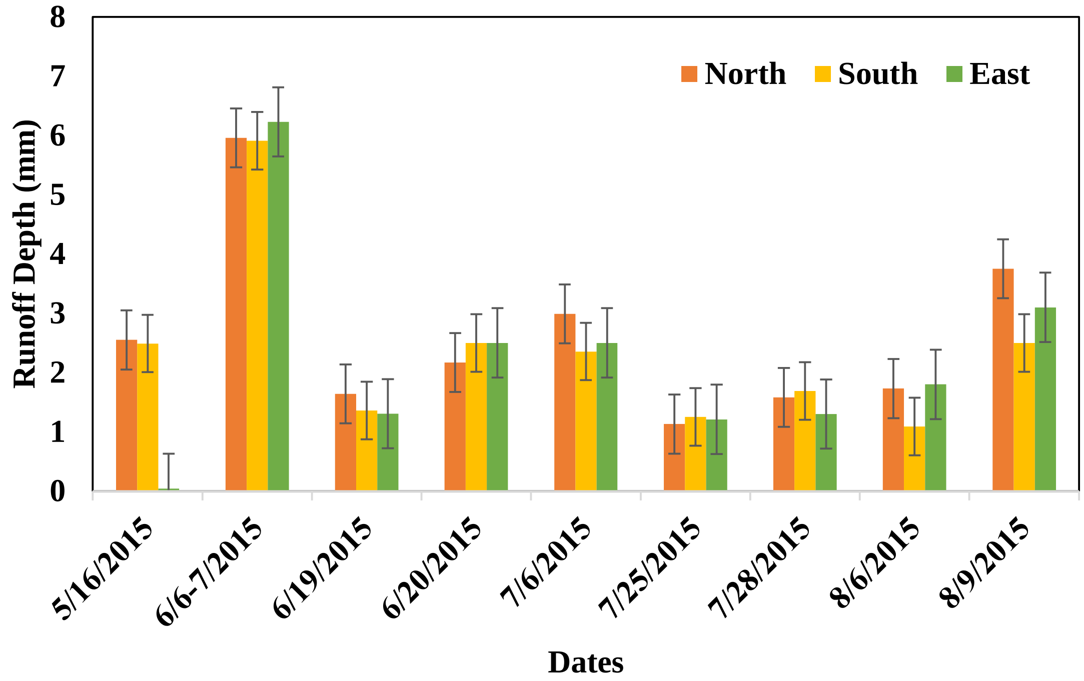

Figure 4.7 Runoff depths (mm) measured from the three watersheds during the storm events in 2015.

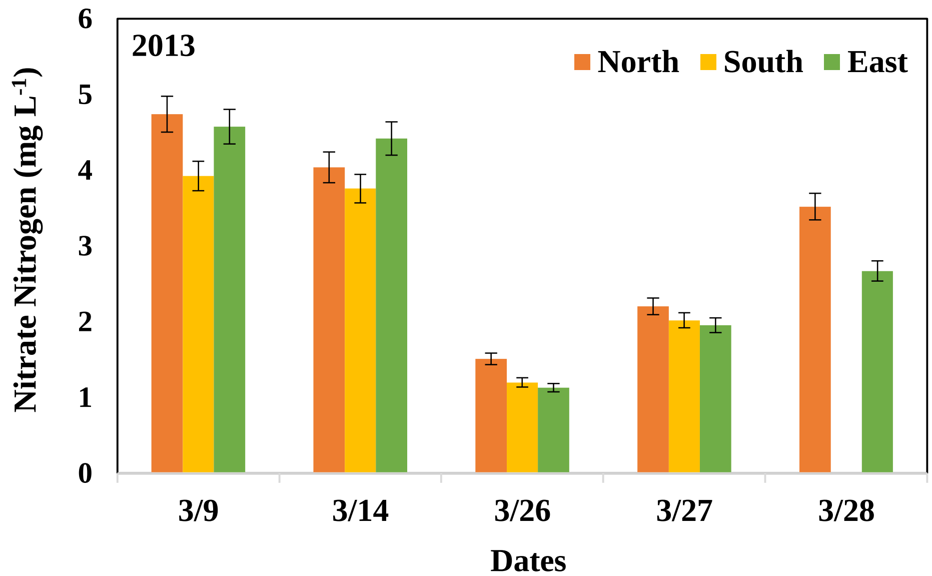

Figure 4.8 Nitrate nitrogen content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2013.

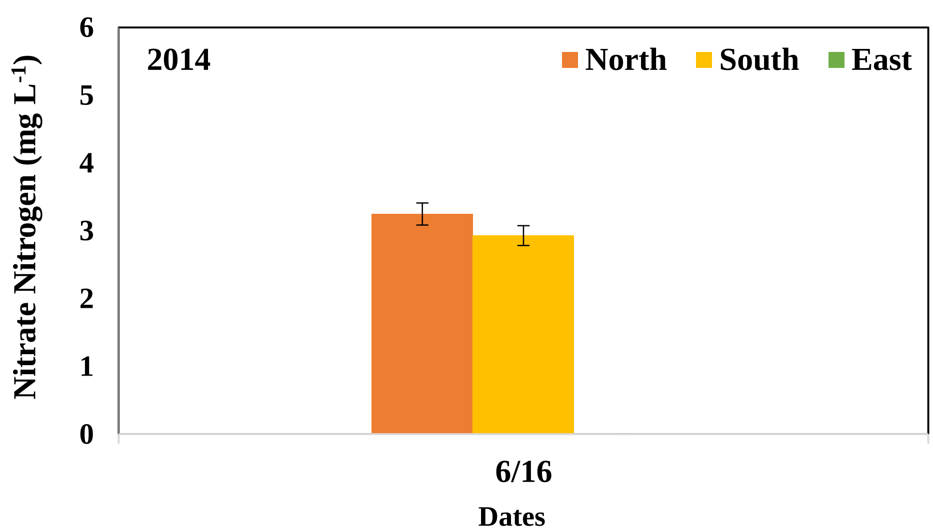

Figure 4.9 Nitrate nitrogen content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2014.

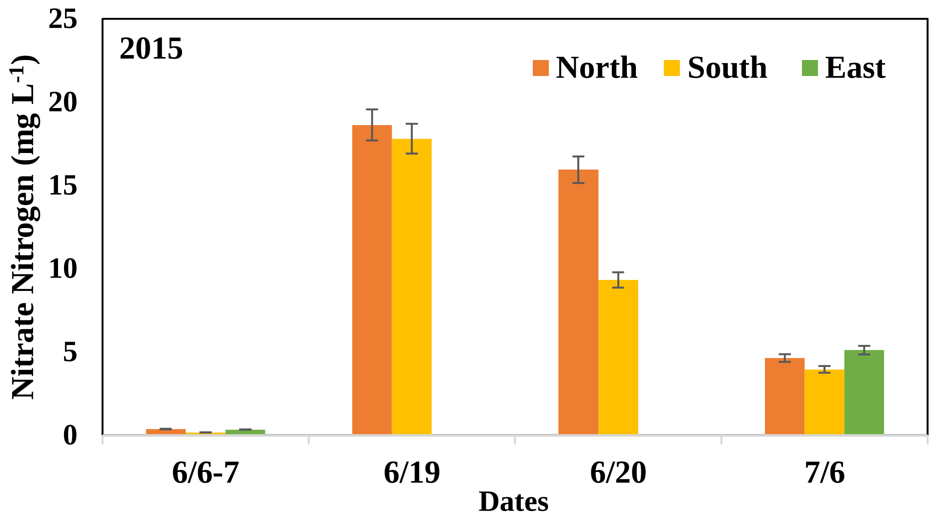

Figure 4.10 Nitrate nitrogen content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2015.

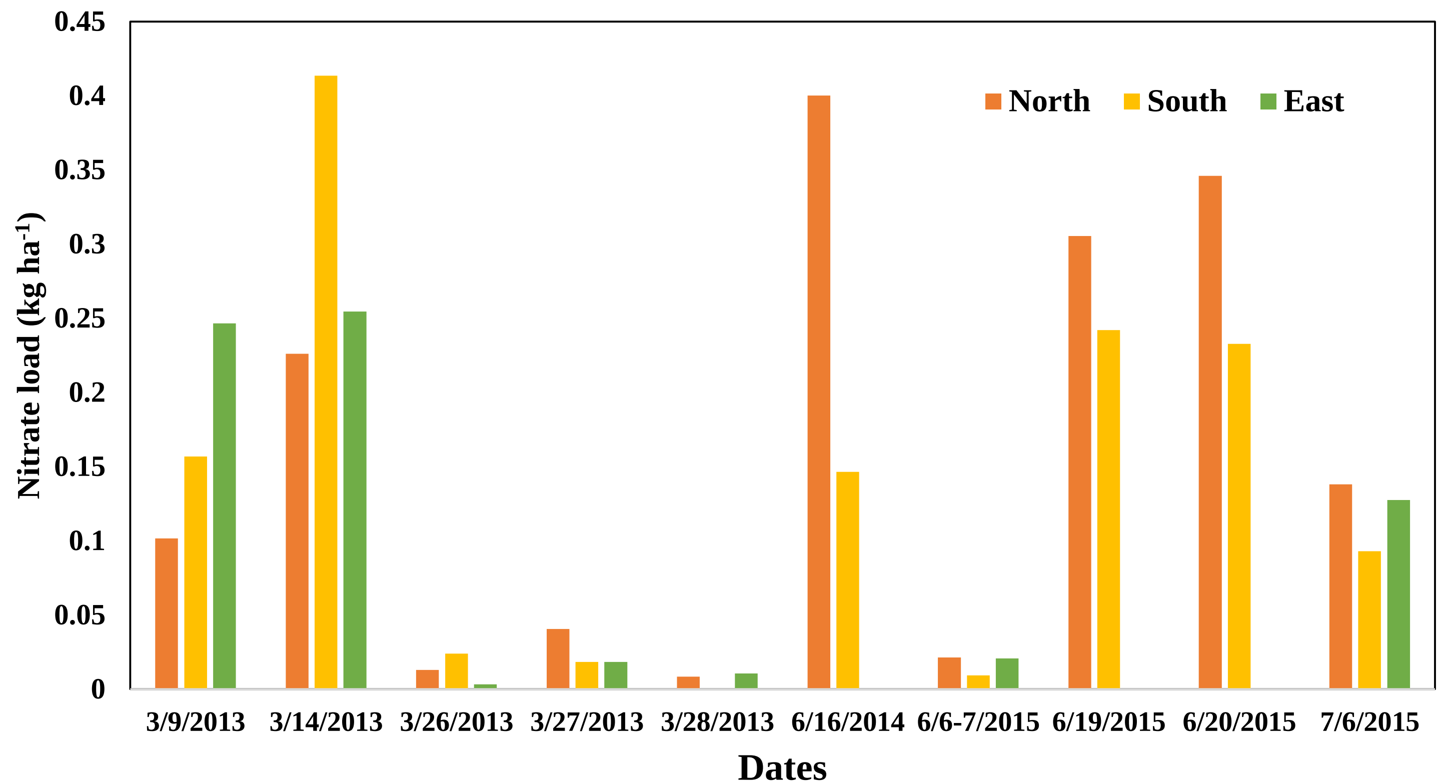

Figure 4.11 Nitrate nitrogen load (kg ha^{-1}) from the surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the years 2013-2015.

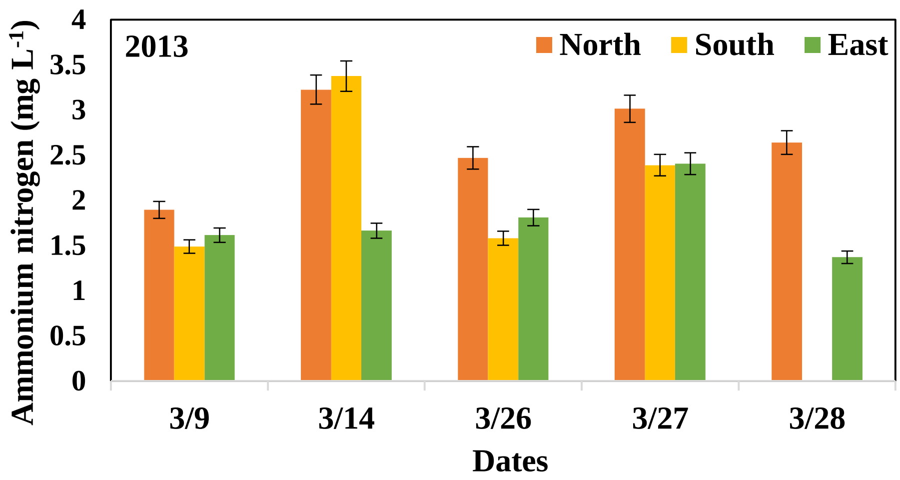

Figure 4.12 Ammonium nitrogen content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2013.

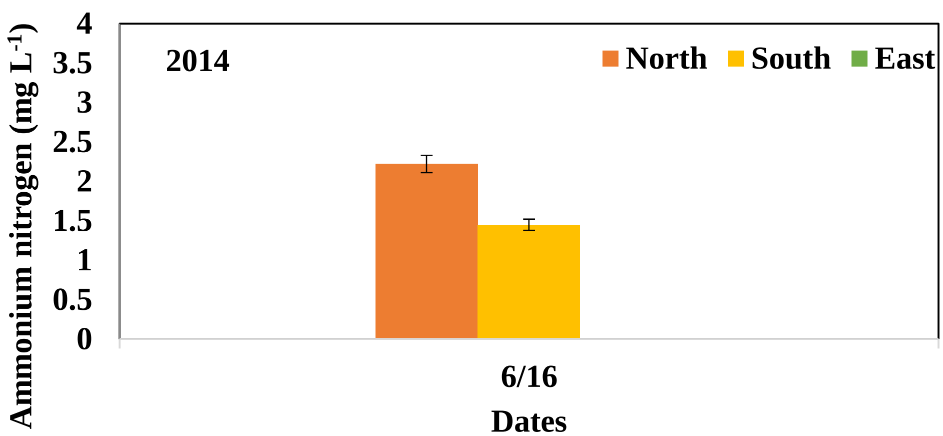

Figure 4.13 Ammonium nitrogen content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2014.

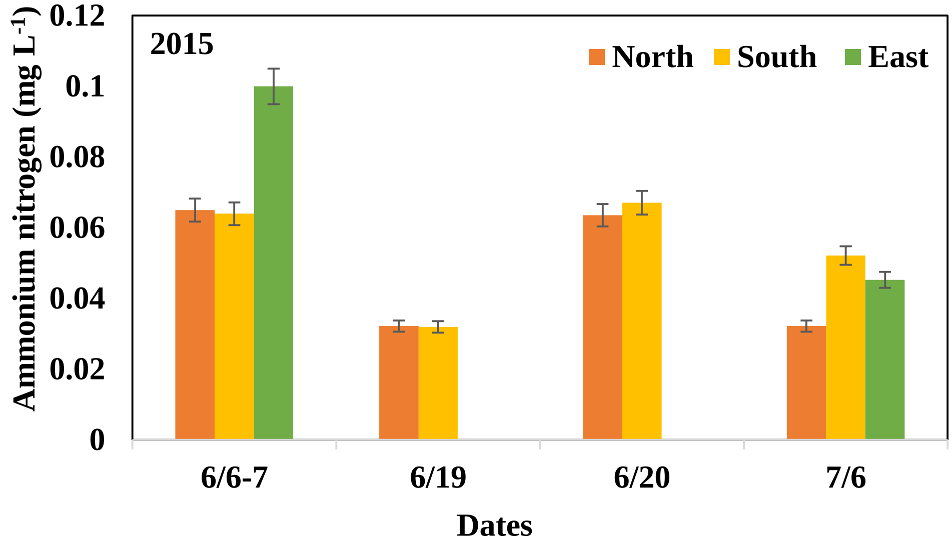

Figure 4.14 Ammonium nitrogen content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the years 2015.

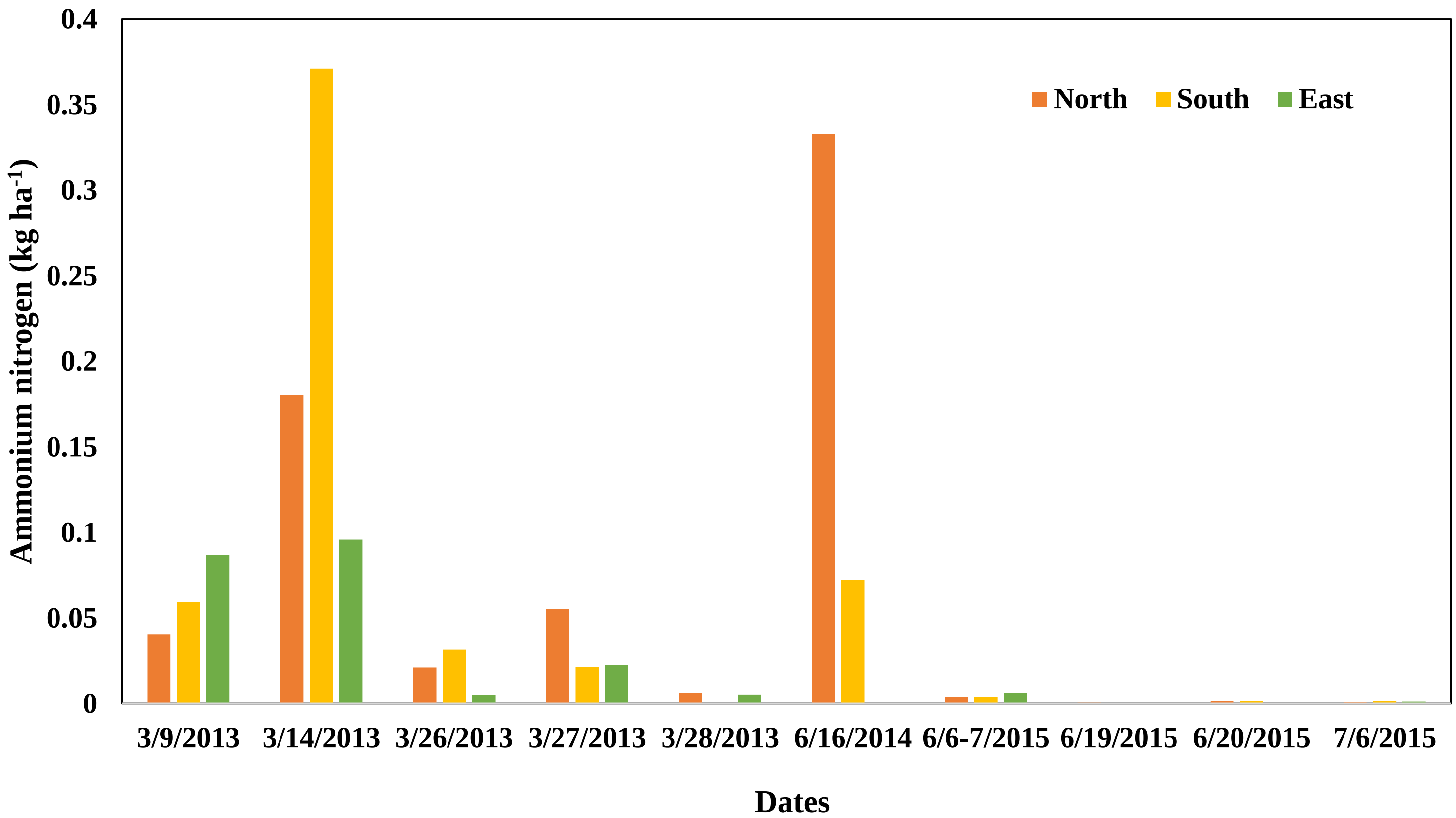

Figure 4.15 Ammonium nitrogen load (kg ha^{-1}) from the surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the years 2013-2015.

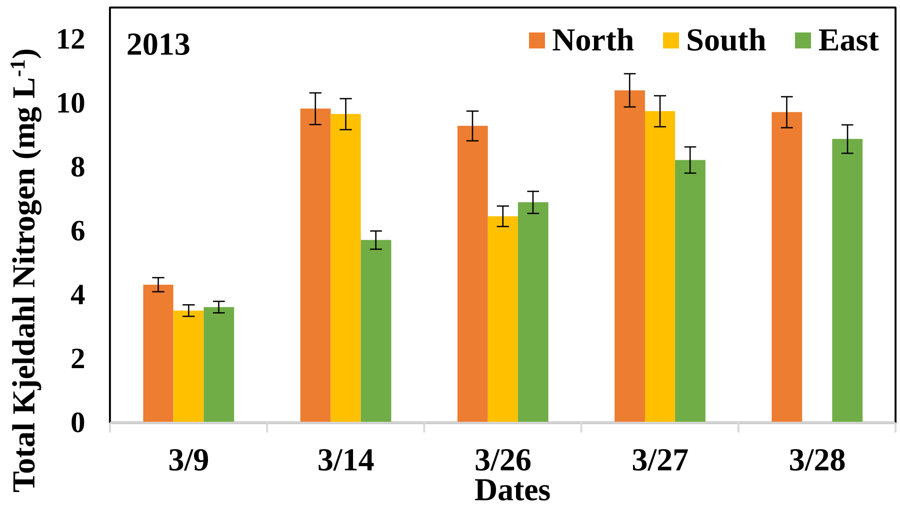

Figure 4.16 Total Kjeldahl nitrogen content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2013.

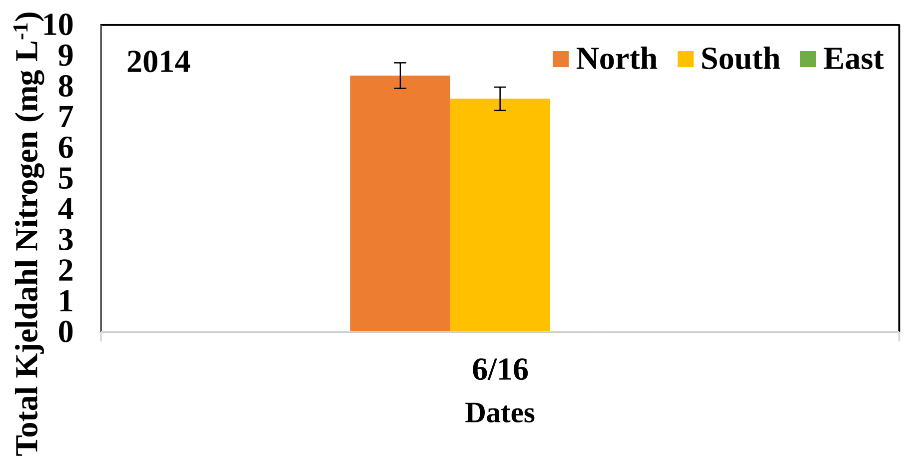

Figure 4.17 Total Kjeldahl nitrogen content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2014.

