Iowa Water Center
Annual Technical Report
FY 2005

Introduction

The Iowa Water Center is a multi-campus and multi-organizational center focusing on research, teaching, and outreach program activities. This center, previously known as the Iowa State Water Resources Research Institute (ISWRRI), was renamed the Iowa Water Center in 2006 and came under a new director, Dr. Richard Cruse, on 1 April 2006. The Center will continue the mission of ISWRRI which has been to develop statewide linkages between universities and public and private sectors and to promote education, research, and information transfer on water resources and water quality issues in Iowa.

Research has repeatedly illustrated the value and necessity of quality water in this state. Iowa is one of the nations leading agricultural producers, if not the leader, with a vast majority of its land under row crop production. Non-point source degradation of water quality is well recognized as a leading cause of Iowa water quality impairment as well as being a major contributor to water quality problems beyond the state border. Additional challenges exist, such as outdated sewer and septic systems and aging tile drainage systems. The water resources of this state will undergo additional quality and quantity stress with the developing bioeconomy. Large amounts of water are being withdrawn from subsurface aquifers and discharged on the surface by ethanol processing plants. Lignocellulose conversion technology will likely result in increased water use, removal of crop residue from the land surface, increased applications of fertilizers, and accelerated rates of soil erosion and water runoff. The need for an active and successful Iowa Water Center is even greater today than it has been in the past.

This year the governor and the state legislature of Iowa devoted $18 million of new money to water quality issues in Iowa at a time when the state budget is stressed. This clearly illustrates the public recognition of waters importance to this state. The Iowa Water Center is building on this and other commitments in the state. This coming year, a new strategic plan will be developed, a team-building symposium linking university and agency personnel will be held, a new web page is being constructed, and scientific and Center management advisory teams are being established. Activities surrounding research, outreach, and teaching will continue, and a reevaluation of focus areas for these activities is underway.

Research Program

Identification of Relationships Between Soil Phosphorus and Phosphorus Loss Through Tile Drainage to Improve the Subsurface Drainage Component of the Iowa Phosphorus Index

Basic Information

Title:	Identification of Relationships Between Soil Phosphorus and Phosphorus Loss Through Tile Drainage to Improve the Subsurface Drainage Component of the Iowa Phosphorus Index
Project Number:	2004IA62B
Start Date:	3/1/2004
End Date:	2/28/2006
Funding Source:	104B
Congressional District:	3rd
Research Category:	Water Quality
Focus Category:	Water Quality, Non Point Pollution, Nutrients
Descriptors:	phosphorus, subsurface drainage, water quality, environmental soil testing
Principal Investigators:	Antonio P. P. Mallarino, Matthew J. Helmers, Ramesh Kanwar

Publication

Identification of Relationships Between Soil Phosphorus and Phosphorus Loss through Tile Drainage to Improve the Subsurface Drainage Component of the Iowa Phosphorus Index

Antonio P. Mallarino, Rameshwar S. Kanwar, Matt J. Helmers

Problem and Research Objectives

This project is at the end of its second and last year. The problem addressed by the study and objectives were explained in the original proposal, did not change during its two years, and only a brief summary is included here.

Many questions related to the impact of current P management practices on P-related water quality are being asked by the public, government agencies in charge of nutrient regulations, producers, and researchers. Excess P often is delivered from agricultural fields to water resources because of inappropriate fertilizer or manure management or the need to dispose of manure at a low cost. Management guidelines and regulations are being established based on a P risk assessment tool often referred to as the P index. Although a P index has been developed for Iowa and many other states, gaps and insufficient information about some processes have created a great deal of uncertainty for some P index components. This project focuses on excess dissolved P loss through subsurface drainage that can occur when soil P concentration increases. Although many studies have monitored P concentration in tile drainage, few (and none in Iowa) have studied relationships between soil-test P and P loss through tiles. Soil-test P is affected by P fertilization, manure application, crop production, and several management practices. Although results of agronomic soil P tests are currently used in P indices, environmental P tests have been proposed as an alternative to these to measure P in soil and runoff water. There is little information concerning correlations of P extracted by either agronomic or routine tests and P loss through subsurface drainage. Limited information led the team that developed the Iowa P index to include approximate estimates of relationships between soil-test P and P loss in its subsurface drainage component.