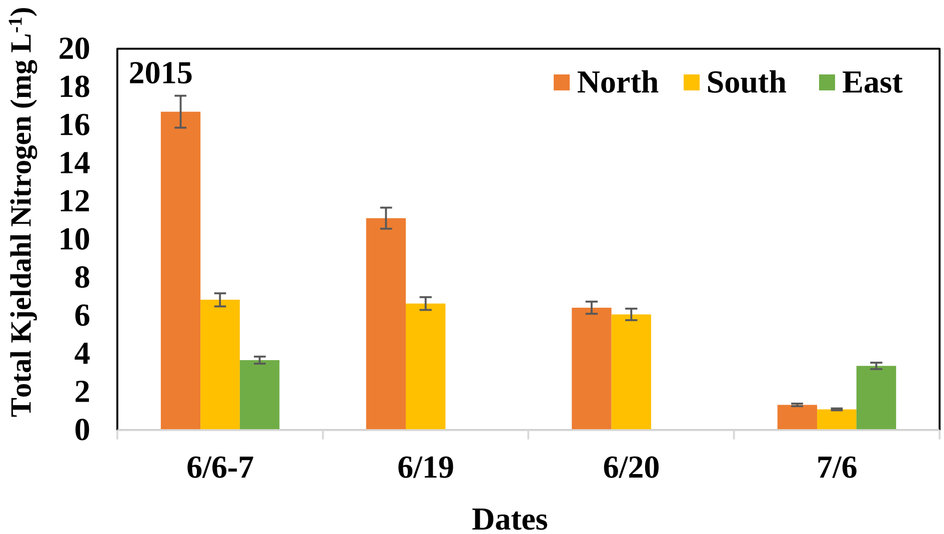

Figure 4.18 Total Kjeldahl nitrogen content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2013.

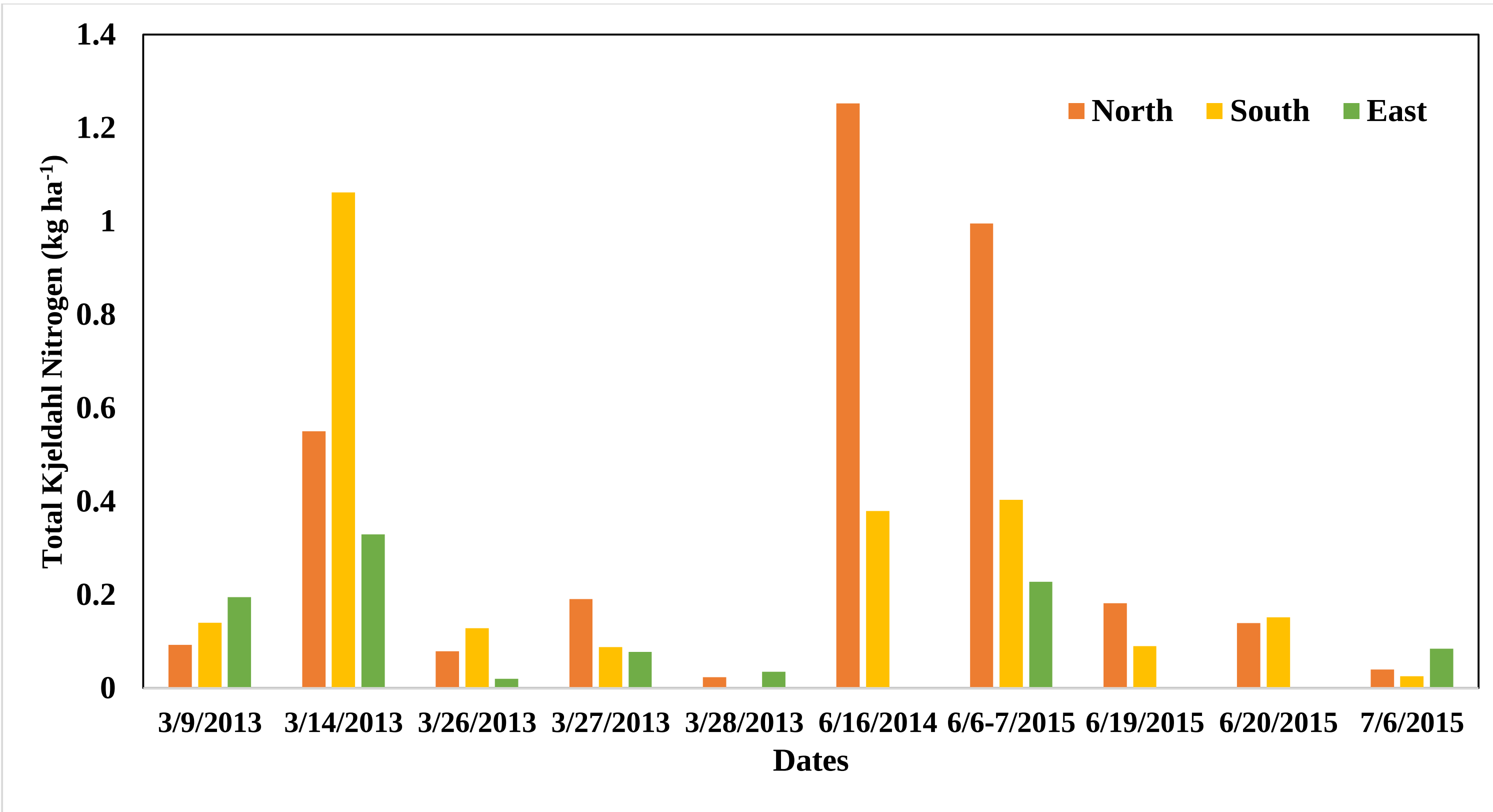

Figure 4.19 Total Kjeldahl nitrogen load (kg ha^{-1}) from the surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the years 2013-2015.

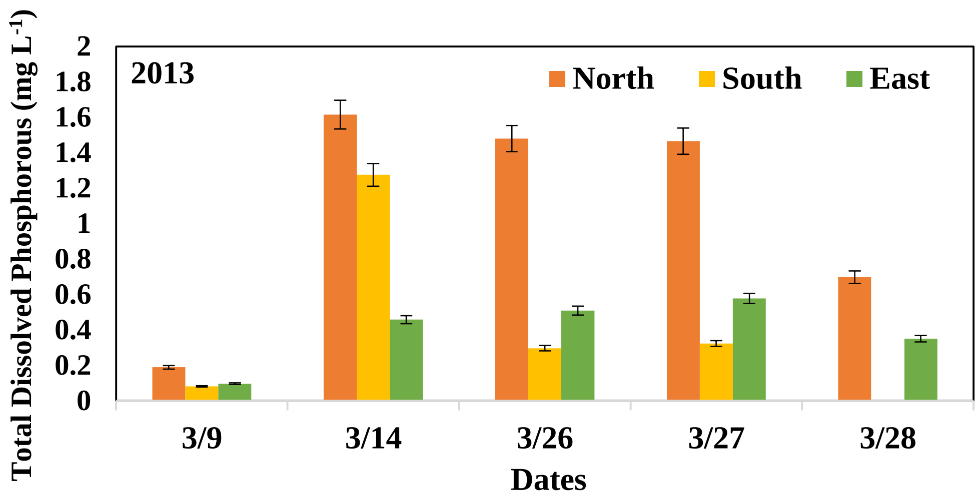

Figure 4.20 Total dissolved phosphorus content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2013.

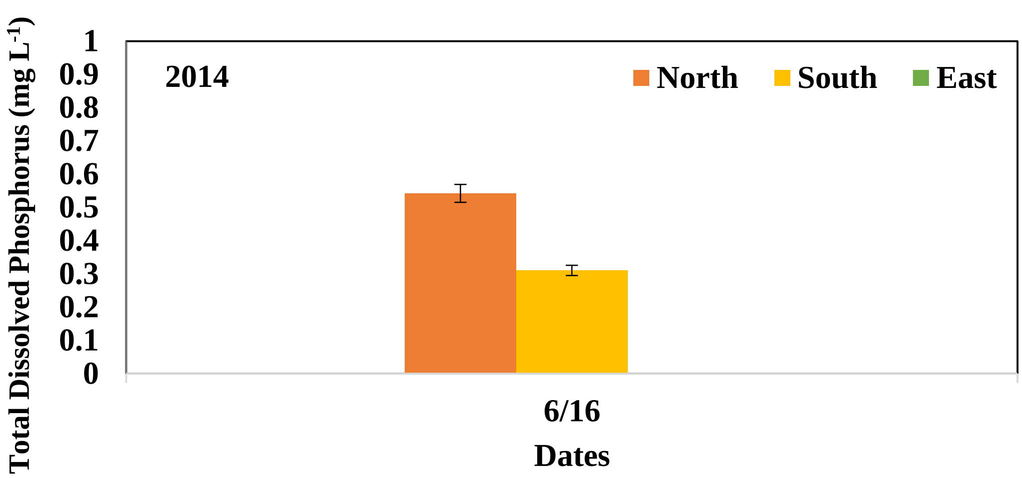

Figure 4.21 Total dissolved phosphorus content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2014.

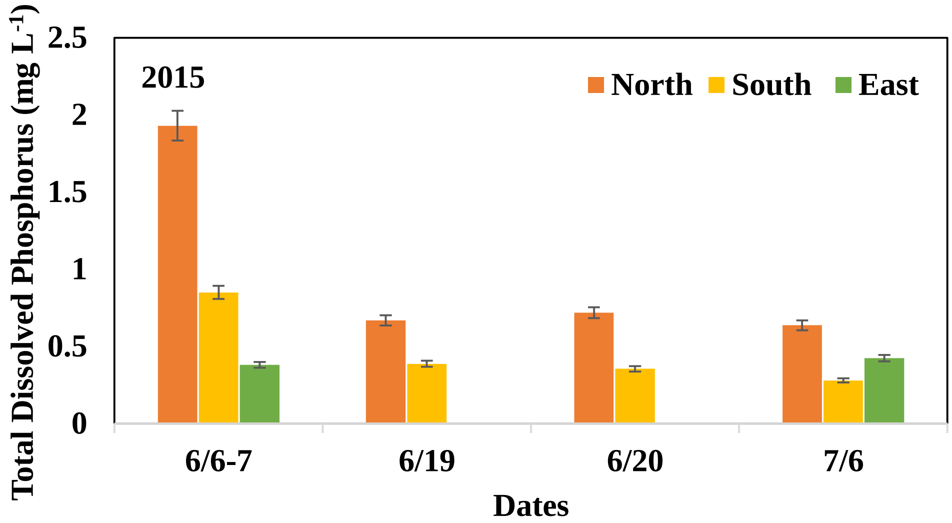

Figure 4.22 Total dissolved phosphorus content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2015.

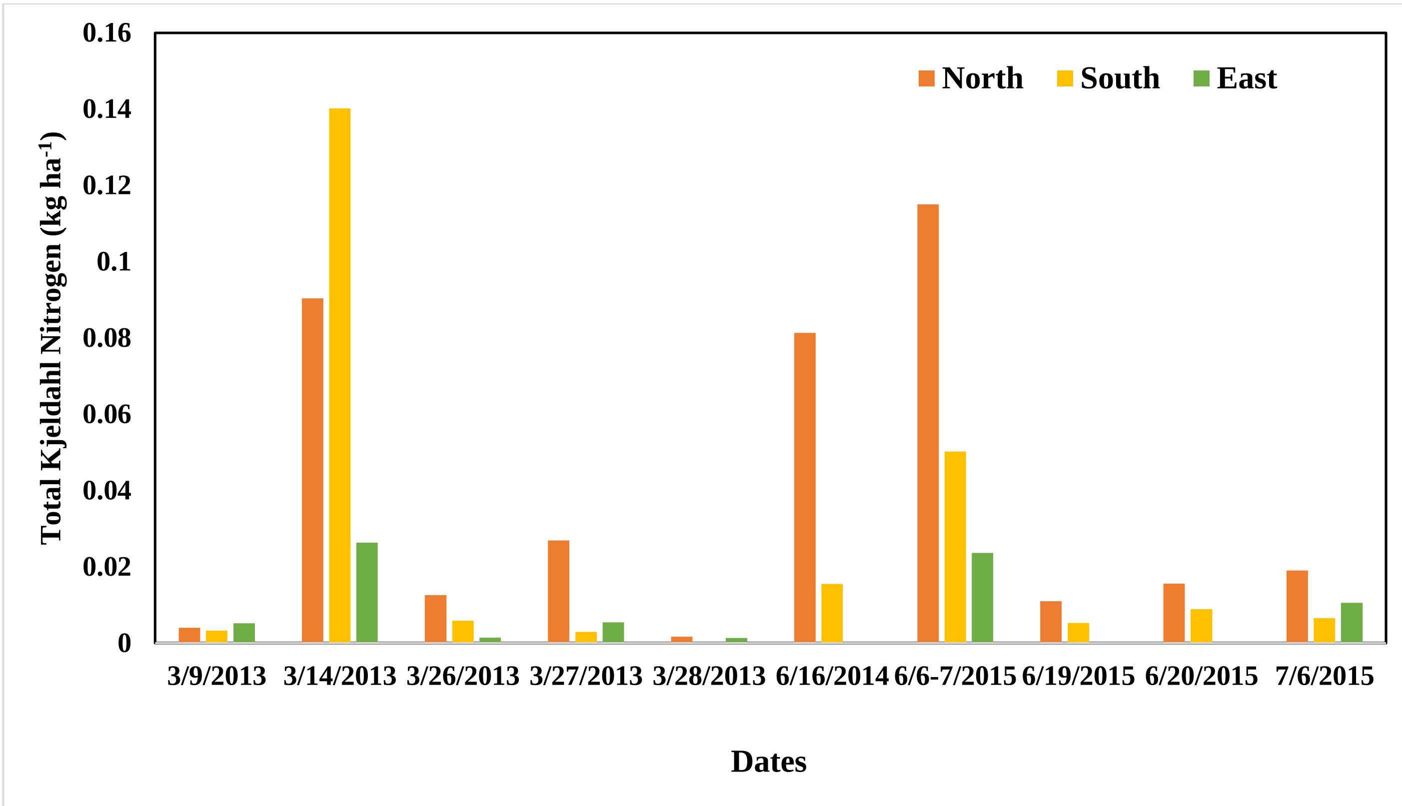

Figure 4.23 Total dissolved phosphorus load (kg ha^{-1}) from the surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the years 2013-2015.

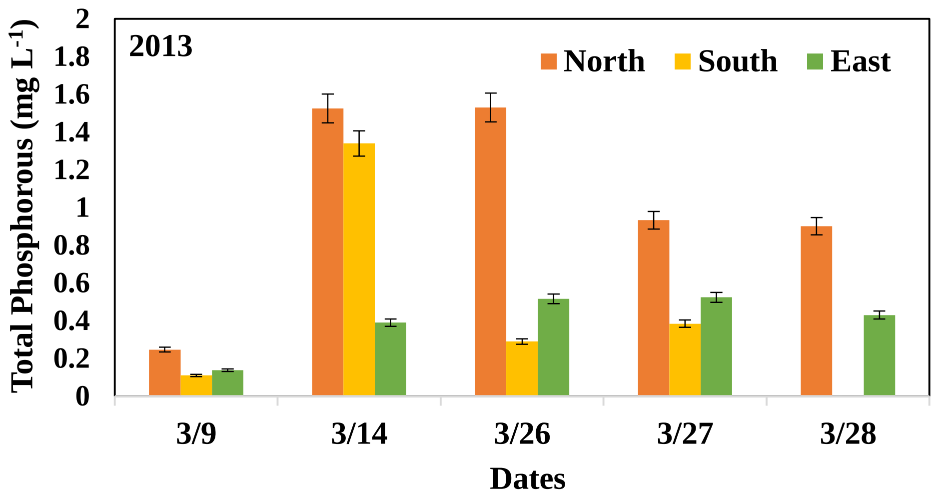

Figure 4.24 Total phosphorus content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2013.

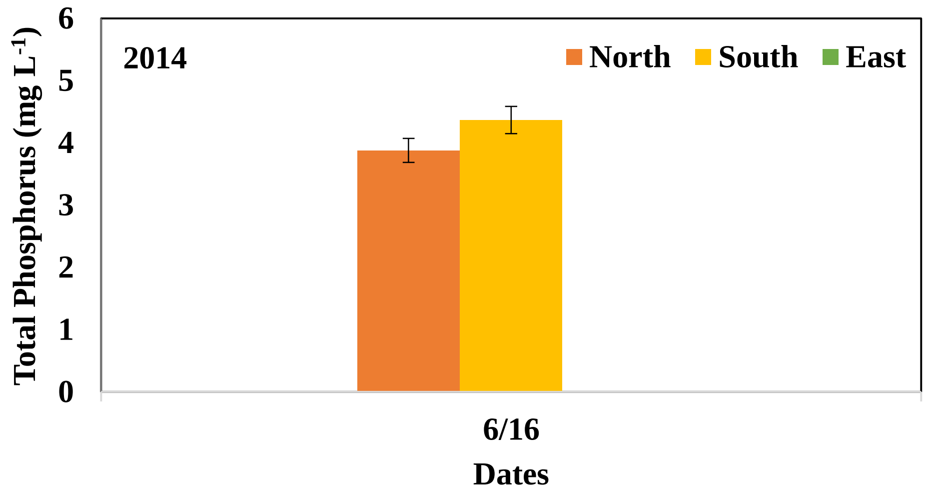

Figure 4.25 Total phosphorus content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2014.

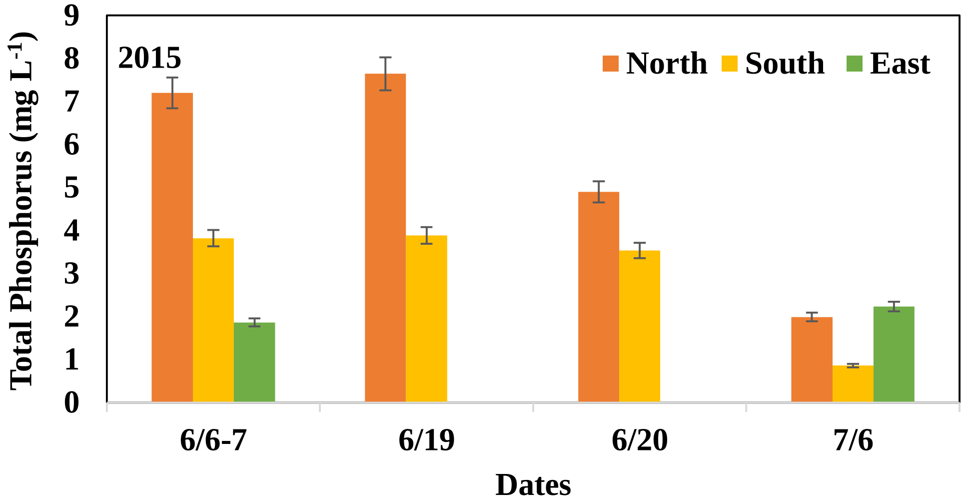

Figure 4.26 Total phosphorus content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2015.

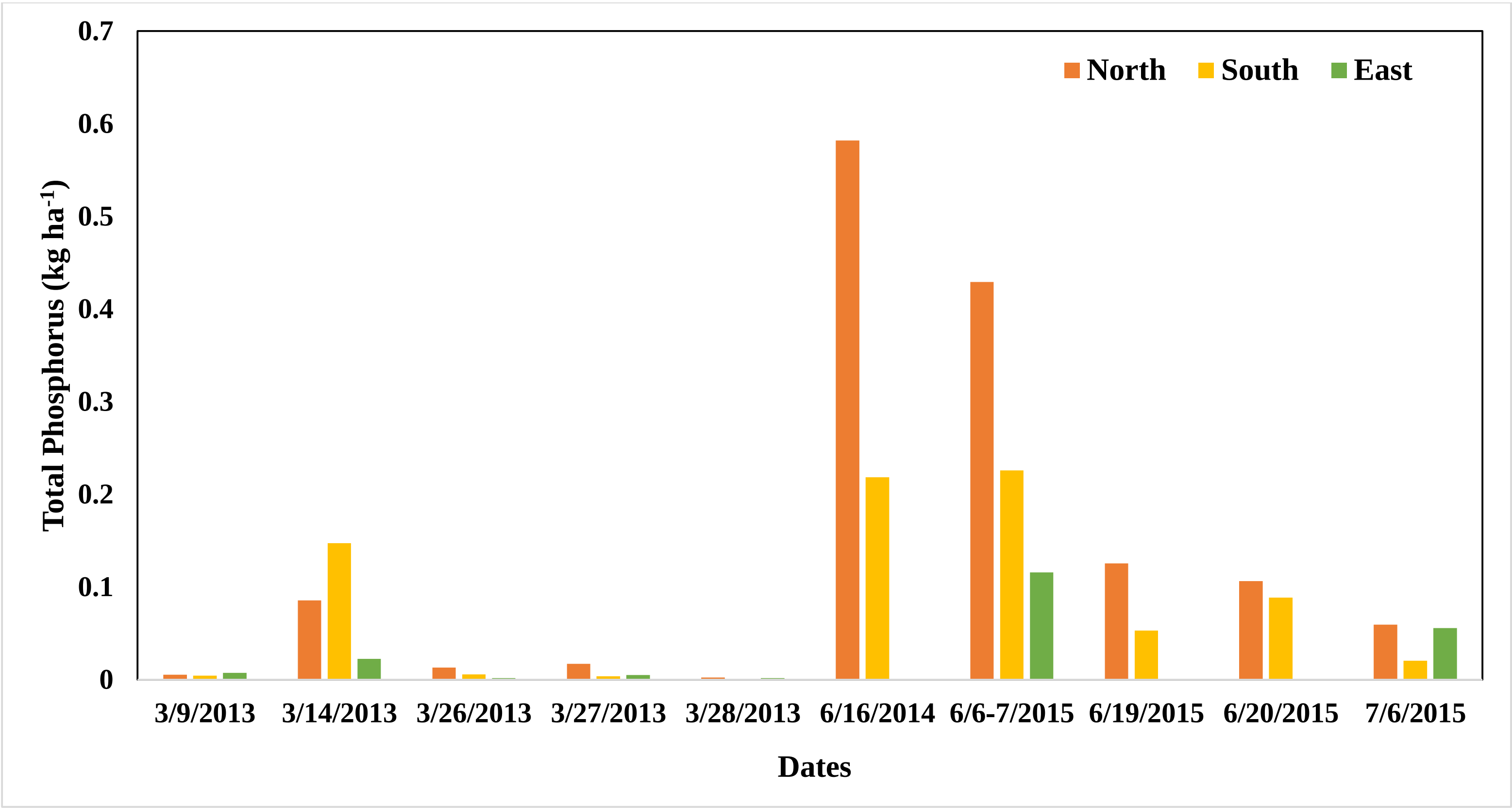

Figure 4.27 Total phosphorus load (kg ha^{-1}) from the surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the years 2013-2015.

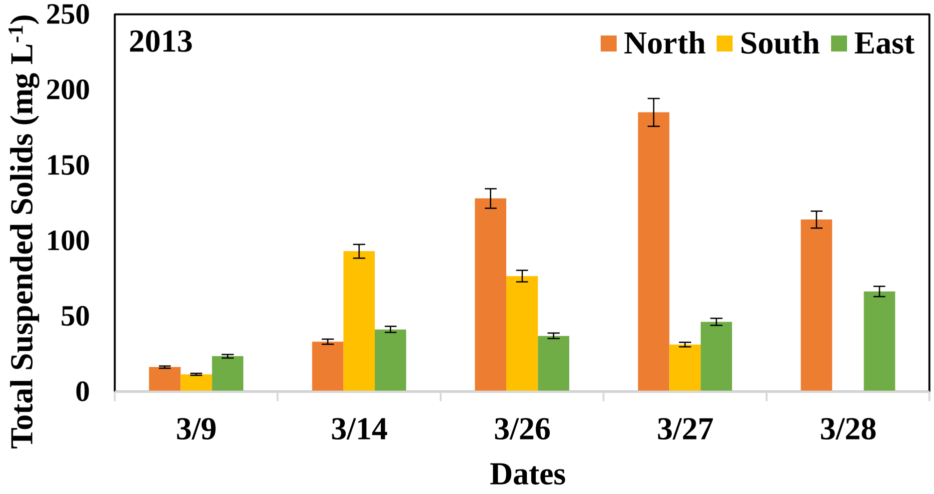

Figure 4.28 Total suspended solids content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2013.

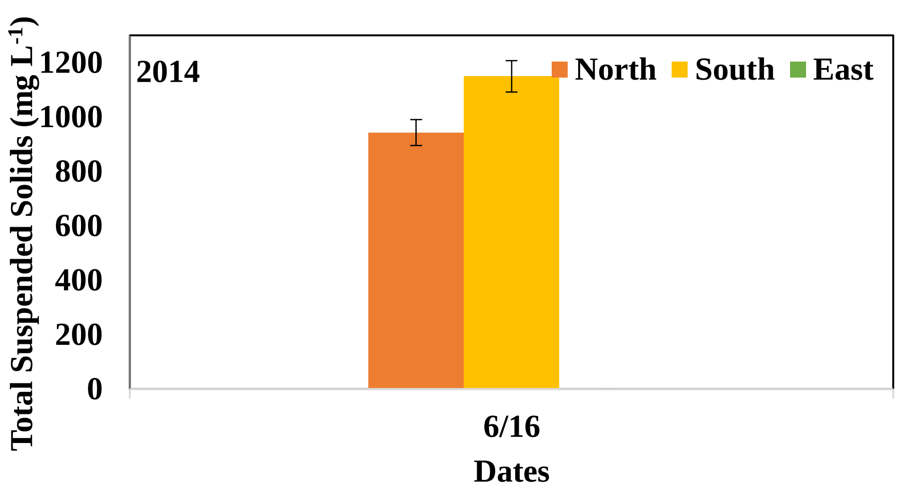

Figure 4.29 Total suspended solids content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2014.

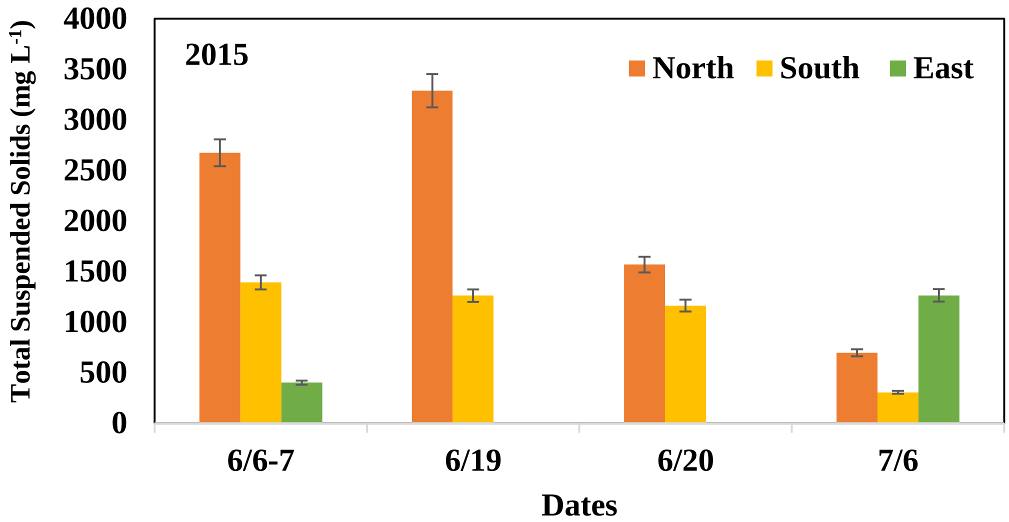

Figure 4.30 Total suspended solids content in surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the year 2015.

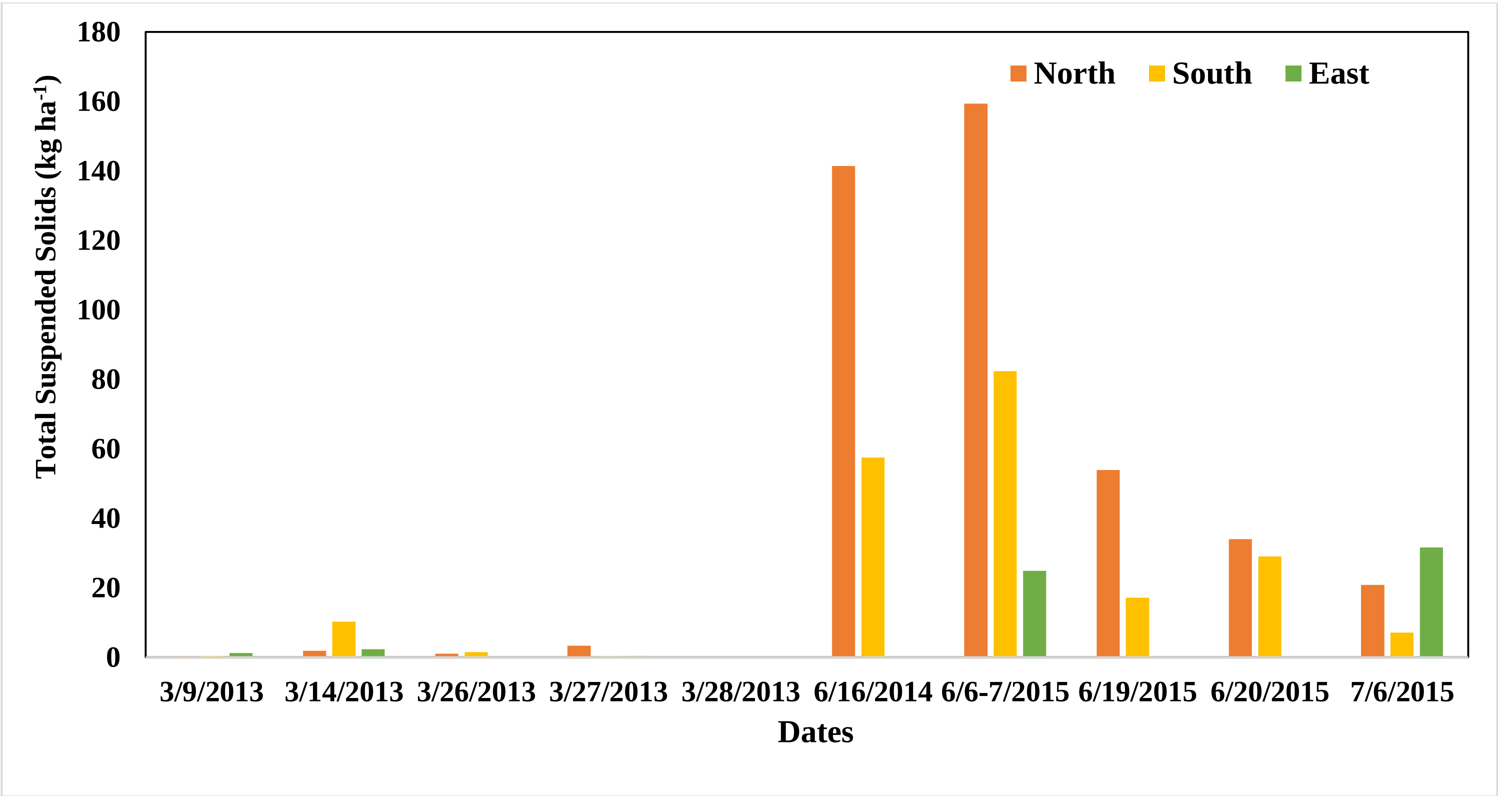

Figure 4.31 Total suspended solids load (kg ha^{-1}) from the surface runoff monitored from small watersheds (north and south) managed with manure (manure applied at upslope in south and at downslope in north watershed) and managed without manure [i.e. control (east)] for the years 2013-2015.

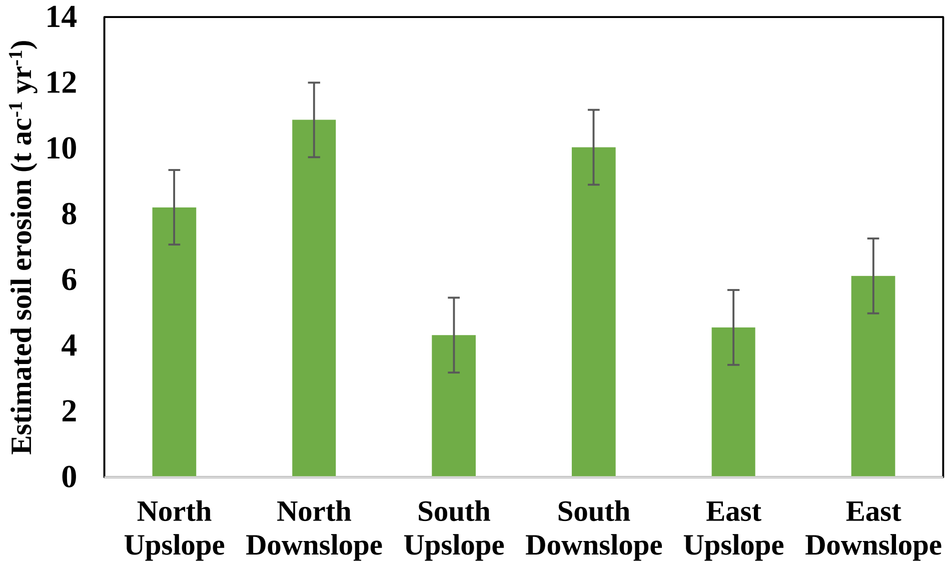

Figure 4.32 The estimated soil loss from the landscape positions of the watersheds.

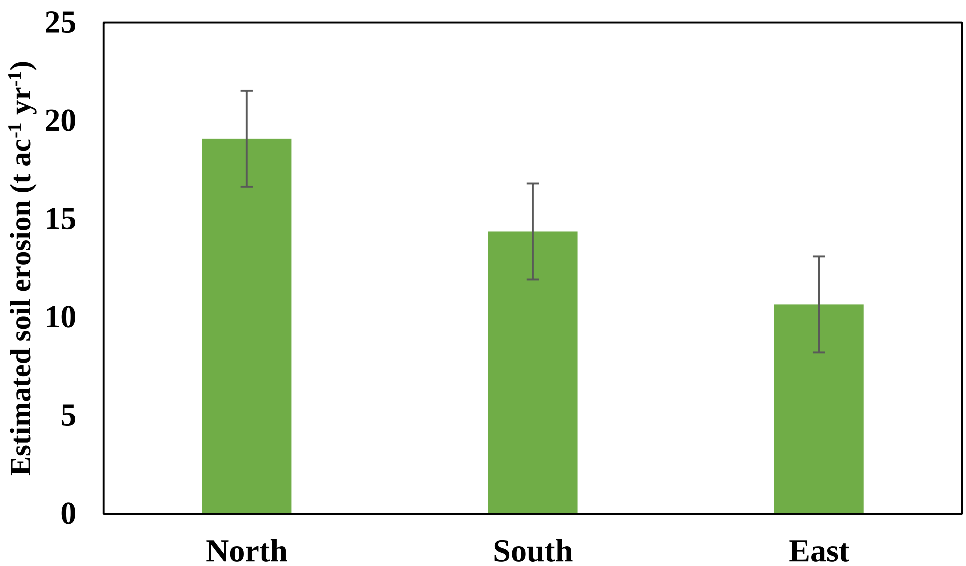

Figure 4.33 The estimated soil erosion from the three watersheds.

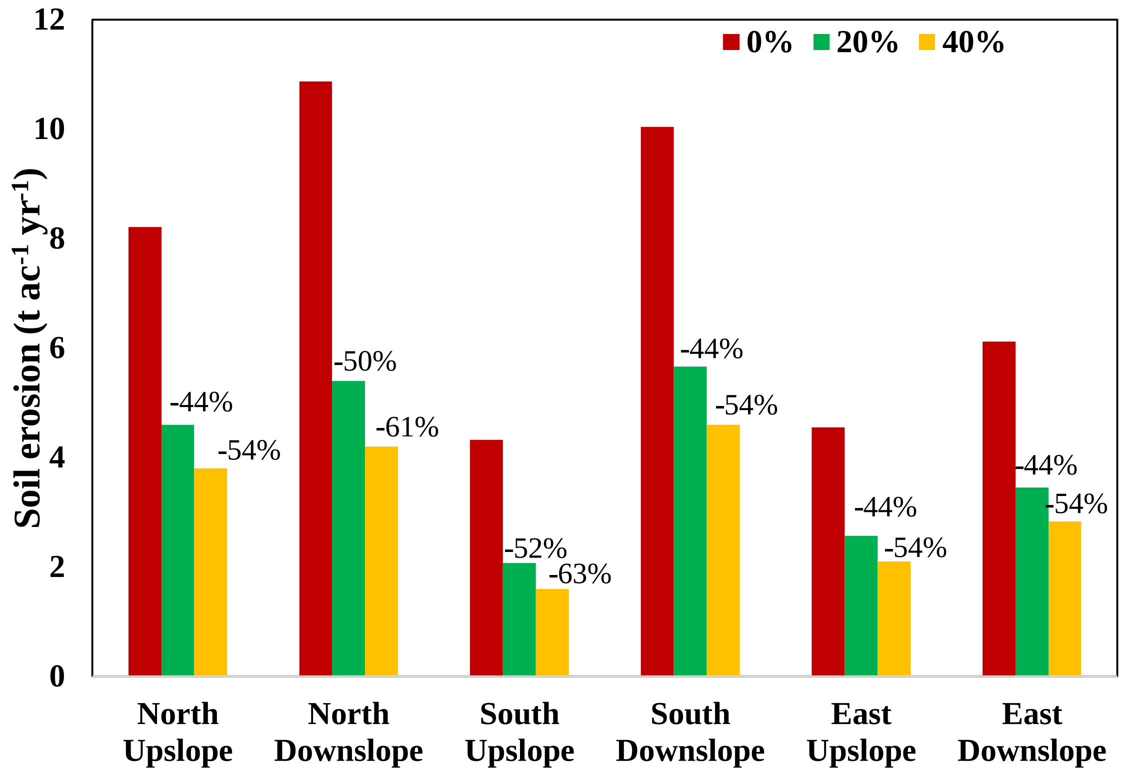

Figure 4.34 The estimated soil erosion by RUSLE after applying an increased residue cover as conservation practice (20% and 40%) on the watersheds

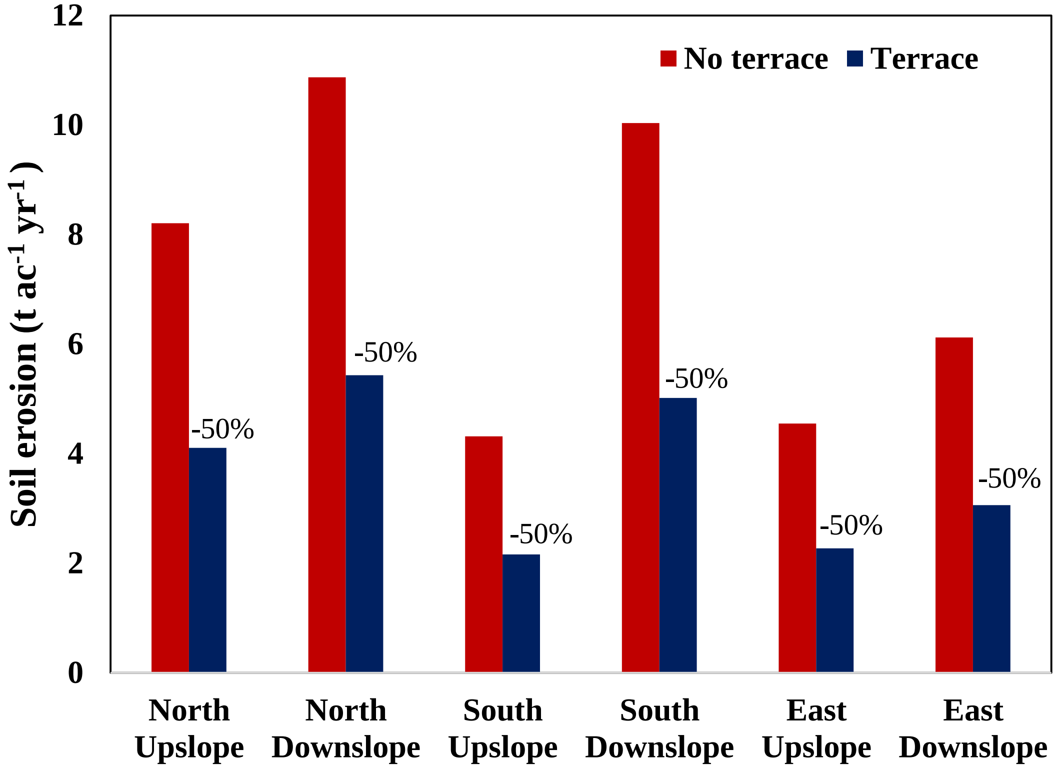

Figure 4.35 The estimated soil erosion by RUSLE after applying terraces as support practice on the watersheds

5. Conclusions

Managing farm waste for longer duration during extreme winters with frozen soils and snow cover is very difficult especially in areas such as South Dakota. A proper implementation of manure management in these areas is strongly encouraged for improving crop productivity, soils quality, and water quantity and quality. Therefore, the present study was conducted in the state of South Dakota with the specific objectives were to assess the manure application on soils and runoff quantity and quality. The three watersheds were studied; two (NW and SW) of them were treated with manure and the third one was the control (CW) watershed with no manure application. The manure was applied at the upper one-half of the SW and lower one-half of the NW. Soil samples were collected during the summer of 2015 for analyzing soil chemical and physical properties, and runoff quantity and quality data was collected from 2013 through 2015.

Results from this study showed that the application of manure, in general, improved selected soil properties such as soil water retention, organic carbon and water infiltration. These improved soil properties due to manure application reduced the runoff from the study watersheds, however, there is no specific trend in the runoff data was observed. The water quality results showed that the water samples from NW had the highest nutrient concentrations as compared to that of SW and the EW. This showed that manure spread on lower terrain (near the outlet) of the NW led to an increased nutrient loss which may increase the risks of eutrophication. It was also observed that the runoff depth from the three watersheds varied greatly and mostly depended on the topography, orientation and slope of the watersheds, and the precipitation occurring throughout the year. However, it cannot be concluded which treatment showed best results in terms of reduced runoff. This was mainly because the precipitation at the study watersheds was highly variable in all the three years. The RUSLE estimated soil erosion was varied with the manure application, residue cover and topography of the watershed. Soil erosion was the highest with no residue left on the ground, and it decreased with the increase in residue cover from 0 to 20 and 40%.

It can be concluded from this study that applying manure on higher terrain (away from the outlet) can reduce nutrient losses into the streams. However, a long-term monitoring of runoff and water quality is needed to assess the impacts of manure at watershed scale.

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Appendix 1

A1.1. pH of soil for 0 – 10 and 10 – 20 cm depths for the six landscape positions with manure and no manure treatment.

Plot ID, the landscape positions; REP, replication; TRT, treatment

0 - 10 cm				10 - 20 cm			
Plot ID	REP	pH	TRT	Plot ID	REP	pH	TRT
North Upslope	1	6.30	No manure	North Upslope	1	6.41	No manure
	2	5.21	No manure		2	5.89	No manure
	3	5.05	No manure		3	6.51	No manure
	4	5.14	No manure		4	7.32	No manure
North Downslope	1	5.89	Manure	North Downslope	1	7.01	Manure
	2	5.71	Manure		2	7.37	Manure
	3	6.78	Manure		3	6.96	Manure
	4	6.96	Manure		4	5.96	Manure
South Upslope	1	4.78	Manure	South Upslope	1	5.35	Manure
	2	5.10	Manure		2	6.35	Manure
	3	5.08	Manure		3	6.30	Manure
	4	5.55	Manure		4	7.06	Manure
South Downslope	1	7.36	No manure	South Downslope	1	7.72	No manure
	2	4.45	No manure		2	4.80	No manure
	3	4.75	No manure		3	5.24	No manure
	4	4.92	No manure		4	5.10	No manure
East Upslope	1	4.65	No manure	East Upslope	1	5.33	No manure
	2	4.85	No manure		2	5.26	No manure
	3	4.68	No manure		3	5.21	No manure
	4	4.73	No manure		4	5.36	No manure
East Downslope	1	4.80	No manure	East Downslope	1	5.23	No manure
	2	4.88	No manure		2	5.38	No manure
	3	4.73	No manure		3	5.24	No manure
	4	4.84	No manure		4	5.28	No manure

A1.2. pH of soil for 20 – 30 and 30 – 40 cm depths for the six landscape positions with manure and no manure treatment.

Plot ID, the landscape positions; REP, replication; TRT, treatment

20 - 30 cm				30 - 40 cm			
Plot ID	REP	pH	TRT	Plot ID	REP	pH	TRT
North Upslope	1	6.55	No manure	North Upslope	1	6.69	No manure
	2	6.24	No manure		2	6.38	No manure
	3	6.65	No manure		3	6.66	No manure
	4	7.29	No manure		4	7.15	No manure
North Downslope	1	7.60	Manure	North Downslope	1	7.85	Manure
	2	7.51	Manure		2	7.82	Manure
	3	7.66	Manure		3	7.63	Manure
	4	7.05	Manure		4	7.69	Manure
South Upslope	1	6.02	Manure	South Upslope	1	7.81	Manure
	2	7.74	Manure		2	7.95	Manure
	3	7.67	Manure		3	7.94	Manure
	4	7.89	Manure		4	7.88	Manure
South Downslope	1	7.87	No manure	South Downslope	1	7.87	No manure
	2	5.85	No manure		2	6.27	No manure
	3	5.74	No manure		3	6.29	No manure
	4	5.37	No manure		4	5.53	No manure
East Upslope	1	7.22	No manure	East Upslope	1	6.12	No manure
	2	6.24	No manure		2	5.91	No manure
	3	5.78	No manure		3	5.85	No manure
	4	5.77	No manure		4	5.94	No manure
East Downslope	1	6.01	No manure	East Downslope	1	7.81	No manure
	2	5.77	No manure		2	7.68	No manure
	3	5.91	No manure		3	6.10	No manure
	4	5.81	No manure		4	5.81	No manure

A1.3. Electrical conductivity ($\mu S\ cm^{-1}$) of soil for 0-10 and 10 -20 cm depths for the six landscape positions with manure and no manure treatment.