The overall goal of the project is to establish relationships between soil P measured by various tests, fertilizer and manure P management, and P loss through subsurface drainage. The work is an interdisciplinary effort that uses existing facilities and inter-departmental cooperation to achieve objectives at a low cost. Specific objectives are:

1. Study the impact of fertilizer and manure applications on soil P measured with routine agronomic soil tests and environmental soil test methods that emphasize an assessment of potential P losses to water supplies.
2. Establish relationships between soil-test values and P concentrations in subsurface tile drainage for selected manure/fertilizer management systems.
3. Develop equations that can be included in future revisions of the soil P factor of the subsurface drainage component of the Iowa P index.

Methodology

The methods used in the study follow those planned and explained in detail by the original proposal. Briefly, soil and tile drainage samples are collected from plots of three long-term field experiments that include replicated manure application treatments or a combination of nutrient and cropping systems treatments. All plots have a tile drainage collection system with automatic water sampling devices. One experiment at the Northeast Research Farm (near Nashua) evaluates six cropping (tilled or no-till) and liquid swine manure/fertilizer management systems for corn-soybean rotations managed with chisel-plow tillage except for one system. The systems compare N and P fertilization according to crop needs (two systems varying only in N fertilizer application method), manure according to the N needs of the corn (in two systems, one with manure applied in spring for no-till and one with manure applied in the fall for chisel-plow tillage) and supplemental P fertilization as needed for soybean, manure according to the P needs of the corn plus supplemental N as needed to apply the same total N rate as the previous treatments, and manure according to the N need of corn and estimated N removal by soybean. The manure is always injected. Two other experiments are conducted at the Agronomy and Agricultural Engineering Research Farm (near Ames). One experiment includes three rates of poultry manure (egg layers) compared with equivalent N fertilizer rates for the corn in corn-soybean rotations managed with chisel-plow tillage. The other experiment evaluates injected liquid swine manure at three rates for corn in corn soybean rotations managed with chisel-plow tillage and broadcast liquid swine manure at one rate for continuous corn also managed with chisel-plow tillage. The total manure-N applied for corn in these two experiments ranges from 100 to 300 lb N/acre, which encompasses the range of 100 to 150 lb N/acre recommended for corn after soybean in Iowa, and at the same time applies manure P at approximately 70 to 250% of the P maintenance rate recommended for the 2-year corn-soybean rotation.

Principal Findings and Significance

Treatment Effects on Soil P

Soil samples were collected from a depth of 0 to 6 inches as planned and their chemical analyses have been completed. All deep soil samples planned in the original proposal could not be collected as a result of a significant budget reduction, although samples were collected in 2004 from plots of two trials. Considering soil samples collected in 2003 that were analyzed for the project and those collected in 2004 and 2005, more than 600 soil samples were analyzed for P and pH. The preliminary results of the soil analyses are summarized in the following points.

1. The treatments applied to these experiments resulted in large differences in soil-test P values for the 6-inch top layer of soil. The results of routine soil P tests (Bray-1, Mehlich-3, and Olsen) indicated that soil-test P remained near optimum for corn and soybean production in plots that received fertilizer or manure P at rates that applied P near the rates suggested by current P management guidelines in Iowa to maintain soil-test P. The optimum soil-test interpretation class for corn and soybean ranges from 16 to 20 ppm by Bray and Mehlich-3 tests and 11 to 14

ppm by the Olsen test, all with a colorimetric determination of extracted P. Soil-test P was lower in plots receiving lower P application rates, although no treatment resulted in soil-test P in the Very Low class. However, manure rates that applied two to three times the N required by corn or the N required by corn plus N removed by soybean significantly increased soil-test P values to values as high as six times the Optimum class (up to 140 ppm by the Bray-1 or Mehlich-3 tests and higher than 80 ppm by the Olsen test). Obviously, this soil P build-up was the result of P applications since the experiments were established in the late 1990s and not exclusively from P applied during the duration of this project.

2. Manure of fertilizer P application at rates that maintained soil-test P near optimum values for crops did not affect soil P concentration at depths deeper than 6 inches as measured in two experiments. Higher manure P rates increased only slightly P concentrations in the 6- to 12-inch layer of soil but did not significantly change subsoil P concentrations to a depth of 12 to 36 inches. The finding that high P applications did not increase subsoil P below a depth of 12 inches strongly suggests that although some P movement down the profile may have existed (as tile P concentrations indicate as discussed below), it was small and not detectable over a period of a few years. Furthermore, the small P increase in the 6–12 inches layer seldom was statistically significant and was observed only for plots that received the highest manure P rates, such as manure for corn at rates two to three times its N requirement or when manure also was for soybean at N-removal rates.