Plot ID, the landscape positions; REP, replication; EC, electrical conductivity; TRT, treatment

0 - 10 cm				10 - 20 cm			
Plot ID	REP	EC	TRT	Plot ID	REP	EC	TRT
North Upslope	1	158.1	No manure	North Upslope	1	158.3	No manure
	2	176.0	No manure		2	143.0	No manure
	3	183.1	No manure		3	153.8	No manure
	4	253.1	No manure		4	161.3	No manure
North Downslope	1	288.6	Manure	North Downslope	1	271.9	Manure
	2	223.0	Manure		2	169.5	Manure
	3	232.5	Manure		3	200.2	Manure
	4	147.0	Manure		4	126.0	Manure
South Upslope	1	324.9	Manure	South Upslope	1	95.38	Manure
	2	508.2	Manure		2	351.8	Manure
	3	236.2	Manure		3	291.4	Manure
	4	217.9	Manure		4	179.6	Manure
South Downslope	1	242.0	No manure	South Downslope	1	150.0	No manure
	2	552.7	No manure		2	220.1	No manure
	3	92.94	No manure		3	115.2	No manure
	4	126.3	No manure		4	165.0	No manure
East Upslope	1	90.92	No manure	East Upslope	1	80.00	No manure
	2	130.0	No manure		2	90.60	No manure
	3	141.6	No manure		3	103.3	No manure
	4	136.0	No manure		4	103.6	No manure
East Downslope	1	195.0	No manure	East Downslope	1	183.0	No manure
	2	109.7	No manure		2	120.0	No manure
	3	120.6	No manure		3	88.86	No manure
	4	195.0	No manure		4	141.2	No manure

A1.4. Electrical conductivity ($\mu S\ cm^{-1}$) of soil for 20-30 and 30 -40 cm depths for the six landscape positions with manure and no manure treatment.
Plot ID, the landscape positions; REP, replication; EC, electrical conductivity; TRT, treatment

20 - 30 cm				30 - 40 cm			
Plot ID	REP	EC	TRT	Plot ID	REP	EC	TRT
North Upslope	1	89.25	No manure	North Upslope	1	72.11	No manure
	2	192.7	No manure		2	175.0	No manure
	3	103.5	No manure		3	118.3	No manure
	4	188.0	No manure		4	180.0	No manure
North Downslope	1	208.1	Manure	North Downslope	1	183.0	Manure
	2	192.3	Manure		2	179.3	Manure
	3	192.9	Manure		3	192.2	Manure
	4	149.2	Manure		4	176.8	Manure
South Upslope	1	248.2	Manure	South Upslope	1	61.62	Manure
	2	125.4	Manure		2	315.3	Manure
	3	292.1	Manure		3	292.8	Manure
	4	150.1	Manure		4	33.27	Manure
South Downslope	1	263.0	No manure	South Downslope	1	183.0	No manure
	2	103.0	No manure		2	129.7	No manure
	3	118.7	No manure		3	127.9	No manure
	4	88.96	No manure		4	69.37	No manure
East Upslope	1	69.28	No manure	East Upslope	1	85.61	No manure
	2	91.06	No manure		2	101.0	No manure
	3	87.17	No manure		3	83.54	No manure
	4	121.2	No manure		4	96.41	No manure
East Downslope	1	150.0	No manure	East Downslope	1	130.0	No manure
	2	130.0	No manure		2	163.1	No manure
	3	96.21	No manure		3	85.26	No manure
	4	150.0	No manure		4	140.0	No manure

A1.5. Soil bulk density (Mg m^{-3}) for 0-10 and 10-20 cm depths for the six landscape positions with manure and no manure treatment.
Plot ID, the landscape positions; REP, replication; BD, bulk density; TRT, treatment

0 - 10 cm				10 - 20 cm			
Plot ID	REP	BD	TRT	Plot ID	REP	BD	TRT
North Upslope	1	1.21	No manure	North Upslope	1	1.15	No manure
	2	1.22	No manure		2	1.25	No manure
	3	1.21	No manure		3	1.32	No manure
	4	1.19	No manure		4	1.30	No manure
North Downslope	1	1.16	Manure	North Downslope	1	1.21	Manure
	2	1.01	Manure		2	1.19	Manure
	3	1.11	Manure		3	1.24	Manure
	4	1.20	Manure		4	1.23	Manure
South Upslope	1	1.21	Manure	South Upslope	1	1.37	Manure
	2	1.02	Manure		2	1.15	Manure
	3	1.13	Manure		3	1.15	Manure
	4	1.31	Manure		4	1.38	Manure
South Downslope	1	1.29	No manure	South Downslope	1	1.17	No manure
	2	1.34	No manure		2	1.29	No manure
	3	1.16	No manure		3	1.37	No manure
	4	1.35	No manure		4	1.37	No manure
East Upslope	1	1.20	No manure	East Upslope	1	1.30	No manure
	2	1.20	No manure		2	1.36	No manure
	3	1.18	No manure		3	1.24	No manure
	4	1.25	No manure		4	1.35	No manure
East Downslope	1	1.14	No manure	East Downslope	1	1.22	No manure
	2	1.19	No manure		2	1.41	No manure
	3	1.28	No manure		3	1.45	No manure
	4	1.30	No manure		4	1.28	No manure

A1.6. Soil organic carbon (g kg-1) for 0-10 and 10-20 cm depths for the six landscape positions with manure and no manure treatment.
Plot ID, the landscape positions; REP, replication; SOC, soil organic carbon; TRT, treatment

0 - 10 cm				10 - 20 cm			
Plot ID	REP	SOC	TRT	Plot ID	REP	SOC	TRT
North Upslope	1	20.30	No manure	North Upslope	1	7.20	No manure
	2	14.50	No manure		2	5.89	No manure
	3	15.60	No manure		3	10.0	No manure
	4	17.96	No manure		4	4.30	No manure
North Downslope	1	27.60	Manure	North Downslope	1	12.7	Manure
	2	24.30	Manure		2	9.57	Manure
	3	22.80	Manure		3	11.8	Manure
	4	22.90	Manure		4	14.2	Manure
South Upslope	1	20.00	Manure	South Upslope	1	14.8	Manure
	2	19.50	Manure		2	14.8	Manure
	3	20.50	Manure		3	14.6	Manure
	4	21.00	Manure		4	17.5	Manure
South Downslope	1	17.40	No manure	South Downslope	1	14.7	No manure
	2	15.20	No manure		2	12.1	No manure
	3	11.80	No manure		3	14.8	No manure
	4	14.80	No manure		4	16.9	No manure
East Upslope	1	8.80	No manure	East Upslope	1	4.30	No manure
	2	9.30	No manure		2	5.66	No manure
	3	8.30	No manure		3	3.49	No manure
	4	9.70	No manure		4	3.00	No manure
East Downslope	1	11.10	No manure	East Downslope	1	7.39	No manure
	2	12.60	No manure		2	3.99	No manure
	3	9.25	No manure		3	6.66	No manure
	4	5.10	No manure		4	5.87	No manure

A1.7. Soil organic carbon (g kg^{-1}) for 20-30 and 30-40 cm depths for the six landscape positions with manure and no manure treatment.
Plot ID, the landscape positions; REP, replication; SOC, soil organic carbon; TRT, treatment

20 - 30 cm				30 - 40 cm			
Plot ID	REP	SOC	TRT	Plot ID	REP	SOC	TRT
North Upslope	1	3.69	No manure	North Upslope	1	0.11	No manure
	2	2.94	No manure		2	0.78	No manure
	3	1.28	No manure		3	0.78	No manure
	4	2.13	No manure		4	0.38	No manure
North Downslope	1	6.19	Manure	North Downslope	1	1.11	Manure
	2	5.13	Manure		2	0.83	Manure
	3	6.61	Manure		3	0.84	Manure
	4	4.6	Manure		4	0.87	Manure
South Upslope	1	10.1	Manure	South Upslope	1	4.5	Manure
	2	10.3	Manure		2	6	Manure
	3	9.6	Manure		3	6.66	Manure
	4	8.4	Manure		4	5.3	Manure
South Downslope	1	5.56	No manure	South Downslope	1	4.69	No manure
	2	5.7	No manure		2	5.38	No manure
	3	6.51	No manure		3	4.71	No manure
	4	7.6	No manure		4	5.3	No manure
East Upslope	1	3.87	No manure	East Upslope	1	1.4	No manure
	2	2.3	No manure		2	1.6	No manure
	3	1.57	No manure		3	1.2	No manure
	4	2.32	No manure		4	1.3	No manure
East Downslope	1	1.07	No manure	East Downslope	1	1.8	No manure
	2	3.2	No manure		2	1.6	No manure
	3	1.9	No manure		3	1.76	No manure
	4	2.7	No manure		4	1.1	No manure

A1.8. Total nitrogen (g kg^{-1}) in soil for 0-10 and 10-20 cm depths for the six landscape positions with manure and no manure treatment.

Plot ID, the landscape positions; REP, replication; TN, total nitrogen; TRT, treatment

0 - 10 cm				10 - 20 cm			
Plot ID	REP	TN	TRT	Plot ID	REP	TN	TRT
North Upslope	1	1.90	No manure	North Upslope	1	1.60	No manure
	2	1.90	No manure		2	0.80	No manure
	3	2.00	No manure		3	1.20	No manure
	4	2.20	No manure		4	1.80	No manure
North Downslope	1	2.50	Manure	North Downslope	1	1.60	Manure
	2	2.10	Manure		2	1.59	Manure
	3	2.30	Manure		3	1.48	Manure
	4	2.50	Manure		4	1.20	Manure
South Upslope	1	2.50	Manure	South Upslope	1	1.98	Manure
	2	2.70	Manure		2	1.81	Manure
	3	2.20	Manure		3	1.63	Manure
	4	2.50	Manure		4	1.86	Manure
South Downslope	1	2.00	No manure	South Downslope	1	1.40	No manure
	2	2.00	No manure		2	1.59	No manure
	3	1.90	No manure		3	1.52	No manure
	4	2.20	No manure		4	2.07	No manure
East Upslope	1	1.80	No manure	East Upslope	1	1.12	No manure
	2	1.60	No manure		2	1.03	No manure
	3	1.70	No manure		3	0.76	No manure
	4	1.80	No manure		4	1.01	No manure
East Downslope	1	1.50	No manure	East Downslope	1	1.40	No manure
	2	1.90	No manure		2	1.56	No manure
	3	1.70	No manure		3	1.46	No manure
	4	1.80	No manure		4	0.93	No manure

A1.9. Total nitrogen (g kg^{-1}) in soil for 20-30 and 30-40 cm depths for the six landscape positions with manure and no manure treatment.

Plot ID, the landscape positions; REP, replication; TN, total nitrogen; TRT, treatment

20 - 30 cm				30 - 40 cm			
Plot ID	REP	TN	TRT	Plot ID	REP	TN	TRT
North Upslope	1	0.88	No manure	North Upslope	1	0.40	No manure
	2	0.70	No manure		2	0.50	No manure
	3	0.66	No manure		3	0.60	No manure
	4	0.70	No manure		4	0.55	No manure
North Downslope	1	1.63	Manure	North Downslope	1	0.65	Manure
	2	1.59	Manure		2	0.87	Manure
	3	1.48	Manure		3	0.77	Manure
	4	1.19	Manure		4	0.48	Manure
South Upslope	1	0.94	Manure	South Upslope	1	0.65	Manure
	2	0.76	Manure		2	0.89	Manure
	3	0.71	Manure		3	0.44	Manure
	4	1.20	Manure		4	0.89	Manure
South Downslope	1	0.66	No manure	South Downslope	1	0.42	No manure
	2	1.17	No manure		2	0.51	No manure
	3	0.66	No manure		3	0.57	No manure
	4	1.20	No manure		4	0.54	No manure
East Upslope	1	0.65	No manure	East Upslope	1	0.85	No manure
	2	0.37	No manure		2	1.20	No manure
	3	0.49	No manure		3	0.90	No manure
	4	0.49	No manure		4	0.80	No manure
East Downslope	1	0.88	No manure	East Downslope	1	0.59	No manure
	2	1.36	No manure		2	0.69	No manure
	3	0.87	No manure		3	0.65	No manure
	4	0.79	No manure		4	0.86	No manure

A1.10. Available Phosphorus (mg kg^{-1}) in soil for 0-10 and 10-20 cm depths for the six landscape positions with manure and no manure treatment.
Plot ID, the landscape positions; REP, replication; P, available phosphorus; TRT, treatment

0 - 10 cm				10 - 20 cm			
Plot ID	REP	P	TRT	Plot ID	REP	P	TRT
North Upslope	1	2.66	No manure	North Upslope	1	3.09	No manure
	2	3.40	No manure		2	2.59	No manure
	3	4.86	No manure		3	2.36	No manure
	4	5.41	No manure		4	2.05	No manure
North Downslope	1	6.81	Manure	North Downslope	1	4.00	Manure
	2	6.58	Manure		2	3.43	Manure
	3	6.82	Manure		3	3.90	Manure
	4	6.11	Manure		4	3.70	Manure
South Upslope	1	3.81	Manure	South Upslope	1	4.44	Manure
	2	3.71	Manure		2	3.56	Manure
	3	3.62	Manure		3	2.71	Manure
	4	4.57	Manure		4	2.01	Manure
South Downslope	1	3.26	No manure	South Downslope	1	3.45	No manure
	2	3.32	No manure		2	1.33	No manure
	3	3.29	No manure		3	2.83	No manure
	4	3.31	No manure		4	1.52	No manure
East Upslope	1	0.54	No manure	East Upslope	1	0.11	No manure
	2	0.97	No manure		2	0.91	No manure
	3	0.54	No manure		3	0.34	No manure
	4	1.21	No manure		4	0.69	No manure

East Downslope	1	2.10	No manure	East Downslope	1	1.49	No manure
	2	3.46	No manure		2	0.44	No manure
	3	2.45	No manure		3	0.40	No manure
	4	2.37	No manure		4	0.60	No manure

A1.11. Available Phosphorus (mg kg^{-1}) in soil for 20-30 and 30-40 cm depths for the six landscape positions with manure and no manure treatment.
Plot ID, the landscape positions; REP, replication; P, available phosphorus; TRT, treatment

20 - 30 cm				30 - 40 cm			
Plot ID	REP	P	TRT	Plot ID	REP	P	TRT
North Upslope	1	1.60	No manure	North Upslope	1	1.92	No manure
	2	3.26	No manure		2	3.94	No manure
	3	2.43	No manure		3	0.40	No manure
	4	0.36	No manure		4	0.11	No manure
North Downslope	1	3.36	Manure	North Downslope	1	0.20	Manure
	2	1.31	Manure		2	2.39	Manure
	3	3.04	Manure		3	2.57	Manure
	4	0.01	Manure		4	2.10	Manure
South Upslope	1	2.54	Manure	South Upslope	1	2.71	Manure
	2	3.90	Manure		2	2.94	Manure
	3	2.67	Manure		3	3.05	Manure
	4	2.00	Manure		4	1.91	Manure
South Downslope	1	1.63	No manure	South Downslope	1	4.13	No manure
	2	1.23	No manure		2	0.90	No manure
	3	1.20	No manure		3	1.03	No manure
	4	2.77	No manure		4	0.93	No manure
East Upslope	1	0.21	No manure	East Upslope	1	0.29	No manure
	2	1.72	No manure		2	0.00	No manure
	3	0.02	No manure		3	0.87	No manure
	4	1.60	No manure		4	0.42	No manure
East	1	1.35	No manure	East	1	0.67	No manure

Downslope	2	0.46	No manure	Downslope	2	0.46	No manure
	3	0.67	No manure		3	0.31	No manure
	4	0.94	No manure		4	0.87	No manure

A1.12. Soil water retention (SWR, m3 m-3) of soil for 0-10 cm depth for the six landscape positions with manure and no manure treatment.

Plot ID, the landscape positions; REP, replication; TRT, treatment

Plot ID	TRT	Pressure (-kPa)						
		0.01	-0.4	-1.0	-2.5	-5.0	-10.0	-30.0
NU	No manure	0.57	0.55	0.51	0.48	0.44	0.42	0.37
ND	Manure	0.51	0.51	0.49	0.47	0.45	0.43	0.40
SU	Manure	0.52	0.51	0.49	0.47	0.45	0.43	0.38
SD	No manure	0.56	0.55	0.52	0.48	0.43	0.42	0.36
EU	No manure	0.55	0.54	0.52	0.49	0.45	0.42	0.38
ED	No manure	0.49	0.48	0.46	0.44	0.41	0.40	0.35

NU, North Upslope; ND, North Downslope; SU, South Upslope; SD, South Downslope; EU, East Upslope; ED, East Downslope

A1.13. Soil water retention (SWR, m3 m-3) of soil for 10-20 cm depth for the six landscape positions with manure and no manure treatment.

Plot ID, the landscape positions; REP, replication; TRT, treatment

Plot ID	TRT	Soil Pressure (-kPa)						
		0.01	-0.4	-1.0	-2.5	-5.0	-10.0	-30.0
NU	No manure	0.52	0.50	0.47	0.43	0.39	0.37	0.32
ND	Manure	0.52	0.50	0.48	0.45	0.42	0.41	0.35
SU	Manure	0.52	0.51	0.48	0.45	0.42	0.41	0.34
SD	No manure	0.53	0.51	0.48	0.43	0.41	0.39	0.33
EU	No manure	0.53	0.52	0.49	0.46	0.43	0.42	0.36
ED	No manure	0.48	0.47	0.44	0.41	0.39	0.38	0.34

NU, North Upslope; ND, North Downslope; SU, South Upslope; SD, South Downslope; EU, East Upslope; ED, East Downslope

Appendix 2

A2.1. Nutrient concentration (mg L-1) in the surface runoff samples collected during the three years 2013, 2014 and 2015 from the North watershed.

Dates	NO_3	NH_4	TKN	TDP	TP	TSS
3/9/2013	4.74	1.90	4.32	0.19	0.25	16.25
3/14/2013	4.04	3.23	9.83	1.62	1.53	33.10
3/26/2013	1.51	2.47	9.29	1.48	1.53	128.00
3/27/2013	2.21	3.02	10.40	1.47	0.93	185.10
3/28/2013	3.52	2.64	9.72	0.70	0.90	114.00
6/16/2014	3.25	2.22	8.35	0.54	3.88	943.00
6/6-7/2015	0.36	0.07	16.70	1.93	7.20	2674.00
6/19/2015	18.62	0.03	11.11	0.67	7.64	3287.21
6/20/2015	15.94	0.06	6.42	0.72	4.90	1568.36
7/6/2015	4.62	0.03	1.33	0.64	1.99	697.19

NO_3, Nitrate nitrogen; NH_4, Ammonium nitrogen; TKN, Total Kjeldahl nitrogen; TDP, Total dissolved phosphorus; TP, Total phosphorus; TSS, Total suspended solids

A2.2. Nutrient concentration (mg L^{-1}) in the surface runoff samples collected during the three years 2013, 2014 and 2015 from the South watershed.

Dates	NO_3	NH_4	TKN	TDP	TP	TSS
3/9/2013	3.93	1.49	3.51	0.08	0.11	11.50
3/14/2013	3.76	3.38	9.66	1.28	1.34	93.00
3/26/2013	1.20	1.58	6.46	0.30	0.29	76.50
3/27/2013	2.02	2.39	9.75	0.32	0.38	31.15
3/28/2013	x	x	x	x	x	x
6/16/2014	2.93	1.45	7.59	0.31	4.37	1150.00
6/6-7/2015	0.16	0.06	6.83	0.85	3.82	1393.60
6/19/2015	17.80	0.03	6.64	0.39	3.89	1260.88
6/20/2015	9.31	0.07	6.07	0.35	3.54	1161.73
7/6/2015	3.94	0.05	1.08	0.28	0.86	304.65

NO_3, Nitrate nitrogen; NH_4, Ammonium nitrogen; TKN, Total Kjeldahl nitrogen; TDP, Total dissolved phosphorus; TP, Total phosphorus; TSS, Total suspended solids; x, no data

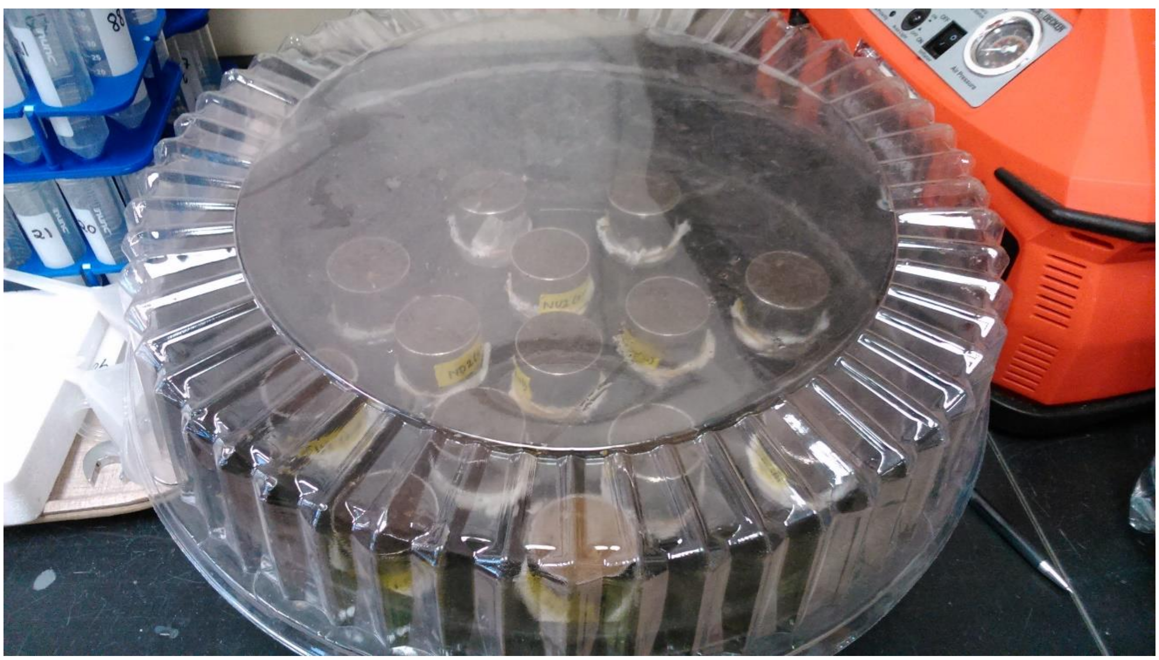
A2.3. Nutrient concentration (mg L^{-1}) in the surface runoff samples collected during the three years 2013, 2014 and 2015 from the East watershed.

Dates	NO_3	NH_4	TKN	TDP	TP	TSS
3/9/2013	4.58	1.62	3.62	0.10	0.14	23.50
3/14/2013	4.42	1.67	5.72	0.46	0.39	41.25
3/26/2013	1.13	1.81	6.90	0.51	0.52	37.00
3/27/2013	1.96	2.41	8.22	0.58	0.52	46.20
3/28/2013	2.67	1.37	8.88	0.35	0.43	66.40
6/16/2014	x	x	x	x	x	x
6/6-7/2015	0.33	0.10	3.66	0.38	1.86	400.55
6/19/2015	x	x	x	x	x	x
6/20/2015	x	x	x	x	x	x
7/6/2015	5.10	0.05	3.37	0.42	2.23	1263.83

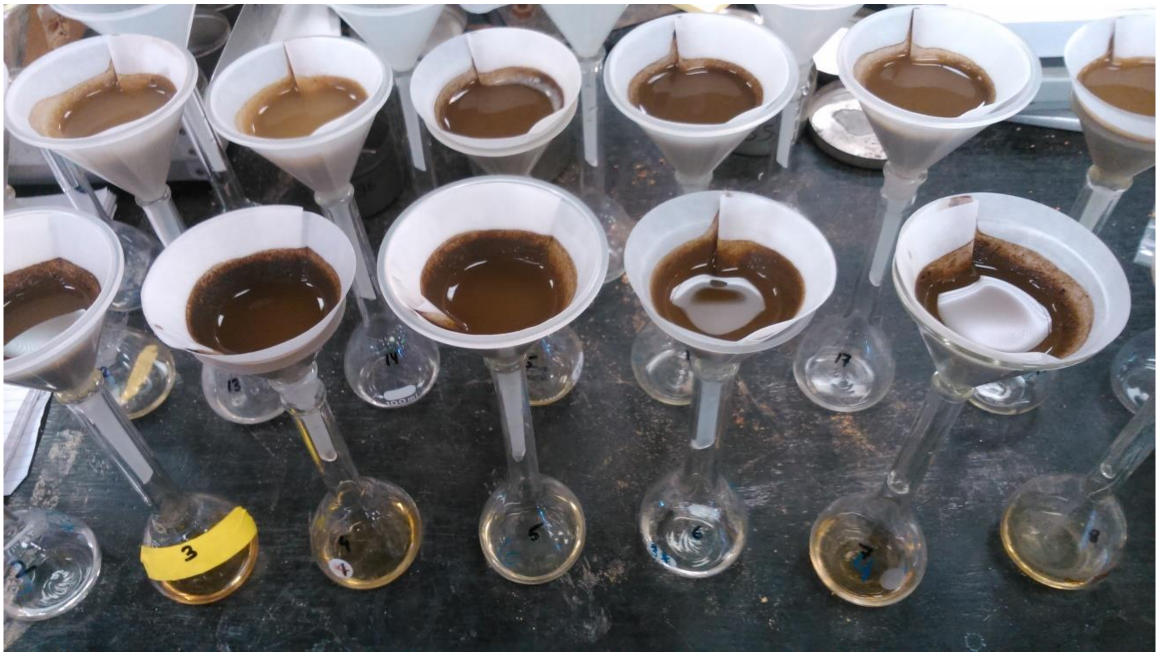
NO_3, Nitrate nitrogen; NH_4, Ammonium nitrogen; TKN, Total Kjeldahl nitrogen; TDP, Total dissolved phosphorus; TP, Total phosphorus; TSS, Total suspended solids; x, no data

A3.3. Pictures taken during soil sampling and analysis;

Soil sampling during October 2015

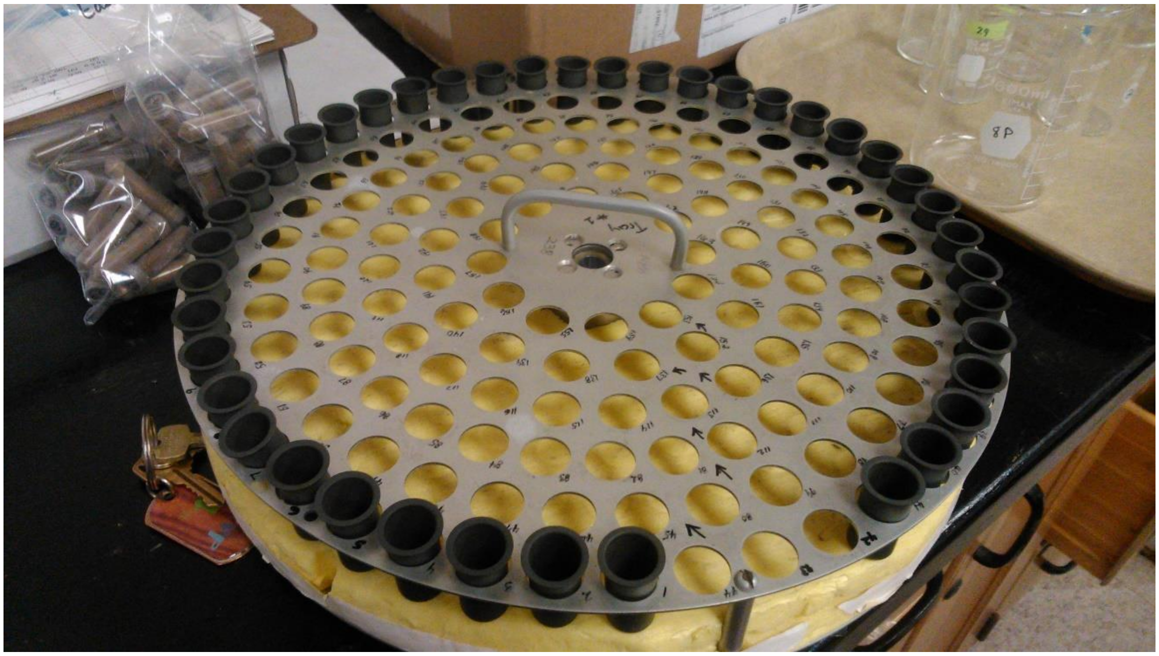


Saturation of soil cores for analysis of the soil water retention



Extraction of soil samples for the analysis of available phosphorus

Extracted soil samples ready to be analyzed for available phosphorus



Samples weighed for the total carbon and total nitrogen analysis

Analysis of soil inorganic carbon

A3.4. Pictures taken during spreading manure at the watersheds,

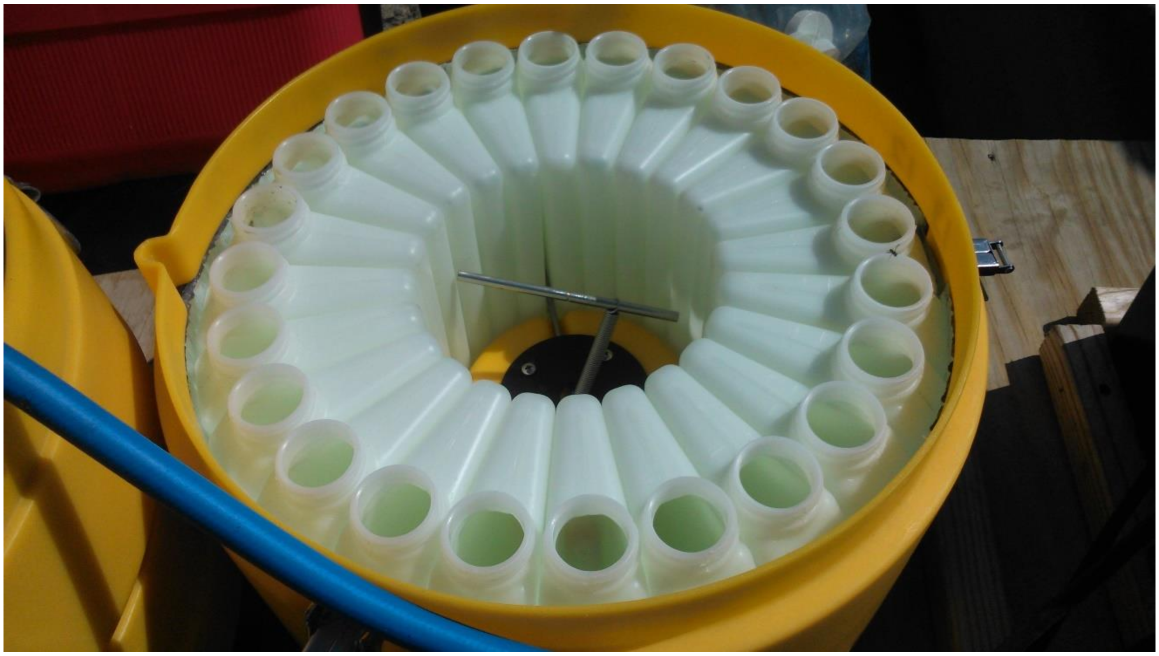
Manure spread during 2016

A3.5. Pictures taken during water sampling;

The set up at the outlet of each watershed

Collection of water samples after the storm events in 2015

Cleaning the clogged flume after the storm events



The inside picture of the automatic sampler containing 24 bottles

Evaluating E. coli particle attachment and the impact on transport during high flows.

Basic Information

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Project Number:	2016SD260B
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Publications

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2. McDaniel, Rachel. 2016. E. coli transport in South Dakota streams. In: 2016 UCOWR/NIWR Annual Water Resources Conference – Conference Proceedings. UCOWR/NIWR. Poster presentation, Pensacola Beach, FL

Evaluating *E. coli* particle attachment and the impact on transport during high flows

Annual Report: March 1, 2016 – February 28, 2017

Report Submitted To:
South Dakota Water Resources Institute under the USGS 104b program

Prepared By:
Rachel McDaniel, Assistant Professor
Louis Amegbletor, Graduate Research Assistant
Department of Agricultural and Biosystems Engineering, South Dakota State University
Brookings, SD 57007

Bruce Bleakley, Professor
Department of Biology and Microbiology, South Dakota State University
Brookings, SD 57007

1. Introduction

59% of South Dakota's assessed rivers and streams are considered impaired, and another 10% are threatened of becoming impaired (EPA, 2015). The primary causes for impairments are total suspended solids (TSS) and bacteria including *Escherichia coli* (*E. coli*). *E. coli* alone is responsible for poor water quality in over 2,000 miles of streams in South Dakota. Indicator organisms, such as *E. coli,* are used to indicate the presence of fecal pollution which can contain pathogenic microorganisms. Indicator organisms have been shown to be positively correlated with increases in gastrointestinal illnesses in recreational waters (e.g. Wade et al., 2006), therefore increasing public health risks.

High flows associated with storm events have been shown to greatly increase *E. coli* concentrations in streams (Krometis et al., 2007) and over 95 % of *E. coli* loading can occur during storm events (McKergow and Davies-Colley, 2010). Sediment resuspension may account for a major amount of fecal coliform (FC) and *E. coli* numbers during or soon after rainfall events (Pachepsky and Shelton, 2011). Bacteria often attach to particles (Krometis et al., 2007), such as silt, which causes them to settle out of the water column more rapidly, thus limiting their transport downstream. Free (unattached) bacteria, on the other hand, are more buoyant and have the ability to remain in the water column longer and travel farther in surface waters. Similarly, different particle sizes settle at different rates; for example, silt will settle faster than clay.

Krometis et al. (2007) evaluated microorganisms, including *E. coli,* during storm events and partitioned out those attached to settleable partilces. About 40% of *E. coli* was found to be attached to settleable particles with the highest concentrations of the associated settleable microbes occurring at the beginning of the storm. Attachment of fecal bacteria to various particle sizes in storm runoff was examined by Soupir et al. (2010). They determined that more bacteria are attached to finer particles (i.e. silt and clay) than courser particles (i.e. sand).

The Big Sioux River flows through Sioux Falls, SD, the largest city in the state, and is used for recreational purposes. However, water quality in the river, including *E. coli*, does not meet water quality standards. The poor water quality of the river has become a major concern. In response, the city of Sioux Falls has initiated an annual conference, the Big Sioux Water Summit, to inform local stakeholders of the current issues and progress in the watershed as well as discuss potential solutions to the water quality problems.

Skunk Creek is a tributary to the Big Sioux River and is a major contributor of *E. coli* to the river. The 2014 Water Quality Assessment Report lists Skunk Creek as impaired for *E. coli*, fecal coliforms, and TSS. Limited Contact Recreational and Warmwater Marginal Fish Life uses are not supported in Skunk Creek. To address these issues, best management practices have been implemented along the creek including Riparian Area Management (RAM) and Seasonal RAM (SRAM). These systems are focused on minimizing fecal bacteria loading from cattle by reducing or eliminating their time in the stream as well as providing a buffer between grazing lands and the stream to reduce overland transport.

The South Dakota Department of Environment and Natural Resources (DENR), East Dakota Water Development District (EDWDD), and the South Dakota Association of Conservation Districts are working to reduce *E. coli* loading in streams; however, high concentrations of *E. coli* are still often observed. Therefore, there is interest in furthering the understanding of how *E. coli* is transported to and within stream environments.

2. Objectives

The overall goal of this project was to evaluate *E. coli* attachment to particles of different sizes and estimate the impact of attachment on *E. coli* transport in streams during high flows. *E. coli* fate and transport are difficult to predict and this information may be incorporated into existing or future models to estimate *E. coli* concentrations in streams as well as contribute to the development of management practices to reduce transport. This goal was achieved through the following objectives:

- Evaluate *E. coli* concentrations and attachment rates to different particle sizes,
- Evaluate the relationship between particle size association of *E. coli* and shear stress,
- Estimate the *E. coli* load contribution by particle size, and
- Estimate the transport distance of the *E. coli* by particle size.

3. Methods

3.1 Study Site

The study site is located on Skunk Creek (Fig. 1) in eastern South Dakota (SD) near the intersection of 248th St. and Burk Ave. Eastern SD's climate has a humid continental climate with an average rainfall of about 27 inches annually. The land use surrounding the study site is largely rangeland and Seasonal Riparian Area Management (SRAM) is practiced in the immediate vicinity of the site. Producers enrolled in the SRAM program agree to keep their cattle out of the stream during the recreation season, roughly May through October, and are allowed to hay the buffer a few times during the year. Skunk Creek is a tributary to the Big Sioux River and is impaired for Total Suspended Solids (TSS) and *E. coli*.

3.2 Sample Collection

An Avalanche Teledyne ISCO refrigerated autosampler (Fig. 2) with velocity, depth, temperature, and turbidity sensors was installed at the site which provided data at 5-15 minute intervals throughout season. Water samples were collected during storm events over a five hour period at 30 minute intervals for a total of ten samples during each storm event. The samples were stored in the refrigerated unit until collected, usually within three hours of the sample collection completion. In addition, periodic grab samples were collected to assess baseflow conditions. All samples were collected in sterilized bottles and transported in an ice chest to the laboratory for processing and analysis.

Figure 1: Sampling was conducted in Skunk Creek located in eastern SD. The creek is impaired for bacteria and is surrounded by Seasonal Riparian Area Management which is intended to assist with the reduction of *E. coli.*

Figure 2: An Avalanche Teledyne ISCO autosampler was used to collect water samples during high flow events.

3.3 Sample Processing

Sample processing began within four hours of the final sample being collected. Stoke's Law (eq. 1) was used to partition bacteria attached to different particle sizes. Three particle size ranges were assessed which included sand and coarse silt (diameter ≥ 0.016 mm), fine silt (0.016 mm to 0.004 mm), clay and unattached particles (diameter < 0.004 mm). Bacteria attached to clay and unattached bacteria were combined due to their similar transport behaviors (i.e. both are relatively buoyant) and time constraints of settling the clay particles out of suspension.

$$v_s = \frac{g}{18}\left(\frac{\rho_s - \rho_w}{\mu}\right) d^2 \qquad \text{(eq. 1)}$$

Where v_s is the settling velocity, ρ_s is the particle density, ρ_w is the density of water, μ is the dynamic viscosity of water, and d is the particle diameter.

To begin the settling process, the sample was inverted in the sample bottle to resuspend the particles and bacteria in the water. Approximately 500 mL of water was transferred to a graduated cylinder which was then placed in a refrigerator to minimize the growth or decay of bacteria during the settling process. One sub-sample was collected from the graduated cylinder immediately to determine the total concentration of bacteria within each sample. Additional sub-samples were collected as the particles settled out of the water column at 6.25 minutes and at 2.22 hours, allowing the sand and course silt, and fine silt to settle out, respectively. Each sub-sample was processed in triplicate using standard membrane filtration on modified mTEC agar.

Once plated, the bacteria were placed in a water bath at 35°C for two hours to allow for the resuscitation of any stressed bacteria. Next, the plates were placed in an incubator at 44.5°C for an additional 22 hours. The resulting magenta colonies were counted and recorded.

3.4 Data Processing and Analysis

Shear stress is the force per unit area parallel to a surface and is associated with resuspension of *E. coli* reservoirs from stream bed sediments into the water column. Shear stress was calculated via Equation 2 (Jamieson et al., 2005) and compared to *E. coli* concentrations measured during the sampling period. The shear stress required to resuspend *E. coli* varies based on if the bacteria are attached to particles and the size of particle they are attached to.

$$\tau_b = yS^{1/4}\left(\frac{n}{A}\right)^{3/2} Q^{3/2} \qquad \text{(Equation 2)}$$

Where τ_b is the bed shear stress, y is the specific weight of water, S is the slope, n is Manning's roughness coefficient, A is the cross-sectional area, and Q is flow.

The *E. coli* loads (Equation 3) from the 5 hour sampling period were evaluated for total *E. coli*, unattached *E. coli*, and the attachment to each particle size. The loads provide knowledge about the quantity of *E. coli* moving through the sampling location during high flow conditions. By splitting the load into size fractions, estimates were made about how each fraction is moving in the stream, including how far each fraction was transported.

$$L = cQt \qquad \text{(Equation 3)}$$

Where L is the *E. coli* load, c in the *E. coli* concentration, Q is the flow, and t is time. The flow data was determined using a velocity and depth meter connected to the autosampler. Evaluating the *E. coli* loading during each time interval for each particle size provides information on when the various partitions are mobile and which particle sizes are contributing the highest *E. coli* load during high flow events.

A preliminary estimate of the transport distance (Equation 4) was made by combining Stoke's law (Equation 1 above), the stream depth, and the stream velocity. The result provides an initial prediction of the distance each fraction of the *E. coli* load is moving during high flow periods. Future work will need to be completed to incorporate other potential forces, stream characteristics, etc. that may affect the transport of *E. coli* in the stream environment.

$$D_T = \frac{Q}{W\, v_s} \qquad \text{(Equation 4)}$$

Where D_T is the transport distance, Q is the flow, W is the average width of the stream, and v_s is the settling velocity.

4. Results and Discussion

Three storm events were processed for *E.coli* concentrations and attachment; however, the first and second storm events were used to perfect the sample collection and processing procedures. Therefore, only one set of storm samples (10 samples in total) and one baseflow sample is presented herein. Additional storm events and baseflow conditions will be monitored during the summer and fall of 2017 to ensure a complete dataset.

4.1 *E. coli* concentrations and attachment rates

During the storm event, the total *E. coli* concentrations ranged from 2.7 x 10^3 to 8.47 x 10^3 CFU $100mL^{-1}$ (Figure 3a), with an average concentration of 4.71 x 10^3 CFU $100mL^{-1}$ (Table 1). The total *E. coli* concentration for the assessed baseflow condition, on the other hand, was lower at 1.0 x 10^3 CFU $100mL^{-1}$. The designated use for Skunk Creek is limited contact recreation meaning that the total *E. coli* concentrations should not exceed 1.178 x 10^3 CFU $100mL^{-1}$ in any one sample, also known as the single sample maximum (SSM). All storm flow samples exceeded the SSM, while the assessed baseflow condition did not. These tentative results indicate that high flow storm events negatively influence the bacterial water quality in Skunk Creek. The total concentration of bacteria fluctuated with the highest concentrations during the beginning and end of the sampling period.

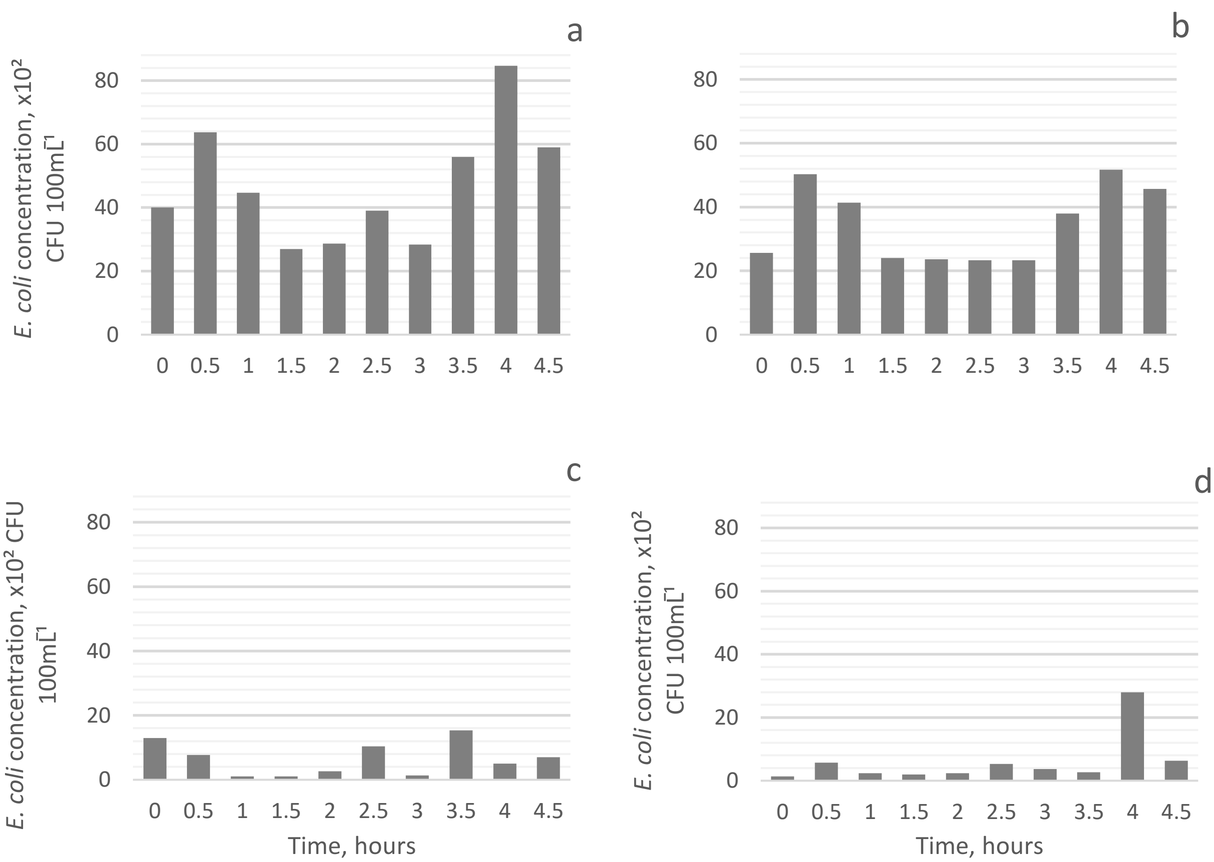

Figure 3: The *E. coli* concentration of the total (a) and each particle fraction, including unattached (b), fine (c), and coarse (d) particles. Samples were collected every half hour for five hours.