3. Chemical analyses of samples and study correlations between soil P measured by agronomic and environmental tests (soil bioavailable P or water-extractable P) are not completed, but the available data indicate high correlation between all P tests. Although, as expected and is known, the amounts of measured soil P varied greatly among P tests, all tests detected the effects of P applications similarly and were highly correlated. Because these results were observed across long-term plots receiving fertilizer P, swine manure, and poultry manure we also conclude that all these tests evaluate similarly effects of these P sources on soil P accumulation.

Relationships BetweenTile Drainage P and Treatments or Soil P Concentrations
Data on P concentration of tile drainage are available for the years 2003 and 2004. Although tile drainage was also sampled as planned from all experiments and plots in 2005, the samples were not analyzed yet. Collaborators of the Department of Agriculture and Biosystems Engineering collected these samples and were planning to analyze them with support from other funding sources. Unfortunately, funding was short due to significant budget cuts and the pace of water sample analysis was slowed significantly. Moreover, analyses of samples collected in 2004 from one site were completed only recently and results have not been carefully studied for possible outliers are available at this time. However, the data available for other years allow for the following preliminary conclusions at this time.

1. Weighted-average orthophosphate P concentration in tile water ranged from values less than 10 ppb (parts per billion) to 220 ppb, although values seldom were higher than 100 ppb. The highest value corresponded to some plots of treatments in which manure was applied every year to both corn and soybean or only for corn at rates that applied amounts twice or higher than the N needed. The lowest values corresponded to treatments in which either fertilizer or manure P was applied at rates similar to or lower than those needed to maintain soil-test P for the 2-year corn-soybean rotation according to current recommendations in Iowa. Interestingly, P concentrations and loads in tile drainage also were very low, most of the time not significantly different from the P-based treatments, when swine or poultry manure was applied only for corn at N-based rates or even about 50% higher. These results match results of soil P tests in that swine manure application at rates that supply N needs of corn approximately supply the P needs of the two crops of the rotation and do not result in large soil P build-up or elevated P concentrations in tile drainage. Application of N-based poultry manure for corn did result in soil P build-up but not significantly higher tile P concentrations compared with lower rates. Calculations of annual P loads even for the treatments that resulted in the highest P concentrations in tile drainage (such as manure applied every year to both crops or only for corn at rates that applied amounts twice or higher than the N needed) indicated very low P loss, less than 0.1 lb P/acre/year.

2. Available relationships between P concentrations in tile drainage or P loads and soil P measured by various methods across sites and years indicate very low P loss (< 20 ppb) up to soil-test P levels five to six times as large as levels required by crops. At higher soil-test P levels the P concentrations in tile drainage were very variable ranging from the lowest values observed at the lowest soil-test P levels up to the highest values observed in the study and there was no clear relationship between the two measurements. These results coincide with results observed in previous years in Iowa. We hoped that continued application of swine manure rates higher than N-based rates for corn and N-based or higher poultry manure rates would result in a faster soil P build-up and increased P concentration in tile drainage. Perhaps the results for samples collected in 2005 will show this effect. The results available at this time do indicate, however, that very high soil P concentrations, higher than levels measured in these experiments, are needed in Iowa soils to increase P loss in tile drainage to levels significantly or consistently higher than the low background levels observed at low soil P levels or those required for optimum crop production.

Implications of Results for the Subsurface Drainage Component of the Iowa P Index

The preliminary results have significant implications for the subsurface component of the Iowa P Index. One important implication is that the relationships between P concentrations in soil and tile drainage did not suggest a change to the current two-class factor of the component (for Bray soil P, for example, the factor indicates low or high P loss at and below or above a threshold value of 100 ppm). We hoped that increased soil P build-up in these trials would result in higher P concentration and loss in tile drainage and would allow for fitting either an exponential or grafted polynomial (discontinue) regression model including a near-plateau at low soil P levels and an increasing linear trend at higher concentrations. This type of relationship sometimes has been identified by research in other states at much higher soil P levels and significantly different soil types. The available results indicated that much, much higher soil P levels (probably higher than 200 to 300 ppm by the Bray-1 method) might be needed to observe significantly high P loss through tile drainage and clear exponential or linear increases in P loss as soil P concentration increases.

Another important preliminary result of this study is a confirmation that P losses through tile drainage are much, much smaller than P loss through surface runoff or soil erosion, which is recognized and accounted for the by the Iowa P Index. Unless soil P is extremely high, much higher than about 150 ppm by the Bray soil test, for example, efforts at reducing P loss from Iowa fields will be more effective by focusing on management practices affecting P loss with runoff and soil erosion.