The unattached and clay fraction (diameter < 0.004 mm), hereafter referred to as "unattached", had the highest *E. coli* concentration for both storm flow and baseflow samples. The highest concentrations of unattached bacteria followed a similar pattern to the total, where the highest concentrations were found at the beginning and end of the sampling period (Figure 3b). For storm events, 75% of the total concentration travels attached to clay or unattached to particles, with a similar proportion (77%) seen in the baseflow (Table 2). However, the concentration of the unattached bacteria was an order of magnitude higher during the storm event (3.47×10^3) than during baseflow conditions (8×10^2). Medium and coarse silt, hereafter referred to as "coarse silt", consisted of 11% and 14% of the storm flow and baseflow *E. coli* concentrations, respectively, and was relatively low throughout the storm event. Fine and very fine silt, hereafter referred to as "fine silt", consisted of a similar proportion of the total *E. coli* concentration as the coarse silt at 14% and 9% for storm flow and baseflow, respectively, and concentrations were relatively low throughout the duration of the sampling period (Figure 3c).

Table 1: Summary of *E. coli* concentrations (10^2 CFU $100mL^{-1}$) during storm flow and baseflow conditions.

Fraction of *E. coli* Concentration	Storm flow 10^2 CFU $100mL^{-1}$ (Mean±SD)	Baseflow 10^2 CFU $100mL^{-1}$
Total *E. coli*	47.1± 18.63	10
Medium and Coarse Silt	5.97±7.93	1.5
Fine and very Fine Silt	6.43 ±5.17	0.9
Clay and Unattached	34.7 ± ×11.94	8

Table 2: Summary of the proportion of each particle fraction contribution to the *E. coli* concentration during both storm flow and baseflow conditions.

Fraction of *E. coli* Concentration	Storm flow	Baseflow
Medium and Coarse Silt	11%	14%
Fine and Very Fine Silt	14%	9%
Clay and Unattached	75%	77%

4.2 Relationship between *E. coli* attachment and shear stress

Spearman's correlation was calculated between *E. coli* concentrations and shear stress as well as *E. coli* concentrations and turbidity. The data collected thus far does not demonstrate a significant correlation between shear stress and *E. coli* concentrations from the different particle sizes. Additional data will be collected and analyzed to provide a more robust dataset for statistical analysis. The strongest correlation ($\rho = 0.54$) was found between shear stress and the *E. coli* concentrations associated with the coarse silt (Table 3). This is unsurprising as this fraction settles out quickly and is thus highly dependent on the forces contributing to resuspension (i.e. shear stress) immediately preceding the sample. Fine silt and unattached bacteria, on the other hand, stay suspended in the water column for longer periods of time, thus the forces responsible for their resuspension occurred, at least in part, some time previous to sample collection.

Table 3: Spearman's correlation (bold) and p-value (in parentheses) for the total *E. coli* concentration within the sample, the concentration associated with the coarse silt, the concentration associated with the fine silt, the concentration that is unattached to particles, and the concentration associated with all particle sizes (i.e. coarse silt and fine silt).

Parameter	Total *E. coli* Concentration	Medium and Coarse Silt	Fine and Very Fine Silt	Clay and Unattached	All Attached
Shear Stress	**0.16**	**0.54**	**0.14**	**-0.03**	**0.51**
	(0.65)	(0.11)	(0.70)	(0.93)	(0.13)
Turbidity	**-0.26**	**-0.67**	**-0.06**	**-0.12**	**-0.40**
	(0.47)	(0.03)	(0.87)	(0.74)	(0.26)

The turbidity was negatively correlated with *E. coli* concentration, which was surprising. Often storm events will increase the movement of particles with similar methods (e.g. resuspension) as bacteria, and thus they are often highly positively correlated. However, this data demonstrates that in the SRAM conditions seen on Skunk Creek, turbidity is negatively correlated with bacteria concentrations (Table 3). The bacteria concentration associated with the coarse silt shows a significant ($p < 0.05$) negative correlation with turbidity with a Spearman's correlation of -0.67. Further investigations will be conducted on both storm flow and baseflow conditions to confirm these relationships.

4.3 *E. coli* load contribution by particle size

The total *E. coli* load over the five hour sampling period was 71.8 x 10^{10} CFU, with the majority (73%) of the load from the unattached fraction of the *E. coli* concentration (Table 4). Both the Coarse and Fine silts contributed nearly equally at 9.8 x 10^{10} and 9.4 x 10^{10} CFU, respectively. This results in all attached *E. coli* contributing nearly 27% of the total *E. coli* load throughout the duration of the five hour storm sampling period.

Table 4: The total E. coli load (x10^{10} CFU) for each particle size fraction. The unattached bacteria contributed the largest proportion of the total load.

Fraction of *E. coli* Concentration	Load (x10^{10} CFU)
Total Loads	71.8
Medium and Coarse Silt	9.80
Fine and very Fine Silt	9.40
Clay and Unattached Loads	52.6

4.4 Estimated transport distance

The transport distance was estimated using the settling velocity for the particle size and the average velocity at the time of the given sample. Bacteria attached to coarse silt settled out the soonest, traveling an estimated distance between 0.09 and 0.14 km. The *E. coli* attached to fine silt traveled about 1.41 to 2.17 km and the unattached traveled approximately between 26.3 and 33.8 km downstream before settling out of the water column. Additional factors, such as turbulent flow, varying channel depth, etc. were not considered. Further investigations will need to be conducted to improve the accuracy of these estimates.

5. Summary and Conclusions

The concentrations of *E. coli* observed during high flows were consistently above the SSM standard for limited contact recreation. Removing the easily settleable, attached bacteria still resulted in concentrations above the SSM. Approximately two-thirds of the bacteria were transported unattached to particles, indicating a long time period would be required to settle the majority of the bacteria out of suspension. The *E. coli* loads attached to the coarse and fine silts particle sizes were nearly equal, while the *E. coli* loads from the unattached fraction were an order of magnitude higher. In addition, shear stress had the highest correlation with bacteria attached to coarse particles. This demonstrates that the force resulting from the change in flow immediately surrounding the sampling period has the largest impact on the heavy particles. These bacteria also settled out relatively rapidly, within 150 m. The unattached bacteria, on the other hand, stayed in suspension and contributed to poor water quality for up to an estimated 33 km. Therefore, these preliminary results demonstrate that management practices that can reduce this unattached load will play a vital role in achieving bacteria levels within the SSM for limited contact recreation in Skunk Creek.

6. Acknowledgements

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7. References

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Controlling Harmful Algal Blooms in Eutrophic Lakes by Combined Phosphorus Precipitation and Sediment Capping (Year 2)

Basic Information

Title:	Controlling Harmful Algal Blooms in Eutrophic Lakes by Combined Phosphorus Precipitation and Sediment Capping (Year 2)
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Descriptors:	None
Principal Investigators:	Kyungnan Min, Guanghui Hua

Publications

There are no publications.

Controlling Harmful Algal Blooms in Eutrophic Lakes by Combined Phosphorus Precipitation and Sediment Capping

Annual Report: March 1, 2016 to February 28, 2017

Report Submitted to the South Dakota Water Resources Institute under the USGS 104b program

Kyungnan Min, Sepideh Sadeghi, Guanghui Hua

Department of Civil and Environmental Engineering, South Dakota State University, Brookings, SD 57007

Introduction

South Dakota Department of Environment and Natural Resources (DENR) assessed 143 lakes based on numeric water quality standards and nutrient-related narrative standards (SD DENR, 2014). Approximately 60% of the assessed lakes do not support one or more assigned beneficial uses. Lakes in South Dakota are impaired by excessive nutrients and siltation generated from non-point source pollution. The trophic status indicates that 118 out of 143 lakes are characterized as eutrophic to hypereutrophic condition. Eutrophication can lead to the development of harmful algal blooms of cyanobacteria (blue-green algae), which will result in detrimental effects on lake water quality including increased turbidity, dissolved oxygen depletion, and scum layers formation (Smith et al., 1999). Moreover, cyanobacteria can release potent toxins that pose significant threats to ecosystem and public health.

Phosphorus loading reduction is critical to eutrophication control because P frequently limits the primary production in lakes. It is generally accepted that the first step to control harmful algal blooms in eutrophic lakes is to reduce the external nutrient loading from point and non-points sources. However, many studies have shown that the lake recovery is a slow process even when the external P loading has substantially reduced. The internal P loading from P-rich sediment is one major factor that is responsible for enhanced eutrophication process (Gulati and Van Donk, 2002; Berg et al., 2004; Cooke et al., 2005). The phosphorus released from the sediment can delay the lake recovery for years to decades (Sondergaard et al., 2001; Cooke et al., 2005). Therefore, effective internal P loading control methods are necessary to accelerate lake recovery and achieve long-term lake eutrophication mitigation.

Sediment dredging is one of the in-lake restoration methods that can be used to reduce the internal P loading from the sediment (Peterson, 1982). However, sediment removal is relatively expensive and often cannot provide a permanent solution (Welch and Cooke, 2005). The in-situ sediment capping technology has been developed to control the P cycling from the sediment. This technology involves placement of a layer of particulate materials at the sediment-water interface to create a barrier between the sediment and overlaying water. The in-situ sediment capping is a promising technology that can stabilize sediment, minimize re-suspension, and reduce nutrient release from the sediment (Simpson et al., 2002; Kim et al., 2007; Lin et al., 2011). Clean sand has traditionally been used for in situ capping of sediment for eutrophication control. Recently, active barrier system has been developed to improve the effectiveness of sediment capping. In this system, reactive materials are used to bind containments in the sediments by adsorption or precipitation, thereby improving the capping efficiency. These reactive materials include activated carbon, gypsum, modified sand, natural and modified zeolite, calcite and others (Berg et al., 2004; Jacobs and Waite, 2004; Park et al., 2007; Viana et al., 2008; Pan et al, 2012). Several studies have shown that calcite is effective in preventing phosphorus release from sediment under anaerobic conditions (Hart et al., 2003; Berg et al., 2004). Lin et al. (2011) used a mixed calcite and zeolite medium for sediment capping in a laboratory study. The results showed that the mixture was able to simultaneously control phosphorus and ammonium release from the sediment. A lanthamum-enriched benthonite clay (Phoslock®) has been proven to be a strong P binding material in several laboratory and field experiments (Lurling and Van Oosterhout 2013). These P binding materials can be used as active barrier systems for sediment capping in eutrophic lakes to provide long-term inhibition of P recycling from the sediments.

Direct precipitation of P and algae cells is an effective remedial method that can quickly reduce the lake water P content and mitigate lake algal blooms. Aluminium-, calcium-, and iron salts are the common chemical coagulants that have been applied for algal bloom control and P reduction in lakes (Cooke et al., 2005). Phytoplankton can be precipitated more effectively when these coagulants are used together with clay particles as ballast (Wang et al., 2012). Although chemical and physical precipitation can remove the total P from the water column, it does not provide a long-term prevention of P release from the sediment due to the re-suspension of the precipitated flocs. The combined phosphorus precipitation and sediment capping technology is a promising method to control harmful algal blooms in eutrophic lakes. The treatment involves precipitation of dissolved and particulate P from water column and subsequent immobilization of any P released from the sediment using reactive materials (Pan et al., 2012; Lurling and Van Oosterhout, 2013). The precipitation-capping technology could also promote the development of sandy sediment and facilitate a sustainable improvement of sediment-water environment.

The objective of this study is to develop an effective technology using precipitation and sediment capping to control harmful algal blooms and P levels in eutrophic lakes. Laboratory coagulation experiments were performed to evaluate factors affecting the dissolved P and algal cells precipitation using coagulants and reactive P binding particles to precipitate dissolved P and algal cells. The long-term sediment incubation experiments were conducted to measure P flux from the sediment-water interfaces after capping with reactive P-binding particles. The results of this project will provide critical information on the application of the P precipitation and sediment capping technology for in lake restoration in South Dakota. This may eventually lead to the development of an effective lake management tool that can be used as a sustainable eutrophication control strategy to accelerate the lake recovery in South Dakota and many other areas.

Materials and Methods

Lake Water Samples and Natural Minerals

The lake water samples were collected from Lake Kampeska located in Watertown, South Dakota, and used as raw water samples for the coagulation experiments. The characteristics of the lake water are shown in Table 1. Aluminum sulfate (98% purity, Fisher Scientific) was used as the coagulant for this study. Calcite (97.9% $CaCO_3$), zeolite, silica sand (99.7% SiO_2) and limestone were obtained from Great Lakes Calcium Co., Bear River Zeolite Co., U.S Silica Company, and Martin Marietta Co., respectively. A six position jar tester (Phipps & Bird) was used for the coagulation experiments using 500 mL glass beakers.

Table.1. Lake Kampeska water characteristics

Water Quality Parameters	Values
pH	8.5
Alkalinity	252 mg/L as $CaCO_3$
Phosphate (mg/L)	2 mg/L
Nitrate (mg/L)	0.2 mg/L
Total Phosphate (mg/L)	2.25 mg/L
Ammonia (mg/L)	0.5 mg/L

Cyanobacteria Strains, Maintenance, and Culture Conditions

Anabaena sp. PCC 7120 (here in referred to as Anabaena sp.), a model species for filamentous cyanobacteria, was obtained from the Pasteur Culture Collection of Cyanobacteria (Paris, France). For long-term storage, strains were frozen at -80°C in 5% v/v methanol. For short-term maintenance the cyanobacteria were grown on BG11 agar (1.5% agar) (Allen and Stanier, 1968) at pH 7.1, incubated at room temperature under constant illumination of 24 µmol m^{-2} s^{-1}, and then stored at room temperature. Light intensity was measured with a Heavy Duty Light Meter with PC Interface (Extech Instruments, Waltham MA, USA).

Cyanobacterial cultures were grown in 40 L photobioreactors (PBRs). PBR trials were conducted in 40 L transparent fiberglass flat bottom tanks (Solar Components Corp., Manchester, NH, USA) that were sparged from the bottom with a mixture of 95-5% air-CO2 at a rate of 0.25 L/min. The culture medium consisted of 30 L of BG11 and was inoculated with 1.5 L (5%) of an Anabaena sp. 7120 culture that had been grown to mid-log phase. The reactors were incubated until 2 days after stationary phase was reached at room temperature (20-22° C) under constant illumination of approximately 40 µE m^{-2} s^{-1} using fluorescent lights.

Experimental Procedure

The experiments for this project were divided into three phases. In Phase 1, the impact of natural particles on alum coagulation of cyanobacteria spiked lake water was investigated. All particles were sieved with Sieves # 100 (0.15 mm), 140 (0.106 mm), 200 (0.075 mm) and 325 (0.044 mm), and washed with deionized water to remove any fine particles. They were then dried prior to use. Different amounts of cyanobacteria (Anabaena7120) were dosed into 500 mL lake water to provide an initial turbidity of 20 NTU or 50 NTU. Then, particles and alum were applied to the water sample at different doses. The coagulation tests proceeded with rapid mixing at 250 rpm for 2 min, followed by slow mixing at 30 rpm for 20 min. Then, sedimentation was allowed to occur for 24 h. Sample were taken for the measurement of phosphate and Chlorophyll α. In order to study the impact of added particles on settling kinetics, coagulated samples were taken after 1, 2, 5, 10, 30 and 1440 min intervals evaluate the turbidity removal. Resuspension tests were also conducted at different velocity gradients (0 - 500 s^{-1}) for 5 minutes for the coagulated samples after 24 h settlement. The stirrer was located at ~3 cm above the sediment layer. For all experiments, water samples were taken from 1 cm below the water surface to measure different water quality parameters.

In Phase 2 experiments, we evaluated the synergistic effects polyaluminium chloride (PAC) and phoslock on precipitation of phosphate and cyanobacteria. Three commercial PACs with different basicity (42%, 55% and 77%) were used for these experiments. Lake Kampeska water samples spiked with Anabaena sp. (20 NTU) were treated with PACs (1-8 mg Al/L) and phoslock (100-800 mg/L) individually at different doses to evaluate their efficiencies for phosphate and Chlorophyll α removal. Then, the water samples were treated with PAC (77% basicity) and phoslock simultaneously at selected doses. The impact of PAC and phoslock on turbidity settling kinetics was also evaluated.

In Phase 3, laboratory column reactors were built for the long-term sediment incubation experiments. The column reactors had a diameter of 8.7 cm. Each reactor contained a 10 cm of sediment collected from Lake Kampeska and 30 cm of lake water. The column reactors were spiked with Anabaena sp. to achieve an NTU of 20. Alum coagulation (4 mg Al/L) was conducted on each reactor. Then, sand, limestone, zeolite, and phoslock were added to the reactors individually to achieve a capping layer of 1 cm. Two control reactors without any capping materials were also used for the experiments. One control reactor only contained the lake water spiked with Anabaena sp. and the other reactor was treated with alum. Water samples at 5 cm above the sediments were taken at different time intervals (10 to 80 days) for phosphate measurement.

Analytical Methods

The analysis of ammonia, nitrate and phosphate was carried out using UV-visible spectrophotometer (HACH, DR 4000, USA). Chlorophyll α was analyzed using UV-visible spectrophotometer (Shimadzu, UV-160/160A, Japan) based on the standard method (10200 H. Chlorophyll) (Clesceri et al, 1998) for examination of water and wastewater. A turbidity meter (HACH, 2100P, USA) was used to measure turbidity.

Results and Discussion

Effect of Alum and Particle Dosage on P and Anabaena sp. Removal

Figure 1 presents the effect of different alum dosages (0 to 8 mg Al/L) on nutrients and chlorophyll removal. Alum coagulation effectively removed total phosphate, orthophosphate and chlorophyll α, especially when the dose was higher than 4 mg Al/L. The maximum alum dosage used in this study (8 mg Al/L) was able to remove up to 90% of these parameters. However, alum coagulation did not affect the concentration of nitrate. The maximum removal efficiency of 60% was achieved for ammonia at the highest dose of 8 mg Al/L.

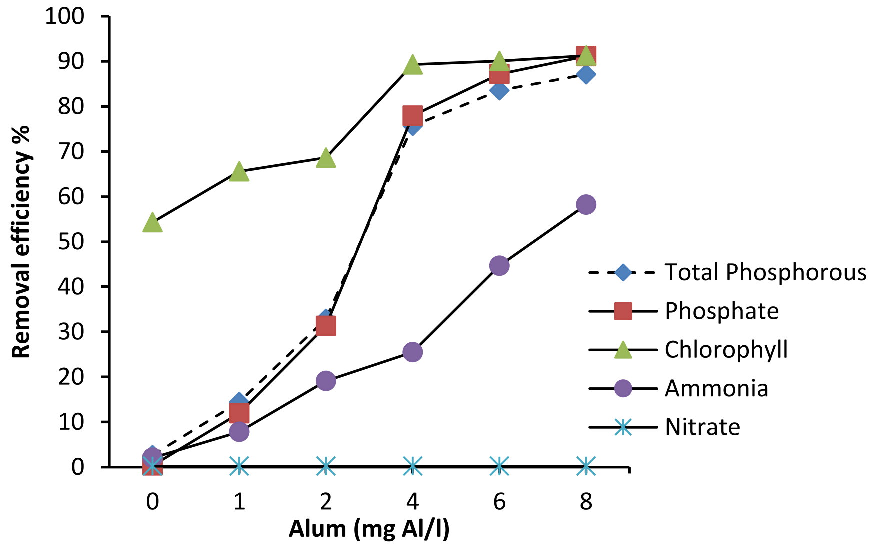

Figure 1 Effect of alum dosage on TP, PO_4^{-3}, Chl-a, NH_3 and NO_3^- removal efficiency

Table 2 shows the effect of different particles doses on the phosphate and chlorophyll α removal efficiency. In this test, alum (4 mg Al/L) was added to the water samples spiked with different amounts of sand, calcite, zeolite and limestone. Water samples were taken after 24 h settling. The added particles has limited impact on the chlorophyll α removal efficiency since the differences were less than 5% for different treatments. However, the phosphate removal gradually decreased with increasing particle doses. This is likely caused by the competitive adsorption of alum by these particles. Zeolite showed the largest reduction in phosphate removal whereas sand showed the least.

Table 2. Effect of particle dose on Chlorophyll α and phosphate removal (alum 4 mg Al/L, particle size 45-75 μm).

Coagulant + clay (g/L)	Chl- α Removal Efficiency (%)	PO_4^{-3} Removal Efficiency (%)
Alum (4 mg Al/L)	89.50	78.15
Alum + 0.2 g/L Calcite	90.56	77.92
Alum + 0.4 g/L Calcite	89.96	77.54
Alum + 1 g/L Calcite	88.81	74.74
Alum + 2 g/L Calcite	88.06	76.01
Alum + 5 g/L Calcite	87.70	71.41
Alum + 0.2 g/L Zeolite	90.56	75.81
Alum + 0.4 g/L Zeolite	89.50	73.90
Alum + 1 g/L Zeolite	89.13	72.89
Alum + 2 g/L Zeolite	88.39	69.65
Alum + 5 g/L Zeolite	85.57	62.85
Alum + 0.2 g/L Lime stone	89.82	77.11
Alum + 0.4 g/L Lime stone	89.87	78.41
Alum + 1 g/L Lime stone	90.19	76.21
Alum + 2 g/L Lime stone	89.45	70.72
Alum + 5 g/L Lime stone	89.13	71.73
Alum + 0.2 g/L Sand	90.56	78.18
Alum + 0.4 g/L Sand	90.88	76.94
Alum + 1 g/L Sand	90.19	78.59
Alum + 2 g/L Sand	89.82	75.23
Alum + 5 g/L Sand	90.26	74.30

Effect of Particle Addition on Floc Settling Kinetics

Figure 2 shows the effect of particle sizes on the turbidity reductions at different settling times. The particles in the size ranges of 0.106 to 0.15 mm did not affect the settling of the coagulated flocs. However, all four particles showed enhanced floc settling when the particles size were less than than 0.075 mm. The added particles in the size of 0.044 mm substantially increased the settling kinetics of the flocs, especially during the first 5 minutes. The effect of the enhanced settling by these particles diminished when the settling time was more than 30 minutes.

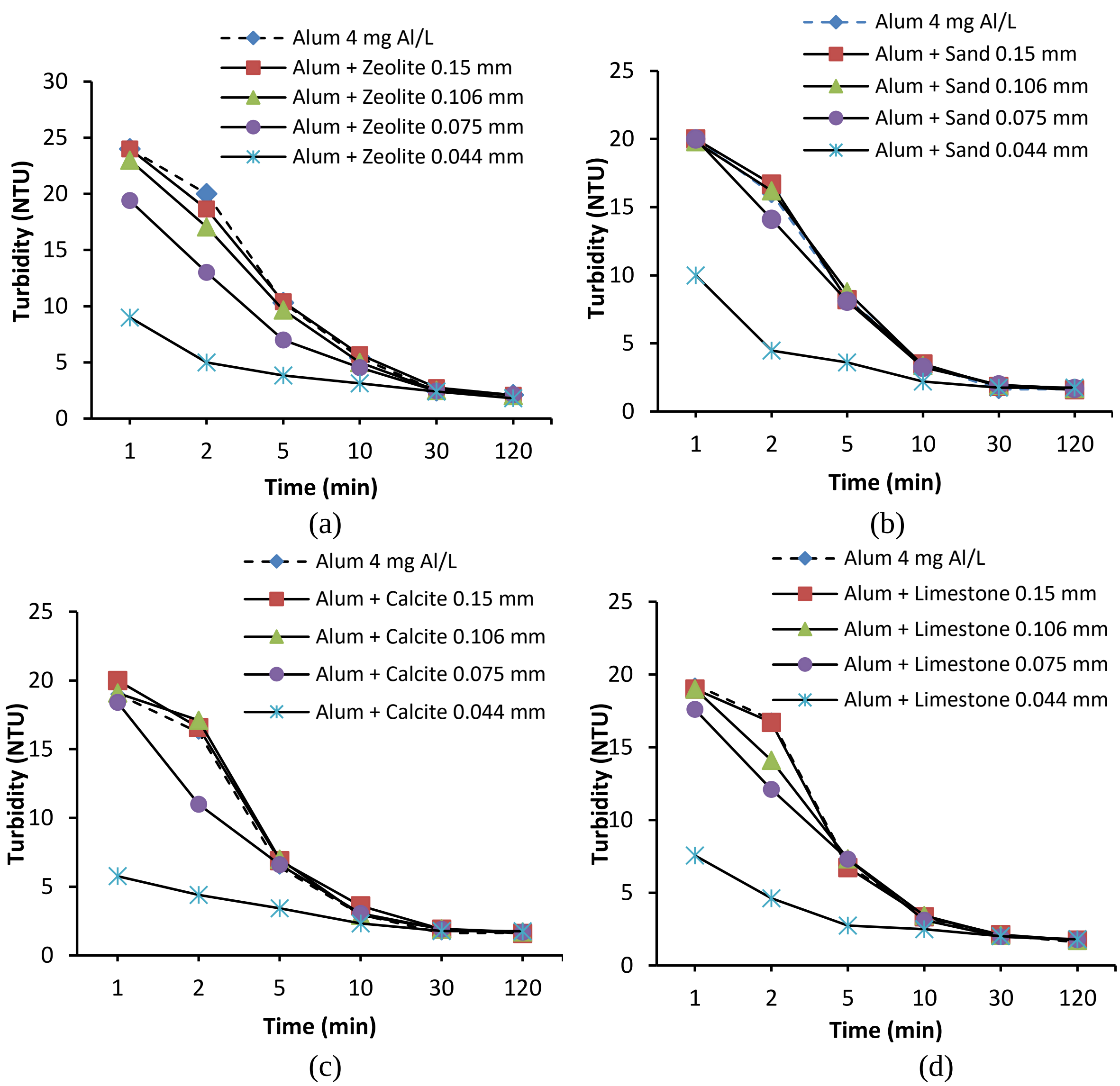

Figure 2 Effect of different particle sizes; (a) zeolite, (b) sand, (c) calcite and (d) limestone on turbidity removal *vs.* time (alum dosage 4 mg Al/L, particle dosage 1 g/L)

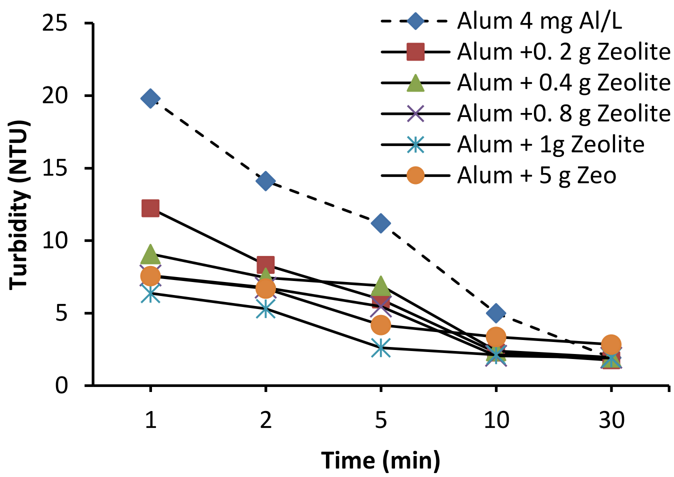

Figure 3 Effect of zeolite dosages on turbidity removal *vs*. time (alum dosage 4 mg Al/L, zeolite size 45-75 μm).

The influence of particle dosage of zeolite on turbidity removal is shown in Figure 3. Compared to alum alone, the added particles at different dosages enhanced the floc settling after coagulation. The optimum zeolite dose was determined to be 1 g/L. When further increasing zeolite dosage to 5 g/L, the performance of the floc settling reduced. This may be attributed to the high turbidity caused by zeolite itself at the highest dosage.

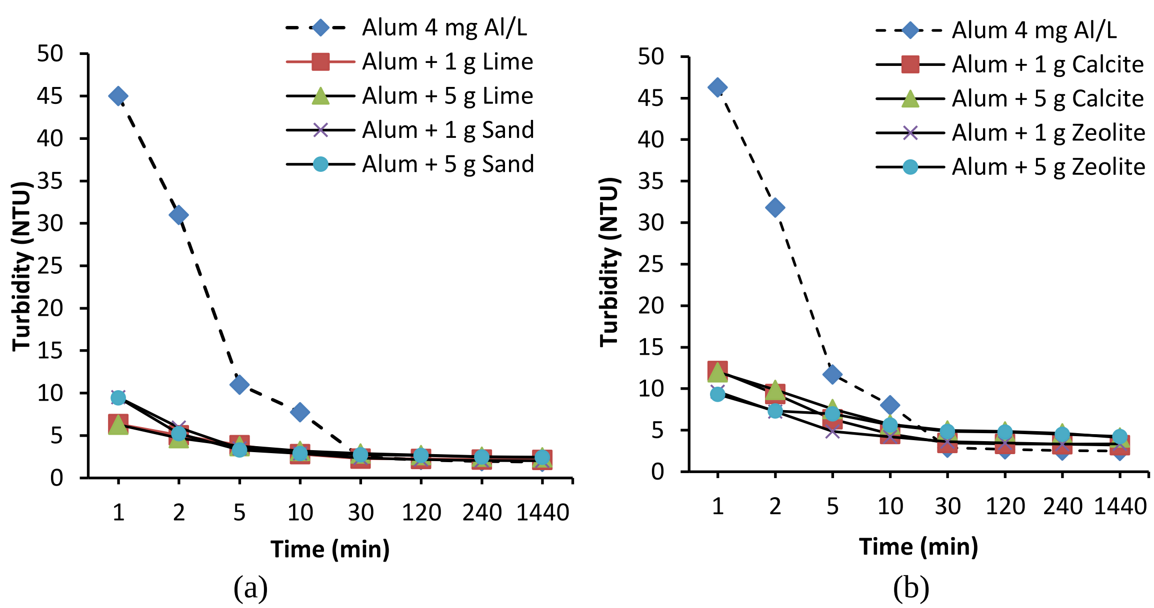

Figure 4 Effect of particles on high turbidity removal *vs*. time (initial turbidity ~ 50 NTU, alum dosage 4 mg Al/L, particle size 45-75 μm)

When the Lake Kampeska water sample were spiked with high concentration of Anabaena to reach initial turbidity levels of 40-50, the alum coagulation with particles also showed faster floc settling kinetics compared to alum only (Figure 4). This indicate that particles assisted coagulation can help the settling of the cyanobacteria and flocs for a wide range of initial cyanobacteria concentrations. Different particles and doses in the range of 1-5 g/L had similar impact on turbidity reduction.

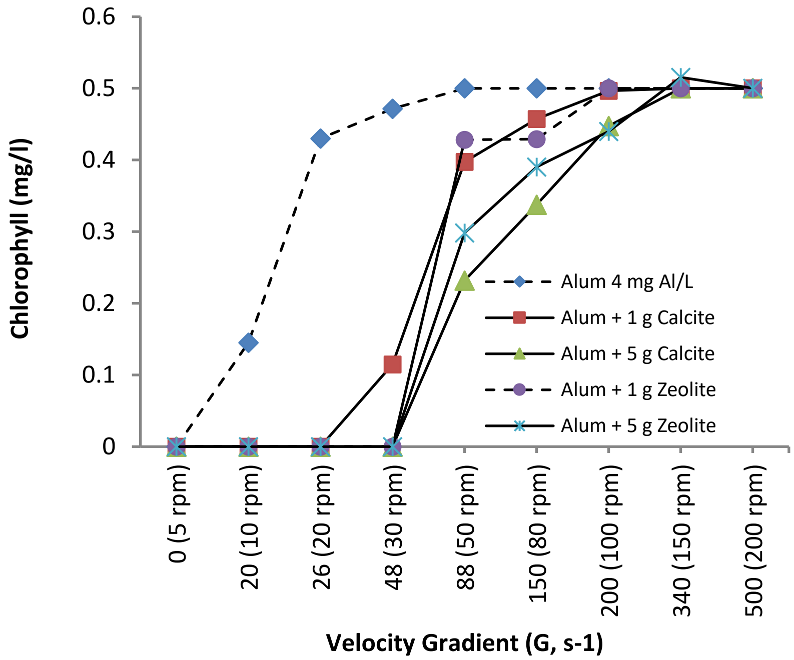

Figure 5 Effect of different velocity gradient (G, s^{-1}) on chlorophyll during resuspension (alum dosage 4 mg Al/L, particle size 45-75 μm)

Figure 5 shows the chlorophyll concentrations during the resuspension test. Coagulated flocs by alum alone were easily disturbed by the stirring at 10 rpm and were fully mixed when the stirring increased to 30 rpm. However, the settled flocs were much more resistant to the disturbance by the mixing. The chlorophyll concentrations started to increase at the mixing speed of 30 rpm and became completely mixed at 100-150 rpm.

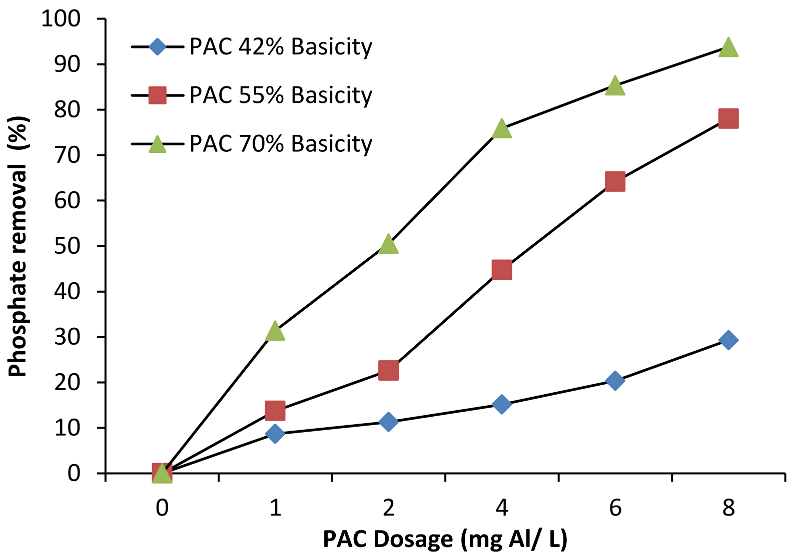

Figure 6 Effect of PAC dose on phosphate removal

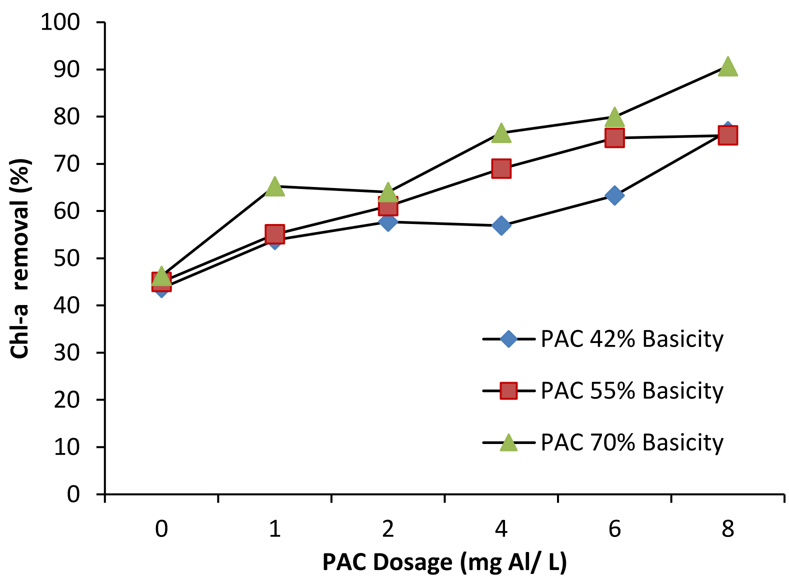

Figure 7 Effect of PAC dose on Chlorophyll α removal

Figure 6 and Figure 7 show the effects of PAC doses on phosphate and Chlorophyll α removal, respectively. The phosphate removal gradually increased with increasing PAC doses. The three PACs showed different efficiencies in removing phosphate from lake water. The PAC with 70% basicity showed much higher phosphate removal capacities than PACs with lower basicity. The phosphate removal percentages were 93.8%, 78%, and 29% for PACs with basicity of 70%, 55%, and 42%, respectively, at the highest dose of 8 mg Al/L. Similar trend was also observed for the removal of Chlorophyll α. The Chlorophyll α removal gradually increased with increasing PAC doses and the PAC with 70% basicity showed higher Chlorophyll α removal capacity than PACs with lower basicity. However, the differences in Chlorophyll α removal among the three PACs were much smaller.

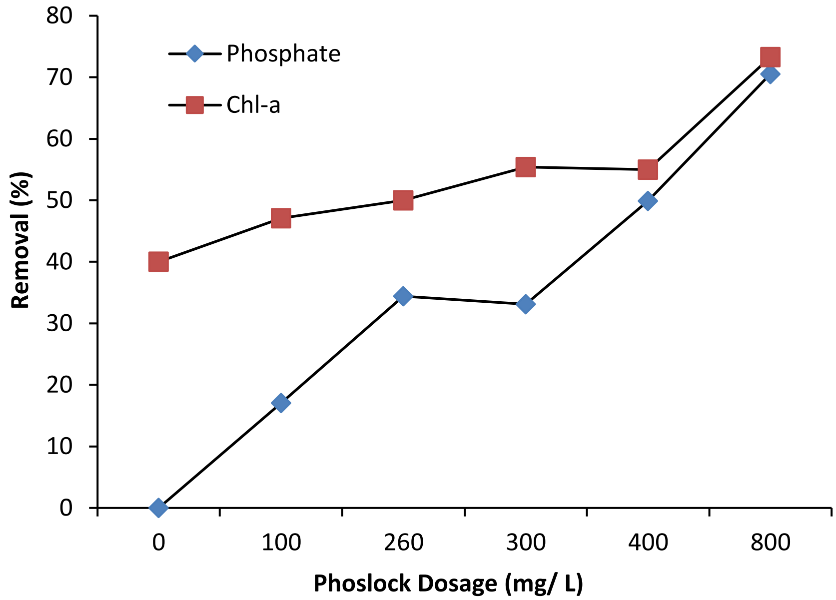

Figure 8 Effect of phoslock dose on phosphate and Chlorophyll α removal

Figure 8 presents the phosphate and Chlorophyll α removal by phoslock at different doses. The added phoslock increased the Chlorophyll α removal from 40% at the ambient condition to 73.2% at a dose of 800 mg/L. This indicate that phoslock was able to flocculate Anabaena sp. and enhance their sedimentation. The phoslock also showed high phosphate adsorption capacity. The phosphate removal increased near linearly with increasing phoslock doses. A 70.5% removal was observed at the highest dose of 800 mg/L.

Table 3. Effect of PAC and Phoslock on phosphate and Chlorophyll α removal

PAC (mg Al/L)	Phoslock (mg/L)	Chl-a removal (%)	Phosphate Removal (%)
1	300	73.2	72.5
2	300	96.1	78.6
4	300	99.4	88.5
4	200	99.8	84.8
4	100	99.4	81.8

Table 3 shows the phosphate and Chlorophyll α removal by combined PAC and phoslock. It is clear that the removal of phosphate and Chlorophyll α was substantially enhanced by using PAC and phoslock simultaneously. This may attributed to the synergistic effects of the high flocculation capacity of PAC and the high phosphate adsorption capacity of phoslock.

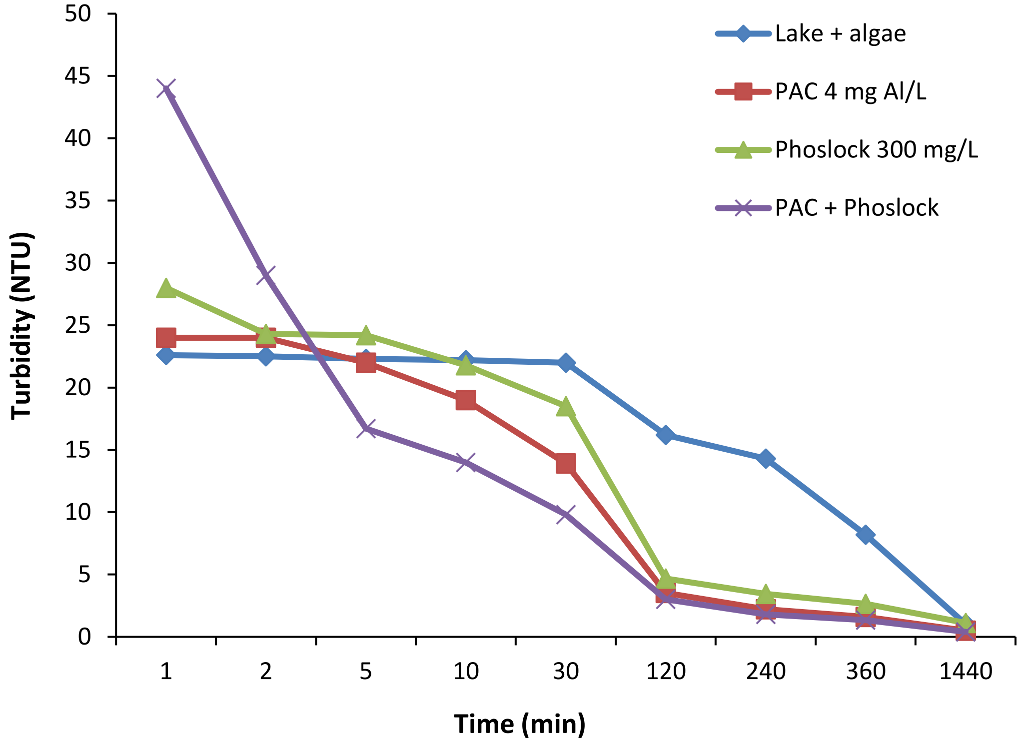

Figure 9 Effect of PAC and phoslock on turbidity settling kinetics

Figure 9 shows the effect of PAC and phoslock on turbidity settling kinetics. The Anabaena sp. spiked lake water showed slow turbidity removal kinetics. The turbidity remained stable during the first 30 min and then gradually decrease to 1 NTU after 1440 min. When the water samples were treated with PAC and phoslock individually, the turbidity showed appreciable reduction after 5 min of settling and reached an NTU below 5 after 120 min. Combined use of PAC and phoslock showed a rapid turbidity removal within the first 5 min and then the turbidity reduced to 3 NTU after 120 min. These results suggest that combined PAC and phoslock enhanced the settling of flocs formed during coagulation of Anabaena sp. spiked lake water.

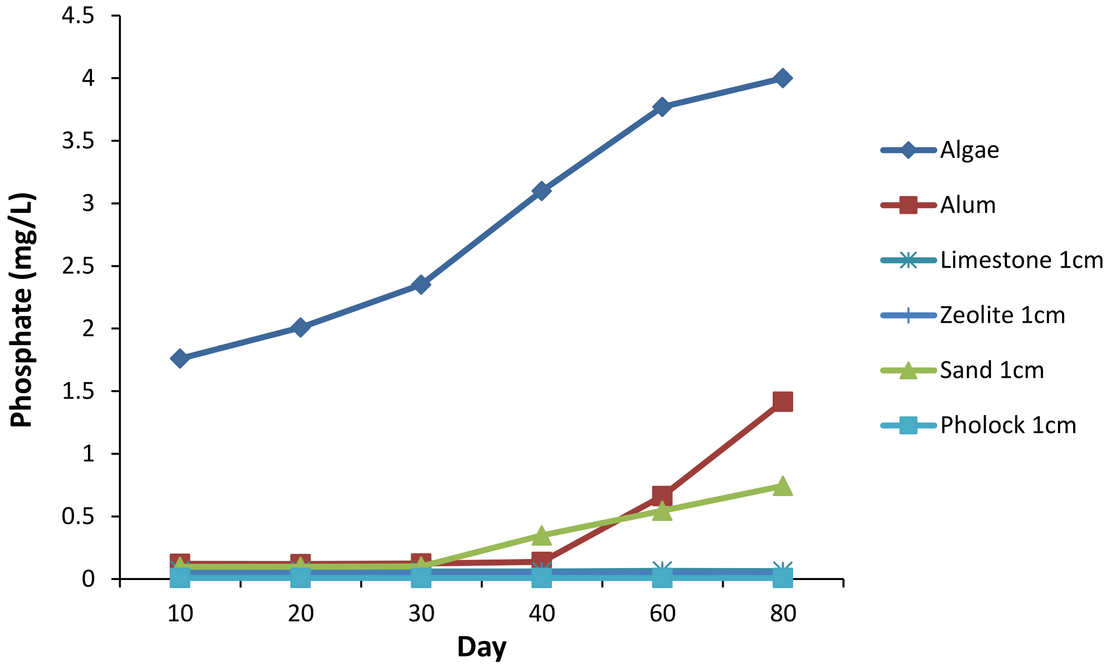

Figure 10 Phosphate release during the long-term incubation experiments

Figure 10 shows the phosphate release during the long-term incubation experiments. Without any treatment, the settled algae (Anabaena sp.) decayed and released a large amount of phosphate to the water. The phosphate concentration reached 4 mg/L after 80 days, which was twice of the ambient concentration of the lake water. Alum treated water showed elevated phosphate concentrations after 40 days of incubation, and reached a phosphate concentration of 1.4 mg/L after 80 days. Among the four capping experiments, sand was the least effective material. Elevated phosphate concentrations were observed in the sand capped column after 40 days of incubation and the phosphate reached 0.7 mg/L after 80 days. All other three capping materials exhibited high capacity in preventing phosphate release from the sediments and the Anabaena sp. The phosphate concentrations in these columns were below 0.1 mg/L during the incubation experiments.

Conclusions

The effect of alum alone and alum combined with particles on chlorophyll α, nutrients and turbidity removal was investigated using laboratory coagulation experiments. The results showed that alum coagulation was able to remove 85-90% of phosphate and chlorophyll α from Anabaena sp. enriched lake water samples. The added particles did not substantially affect the removal of phosphate and chlorophyll α during combined alum and particle coagulation. However, the particles substantially increased the floc settling kinetics. The combined treatment with alum and particles also increased the resistance of the sediment to disturbance by mixing.

The results showed that ballasted alum coagulation using natural minerals can increase the floc settling kinetics and prevent the resuspension of the settled flocs.

The PAC with higher basicity showed higher phosphate and chlorophyll α removal during coagulation. Phoslock showed high phosphate adsorption capacity. Combined use of PAC and phoslock increased the phosphate and chlorophyll α removal efficiencies. Simultaneous use both of PAC and phoslock also enhanced the settling kinetics of flocs. The results of long-term incubation experiment showed that alum coagulation alone was not effective at preventing the release of phosphate. Capping the sediments with natural minerals and phoslock reduced the phosphate release during long-term incubation.