No articles or publications have been prepared because only partial and preliminary results are available at this time. However, the project and results have been presented and discussed at six field days or indoor meetings during the last two years, an annual meeting of the American Society of Agronomy, a P conference with Iowa and Illinois technical personnel of NRCS and state agencies, a meeting of the Iowa Nutrient Task Force, and a meeting of the Environmental Protection Commission of the Iowa legislature.

Vegetative Filter Education and Assessment in the State of Iowa

Basic Information

Title:	Vegetative Filter Education and Assessment in the State of Iowa
Project Number:	2004IA63B
Start Date:	8/1/2004
End Date:	2/28/2006
Funding Source:	104B
Congressional District:	4th
Research Category:	Water Quality
Focus Category:	Education, Non Point Pollution, Surface Water
Descriptors:	vegetative filters, water quality, nutrients, sediment, assessment, education, impaired water bodies, surface runoff, TMDL
Principal Investigators:	Steven K. Mickelson, Kapil Arora, Matthew J. Helmers

Publication

Vegetative Filter Education and Assessment in the State of Iowa

Steven K. Mickelson, Matt Helmers, Kapil Arora

Problem and Research Objectives

At present, quality of water is the largest issue globally, with concerns for the alarming death rate of aquatic organisms, human health hazards, and aesthetic beauty of the world's famous water bodies. In order to combat pollution resulting from diffuse sources, the U.S. government is taking considerable measures towards mitigation by means of employing Best Management Practices (BMPs). Vegetated filter strips (VFS) is one of the best management practices that helps reduce the transport of these substances to receiving waters. Reduction of sediment and nutrients in surface runoff to rivers and lakes is important to Iowa's initiative to reduce the number of impaired water bodies. VFS help to reduce the deterioration of the surface waters through retention of the sediments and nutrients from surface runoff from agricultural fields. VFS have been shown to be most effective for shallow, uniform surface runoff conditions, but it has also been shown that in case of heavy overland flows, the flow concentrates and only a portion of the vegetative filters proves to be effective in sediment and nutrient retention in the filter. Determining the most important design considerations for VFS is important for maximizing the water quality benefits for the VFS. Therefore, the on-site assessment of existing vegetative filter strips to examine and document critical design criteria is highly significant. It is also important to extend the findings and knowledge found in the field assessment to the stakeholders and upcoming generation so that they implement these designs and protect the environment. Therefore, the following objectives are considered important to be achieved in relevance to the existing scenario. The objectives include:

1. Identification of VFS sites for in-field data collection and assessment.
2. Development of an assessment tool for evaluating the effectiveness of VFS using past and current research literature findings.
3. Determination of the effectiveness of VFS by visual field observation and validation by flow mapping procedures in ArcGIS 9.
4. Comparison of the area ratios and percentage of flow along each stream segment at various resolutions (5X5, 10X10, 20X20 and 30X30) for different sizes of the survey data sets.
5. Comparison of the flow routing for USGS 7.5 Quad Angle values and spatial analysis of the elevation data at resolution of 30X30.
6. Education of grade school, junior high, and high school students on VFS performance and surface water runoff issues related to water quality and biodiversity.

Methodology

Using data from past and current research projects on VFS, an on-site assessment tool has been developed to evaluate the performance of Iowa's VFS in key impaired water bodies. The major component of this study constitutes assisting grade school, junior high, and high school students in evaluating current VFS in an impaired watershed(s) close to their location. The Rock Creek Watershed next to Newton, Iowa was selected by the research team for this purpose (shown in Figure 1). This site was selected since it was in the extension Agricultural Engineer's (Kapil Arora) region and due to the ease of collaboration with the local NRCS county office and the local educators.