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Information Transfer Program Introduction

The SDWRI Information Transfer Program includes public outreach; steering committee representation and leading involvement in the Big Sioux Water Festival hosting about 1,100 fourth grade students; interactions with extension agents and local, state and federal agencies; participation and presentations at regional and national conferences; youth education, adult education and university student training and education. Publications, such as pamphlets, educational materials, reports and peer-reviewed journal entries are made available in paper format and electronically through the Institute's website and are designed to support the mission of the Institute.

South Dakota Water Resources Institute FY2016 Information Transfer Program

Basic Information

Title:	South Dakota Water Resources Institute FY2016 Information Transfer Program
Project Number:	2016SD262B
Start Date:	3/1/2016
End Date:	2/28/2017
Funding Source:	104B
Congressional District:	SD-001
Research Category:	Biological Sciences
Focus Category:	Education, Management and Planning, Conservation
Descriptors:	None
Principal Investigators:	Van Kelley, Scott Cortus

Publications

There are no publications.

PUBLIC OUTREACH

Public outreach and dissemination of research results are cornerstones of the South Dakota Water Resources Institute's (SDWRI) Information Transfer Program. The Institute distributes information through a variety of outlets, including interactive information via the Internet, pamphlets and reports, direct personal communication, hands-on demonstrations and through presentations and discussions at meetings, symposia, and conferences. These outlets are described below.

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The SDWRI website is accessible through https://www.sdstate.edu/water-resources-institute. We are currently in the process of updating the SDWRI website. The website will contain information relating to water resources, current and past research projects, reference materials, and extension publications. One feature of the SDWRI website will allow users access to links such as publications and on-line tools to help diagnose and treat many water quality problems. The "Research Projects" section of the SDWRI website will contain past and present research projects, highlighting the Institute's commitment to improving water quality.

Conference Proceedings

Akhavan Bloorchian, Azadeh; Ahiablame, Laurent; Zhou, Jianpeng; Osouli, Abdolreza. (2016). Modeling BMP and Vegetative Cover Performance for Highway Stormwater Runoff Reduction. Procedia Engineering 145, 274 – 280. The 2016 International Conference on Sustainable Design, Engineering and Construction, May 18-20, Arizona State University, Tempe, AZ.

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McDaniel, Rachel. 2016. E. coli fate and transport in South Dakota waters. In: 2016 Eastern South Dakota Water Conference Abstract Book. Oral presentation, Brookings, SD.

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McDaniel, Rachel. 2016. E. coli transport in South Dakota streams. In: 2016 UCOWR/NIWR Annual Water Resources Conference – Conference Proceedings. UCOWR/NIWR. Poster presentation, Pensacola Beach, FL

Conference Papers or Posters Presented, Invited Lectures

Thapa, U., Ahiablame, L., Trooien, T., Hay, C. Kjaersgaard, J., Hua, G. (2016). Combined Treatment of Nitrogen and Phosphorus from Subsurface Drainage Using Low-cost Industrial By-products and Woodchip Bioreactors. Poster presented at Soy100, March 10, Brookings, SDSU, SD.

Ahiablame, L., Hay, C., Sahani, A., Kringen, D. (2016). Drainage management practices to improve water quality in eastern South Dakota. Poster presented at the North Central Region Water Network Spring 2016 Conference, March 21-23, 2016, Lincoln, NE.

Thapa, U., Ahiablame, L., Kringen. D., Hua, G., Hay, C., Kjaersgaard, J., Trooien, T. (2016). Using Denitrification Bioreactors and Phosphate Adsorption Media to Remove Nutrients from Agricultural Subsurface Drainage. Poster Presented at the 2016 Student Water Conference, March 24-25, Stillwater, OK. [Travel Grant Awarded by Oklahoma State University].

Singh, S., Ahiablame, L., Hay, C. (2016). Modeling the effects of conservation practices on drainage flow and Nitrate-N loads in the Western U.S. Corn Belt. Oral presentation at 2016 Student Water Conference, March 24th-25th, Stillwater, OK [Travel Grant Awarded by Oklahoma State University].

Paul, M., Ahiablame, L., Rajib, M. A. (2016). Spatial and temporal evaluation of hydrological response to climate and land use change in South Dakota watersheds. Oral Presentation at 2016 Student Water Conference, March 24-25, Stillwater, OK.

Sahani, A., Ahiablame, L., Hay, C., (2016). Hydrologic and Water Quality impacts of Drainage Management Strategies in Eastern South Dakota. Oral Presentation at 2016 Student Water Conference, March 24-25, Stillwater, OK. [Awarded travel grant of $500 by Oklahoma State University].

Sahani, A., Ahiablame, L., Hay, C., (2016). Impacts of Drainage Water Management on field-scale hydrology and water quality in Eastern South Dakota. Poster presentation at 2016 Annual Transforming Drainage meeting, March 29-31, West Lafayette, IN.

Singh, S., Ahiablame, L., Hay, C., (2016), Modeling the effects of conservation practices on drainage flow and Nitrate-N loads in the Western U.S. corn belt. Poster presentation at 2016 Gamma Sigma Delta poster competition, South Dakota State University, April 4th, Brookings, SD.

Thapa, U., Ahiablame, L., Kringen. D., Hua, G., Hay, C., Kjaersgaard, J., Trooien, T. (2016). Using Denitrification Bioreactors and Phosphate Adsorption Media to Remove Nutrients from Agricultural Subsurface Drainage. Poster Presented at 2016 Gamma Sigma Delta poster competition, South Dakota State University, April 4th, Brookings, SD.

Sahani, A., Ahiablame, L., Hay, C., (2016). Hydrologic and Water Quality impacts of Drainage Management Strategies in Eastern South Dakota. Oral Presentation at 2016 Western South Dakota Hydrology Meeting, April 7-8, Rapid City, SD.

Paul, M., Ahiablame, L., Rajib, M. A. (2016). Impacts of Land Use and Climate Change on Hydrological Processes in James River Watershed. Oral Presentation at 2016 Western South Dakota Hydrology Meeting, April 7-8, Rapid City, SD.

Hong, J., Ahiablame, L., Lim, K. J., (2016). Impacts of Grassland Conversion on Hydrology and Water Quality in the Bad River Watershed, South Dakota. Poster presented at The 2016 Western South Dakota Hydrology Conference, April 7th – 8th, Rapid City, SD.

Boger, A., Ahiablame, L., Beck, D., (2016). Vegetative Best Management Practices For Roadway Runoff. Poster presented at Western South Dakota Hydrology Conference, April 7th – 8th, Rapid City, SD [2nd place, student poster contest].

Singh, S., Ahiablame, L., Hay, C., (2016), Evaluation of integrated drainage water and agricultural management strategies for water quality protection. Poster presentation at Western South Dakota Hydrology Conference, April 7th-8th, Rapid City, SD.

Thapa, U., Ahiablame, L., Kringen. D., Hua, G., Hay, C., Kjaersgaard, J., Trooien, T. (2016). Using Denitrifiaction Bioreactors and Phosphate Adsorption Media to Remove Nutrients from Agricultural Subsurface Drainage. Poster Presented at the 2016 Western South Dakota Hydrology Meeting, April 7-8, Rapid City, SD.

J. Strock., L. Ahiablame, B. Dalzell, A. Garcia y Garcia, C. Hay, J. Kjaersgaard, J. Magner, G. Sands, T. Trooien and L. Zhang. (2016). Quantifying hydrologic impacts of drainage under corn production systems. Invited Presentation at The14th Semi-annual Watershed Professionals Networking and Learning day, April 21, Redwood Falls, MN.

Frankenberger, J., B. Reinhart, E. Kladivko, L. Bowling, B. Engel, L. Prokopy, M. Helmers, L. Abendroth, G. Chighladze, J. Strock, D. Jaynes, K. Nelson, M. Youssef, L. Brown, B. Sohngen, X. Jia, L. Ahiablame. (2016). Multi-state Project to Advance Hydrologic Storage in Tile-Drained Landscapes for Improved Water Quality. Poster presentation at the Hypoxia Task Force Spring Public Meeting, April 26, St. Louis, MO.

Thapa, U., Ahiablame, L., Hua, G., Trooien, T. (2016). Nitrogen and Phosphorus Removal from Subsurface Drainage Using Woodchip Bioreactors and Low-cost Filters. Oral Presentation at the 2016 MWAOAC Annual meeting, April 25-27, Sioux Falls, SD.

Singh, S., Gonzalez A., Kjaersagaard, J., Trooien, T., Ahiablame, L., Kumar, S. (2016). Response of Soil Nutrients and Water Quality to Winter Manure Application from small agricultural watersheds in South Dakota. Oral presentation at Research Day at South Dakota State University, April 28, 2016, Brookings, SD.

J. Strock., L. Ahiablame, B. Dalzell, A. Garcia y Garcia, C. Hay, J. Kjaersgaard, J. Magner, G. Sands, T. Trooien and L. Zhang. (2016). Quantifying hydrologic impacts of drainage under corn production systems. Department of Soil, Water & Climate Spring Seminar Series. University of Minnesota, May 4, St. Paul, MN.

Akhavan Bloorchian, Azadeh; Ahiablame, Laurent; Zhou, Jianpeng; Osouli, Abdolreza. (2016). Performance Evaluation of Combined Linear BMPs for Reducing Runoff from Highways in Urban Area. Proceeding paper in The 2016 EWRI World Environmental & Water Resources Congress, May 22-26, West Palm Beach, FL
.
Sahani, A., Thapa, U., Ahiablame, L., Hay, C. (2016). Demonstration of conservation drainage strategies to conserve water quality in Eastern South Dakota. Poster presentation at 6th Annual All About Science Fair, June 11, Sioux Falls, SD.

Bloorchian, A.A., Shakya, R., Ahiablame, L., Zhou, J., Osouli. R. (2016). Use of Retention Pond for Storm Runoff Control- A Modeling Approach. Poster presented at the 2016 UCOWR/NIWR Annual Conference, June 21-23, Pensacola, FL.

Mosase, E., Ahiablame, L., Srinivasan, R. (2016) Blue and Green water modeling in the Limpopo River Basin. Oral presentation at UCOWR 2016 Annual meeting, June 21-23, Pensacola, FL.

Akhavan Bloorchian, Azadeh; Zhou, Jianpeng; Osouli, Abdolreza; Ahiablame, Laurent; Grinter, Mark, (2016). Modeling Low impact development (LID) for controlling highway stormwater runoff. Presented The 2016 EWRI International Low Impact Development Conference, August 29-31, Portland, ME.

Paul, M., Ahiablame L., Rajib, A.M. (2016). Hydrologic and Water Quality Impacts of Climate and Land Use Changes in James River watershed. Poster Presented at the 2016 ASABE Annual International Meeting, July 17-20, Orlando, FL.

Mosase, E., Ahiablame, L., Srinivasan, R. (2016). Quantification of blue and green water resources in the Limpopo River Basin using Earth Observation data and SWAT model. Oral presentation at ASABE 2016 Annual International Meeting, July 17 -21, Orlando, FL.

Sahani, A., Ahiablame, L., Hay, C., (2016). Impacts of Drainage Water Management on field-scale hydrology and water quality in Eastern South Dakota. Oral presentation at ASABE 2016 Annual International Meeting, July 17 -21, Orlando, FL.

Sellner, B., Hua, G., Ahiablame, L., Trooien, T., Hay, C., Kjaersgaard, J. (2016). Evaluating Filter Materials for Phosphate Adsorption from Agricultural Subsurface Drainage. Oral presentation at ASABE 2016 Annual International Meeting, July 17 -21, Orlando, FL.

Boger, A., Ahiablame, L., Beck, D., (2016). Vegetative Best Management Practices For Roadway Runoff. Oral Presentation at Eastern South Dakota Water Conference, October 27th 2016, Brookings, SD [1st place, student oral presentation contest].

Thapa, U., Ahiablame, L., Hua, G., Trooien, T., Hay, C., Kjaersgaard, J. (2016). Phosphorus Removal from Subsurface Drainage Using Low-cost Filters. Poster Presentation at the 10th International Drainage Symposium, September 6-9 2016, Minneapolis, MN.

Thapa, U., Ahiablame, L., Trooien, T., Hay, C., Kjaersgaard, J. (2016). Denitrification Bioreactors to Support Conservation Drainage in Eastern South Dakota. Oral Presentation at the 10th International Drainage Symposium, September 6-9 2016, Minneapolis, MN.

Thapa, U., Ahiablame, L., Hua, G., Hay, C., Kjaersgaard, J., Trooien, T. (2016). Using Denitrification Bioreactors and Phosphate Adsorption Media for Water Quality Conservation in Subsurface Drainage Systems in Eastern South Dakota. Abstract and Oral Presentation at Eastern South Dakota Water Conference, October 27th 2016, Brookings, SD [2nd place, student oral presentation contest].

Hong, J., Ahiablame, L., Mosase, E., (2016). Assessing Hydrologic and Water Quality Impacts of Grassland in Skunk Creek Watershed. Oral Presentation at Eastern South Dakota Water Conference, October 27th, Brookings, SD.

Sellner, B, Guanghui Hua, U Thapa, L Ahiablame, TP Trooien, CH Hay, and J Kjaersgaard. 2016. Absorption of phosphate from agricultural subsurface drainage using natural minerals and industrial byproducts. Presentation at the Oklahoma State Student Water Conference. March 2016.

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Sahani, A., Ahiablame, L., Hay, C., (2016). Assessment of the Impacts of Subsurface Drainage and Agronomic Conservation Practices on Hydrology and Water Quality. Oral Presentation at the 10th International Drainage Symposium, September 6-9, Minneapolis, MN.

Ahiablame, L., Sahani, A., Hay, C., (2016). Demonstration of Drainage Management Practices in Eastern South Dakota. Poster Presentation at the 10th International Drainage Symposium, September 6-9, Minneapolis, MN.

Sahani, A., Ahiablame, L., Hay, C., (2016). Demonstration of Drainage Management Practices in Eastern South Dakota. Oral Presentation at Eastern South Dakota Water Conference, October 27th, Brookings, SD.

Ahiablame, L. (2016). March 3, Invited Speaker, Drainage at Southeast Research Farm Beresford, South Dakota, 4R Workshop, Denver, CO.

Ahiablame, L., Shakya, R. (2016). March 7-9, Invited Speaker, Watershed-Scale Evaluation of Flood Reduction Effects of LID Practices, 2nd Biennial Great Plains LID Research and Innovation Symposium , Omaha, NE.

Singh, S., Ahiablame, L., Hay, C. (2016). March 29th-30th, Modeling the effects of conservation practices on drainage flow and Nitrate-N loads in the Western U.S. Corn Belt. Poster presentation at 2016 NCERA-ADMS Task Force Annual Meeting, Purdue University, West Lafayette, IN.

Ahiablame, L., Hay, C., Sahani, A. (2016). April 7-8, Drainage Management Practices to improve water quality in Eastern South Dakota. Poster Presentation at 2016 Western South Dakota Hydrology Meeting, Rapid City, SD.

Ahiablame, L., Hua, G., Hay, C., Kjaersgaard, J., Trooien, T. (2016). June 21-23, Nutrient Removal in Subsurface Drainage Using Denitrification Bioreactors and Filter Materials for Phosphate Adsorption. Oral presentation at the 2016 UCOWR/NIWR Annual Conference, Pensacola, FL.

Ahiablame, L., Mosase, E., Srinivasan, R. (2016). June 21-23, Spatial and Temporal Variations of Hydroclimatic Variables in the Semi-Arid Southern Africa. Oral presentation at the 2016 UCOWR/NIWR Annual Conference, Pensacola, FL.

Ahiablame, L. (2016). June 13, Invited Speaker, Integration of Computer Modeling and Field Observations to Support Sustainable Management of Water Resources, US Geological Survey, Reston, VA.

Ahiablame L., Rajib, A.M., Paul, M. (2016). July 17-20, Modeling the effects of future land use change on water quality under multiple scenarios: A case study of low-input

agriculture with hay/pasture production. Oral Presentation at the 2016 ASABE Annual International Meeting, Orlando, FL.
Ahiablame L., Shakya, R. (2016). July 17-20, Watershed-scale evaluation of flood reduction effects of low impact development. Oral Presentation at the 2016 ASABE Annual International Meeting, Orlando, FL.

Reyes-Gonzalez, A, TP Trooien, CH Hay, and J Kjaersgaard. 2016. Comparison of crop reference evapotranspiration estimated by automated weather station and measured with an atmometer. Poster presentation to the South Dakota Hydrology Conference, Rapid City, SD. April 2016.

Reyes-Gonzalez, A, TP Trooien, J Kjaersgaard, CH Hay, and L Ahiablame. Comparison of Actual Crop Evapotranspiration Estimated from Remote Sensing-based METRIC Model and Measured with Atmometers. Poster presented to the 2016 ESDWC.

Singh, S, A Gonzalez, J Kjaersgaard, TP Trooien, L Ahiablame, and Kumar. Impacts of winter manure application to soil nutrients and water quality from small agricultural watersheds. Poster presented to the 2016 ESDWC.

Peer-Reviewed Publications

Paul, M., Rajib, A., Ahiablame, L. (2016). Spatial and temporal evaluation of hydrological response to climate and land use change in three South Dakota watersheds. Journal of the American Water Resources Association, 53, 69–88.

Rajib, A., Ahiablame, L., Paul, M. (2016). Modeling the effects of future land use change on water quality under multiple scenarios: A case study of low-input agriculture with hay/pasture production. Sustainability of Water Quality and Ecology, 8, 50-66.

Extension Articles

Kringen, David. 2016. Immobilizing Nitrogen through the Use of Cover Crops. SDSU iGrow Article.

Ostrem, DT, TP Trooien, and CH Hay. 2016. Chapter 33: Irrigating Corn in South Dakota. In Clay, D.E., C.G. Carlson, S.A. Clay, and E. Byamukama (eds). iGrow Corn: Best Management Practices. South Dakota State University.

Hay, CH and TP Trooien. 2016. Chapter 30: Managing High Water Tables in Corn Production. In Clay, D.E., Carlson, C.G. Clay, S.A., and E. Byamukama (eds). iGrow corn: Corn Best Management Practices. South Dakota State University.

Hall, RG, TP Trooien, DP Todey, and KD Reitsma. 2016. Appendix B: Seasonal Hazards – Frost, Hail, Drought and Flood. In Clay, D.E., C.G. Carlson, S.A. Clay, and E. Byamukama (eds.). iGrow Corn: Best Management Practices. South Dakota State University.

Kringen, David. 2016. The South Dakota Wetland Exchange. SDSU iGrow Article.

Kringen, David. 2016. New Technology for an Old Problem. SDSU iGrow Article.

Kringen, David. 2016. Sustainability in the Loess Hills of Minnehaha County. SDSU iGrow Article.

Kringen, David. 2016. Water Conservation and Efficiency During Times of Drought. SDSU iGrow Article.

Kringen, David. 2016. Saturated Buffers for Drainage Water Treatment in SD. SDSU iGrow Article.

Kringen, David. 2017. Conservation Stewardship Program: FY 2017 application due Feb. 3. SDSU iGrow Article.

Other Media

Kringen, David. Nutrient capture from various livestock feeding systems. SDSU iGrow Radio May 4, 2016

Kringen, David. Water management research. SDSU iGrow Radio July 27, 2016.

David Kringen. 2016 Ag Phd Field Day. 2016. Print.

David Kringen. North American Manure Expo to SD in 2018. 2016. Print.

AGENCY INTERACTIONS

SDWRI staff and affiliates served on several technical committees and boards, including:

- Member of the AmericaView Board of Directors
- Steering Committee for the National eXtension Conference
- Steering Committee for the Big Sioux Water Festival

Several other local, state, and federal agencies conduct cooperative research with SDWRI or contribute funding for research. Feedback to these agencies is often given in the form of reports and presentations at state meetings, service through committees and local boards, and public informational meetings for non-point source and research projects.

YOUTH EDUCATION

Non-point source pollution contributes to the loss of beneficial uses in many impaired water bodies in South Dakota. An important part of reducing non-point pollution is modifying the behavior of people living in watersheds through education. Programs designed to educate youth about how their activities affect water are important because attitudes regarding pollution and the human activities that cause it are formed early in life. For these reasons, Youth Education is an important component of SD WRI's Information Transfer Program.

Big Sioux Water Festival

Water Festivals provide an opportunity for fourth grade students to learn about water. SDWRI personnel were part of the organizing committee for the 2016 Big Sioux Water Festival held on May 10, 2016 with about 1100 fourth grade students from eastern South Dakota participating. SDWRI was responsible for coordination of volunteers and helpers, and co-coordinating the exhibit hall.

ADULT EDUCATION

DakotaFest

As part of SDWRI's outreach to the agricultural community, WRI affiliates host a booth at DakotaFest, a three-day agricultural fair held in August each year near Mitchell, SD, which each draws approximately 30,000 people. Personnel field a variety of questions concerning water quality and current research for farm and ranch families.

Eastern South Dakota Water Conference

SDWRI was a sponsor of the annual Eastern South Dakota Water Conference. This event was held on October 27, 2016 at the University Student Union on the SDSU campus. The conference covered the latest strategies and research for water managers and water users in the Northern Great Plains through oral and poster presentations. Presenters from several Great Plains states including South Dakota, North Dakota, Iowa, and Minnesota were in attendance. The theme for the 2016 conference was entitled "The Economics of Water Quality" where strategies and costs associated with water quality improvements were explored.

The event attracted attendees from academia; students; agriculture interest groups; local, state, and federal government agencies; and other concerned stakeholders. Attendance at the 2016 event consisted of approximately 100 paid registrants with an additional 140 SDSU students that joined the conference as class schedule allowed. Topics covered during the conference gave participants a better understanding of the current focuses of concern, research, and management of regional water resources; including the fate & transport of E. coli in SD waters, soil health, and drainage water management.

SDWRI staff and affiliates additionally participated in and presented at several regional and national meetings and conferences, including:

Conference Name	Organizing Organization	Location	Date
Western SD Hydrology Conference	USGS	Rapid City, SD	Apr 7-8, 2016
Eastern South Dakota Water Conference	EDWDD, SDSU, SD WRI, USGS, and SDSU WEERC	Brookings, SD	Oct 27, 2016
2016 UCOWR Conference	UCOWR, NIWR	Pensacola, FL	Jun 21-23, 2016
North Central Region Water Network Conference	NCRWN, UNL	Lincoln, NE	Mar 21-23, 2016
Southeast Research Farm Field Day	SD Extension Service	Beresford, SD	Jul 12, 2016
Ag PhD Field Day	Hefty Seed	Baltic, SD	Jul 28, 2016

USGS Summer Intern Program

None.

Student Support					
Category	Section 104 Base Grant	Section 104 NCGP Award	NIWR-USGS Internship	Supplemental Awards	Total
Undergraduate	2	0	0	0	2
Masters	4	2	0	0	6
Ph.D.	0	0	0	0	0
Post-Doc.	0	0	0	0	0
Total	6	2	0	0	8

Notable Awards and Achievements