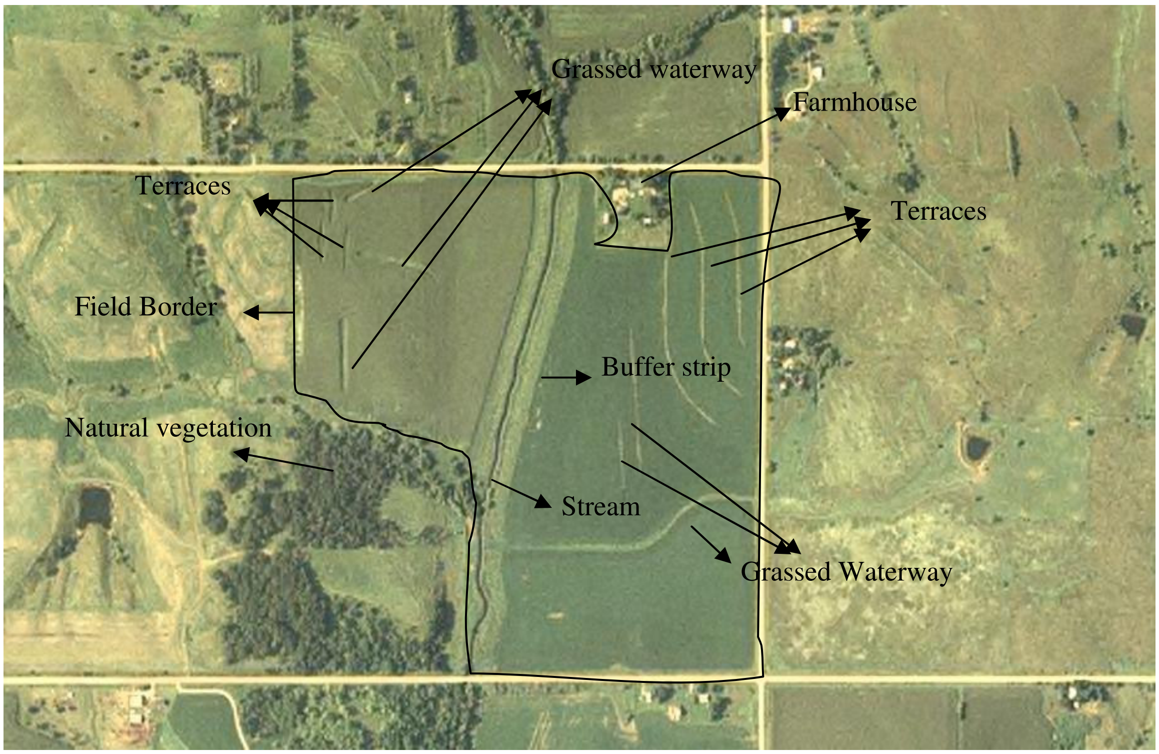

Fig. 1. Rock Creek Watershed field layout

The team visited the Rock Creek Lake Watershed in November 2004 and conducted a training session for high school students. The students made an efficient site evaluation with the guidance of the field staff and research team. The site evaluation included the parameters like vegetative filter length and width, type of vegetation, uniformity/space distribution of vegetation, rill and gully erosion evidence, etc. Validation checks were made at each subwatershed site within the Rock Creek watershed by the research team. Through visual observation and in-field surveying, it was found that the buffer strips are ineffective in sediment and chemical retention, as the surface runoff does not flow through the VFS. Owing to the presence of undulations in the field, the flow leads towards the natural vegetation instead of draining into buffer strips which led to significant growth in this region. Some traces of sedimentation at the buffer edge were seen which makes it evident that there were times when runoff reached the buffer strips, but from the topographic observations, it is possible only in the case of a larger rainfall event. Another training session was conducted for grade school students in September 2005. Grinnell High School FFA, Newton Senior High School FFA, and Structure of Intellect (SOI) at Thomas Jefferson Elementary in Newton showed keen interest and participation in the project. These training sessions helped the students to better understand the processes and impacts of nutrients from agriculture on water quality and the impact of sediment accumulation on aquatic life in lakes and streams. Using the past research, we have come up with an extension bulletin which includes an assessment form for the VFS site. This document would facilitate both qualitative and quantitative assessment at a given VFS site. Qualitative measurements on the form will have supportive rubrics attached that more adequately describe various levels of quality related to a specific measurement. Please find this document as an attachment to the report.

The team also met NRCS staff in September 2004 in regard to acquisition of the data where it was decided to conduct the site surveys in relation to the drainage and the filter area. As a result of this meeting, several potential sites were identified based on their vegetation growth, time (years since establishment), filter width, and drainage area served. Out of these sites, land owners of three sites agreed to partner in this project and provide access to the sites for the team to carry out their evaluations. These three sites were surveyed by the NRCS team in Fall 2005. In the meantime, the literature was reviewed to gain knowledge about the sediment trapping and pesticide retention with the help of VFS.

The elevation data for the sites was recorded throughout the field by the local NRCS office staff using Global Positioning System (GPS) Real Time Kinematics (RTK) equipment. Geographic Information Systems (GIS) was employed for validation of the visual observation regarding the flow accumulation and outlet points in the field. This software combines the site evaluation data of each point on the field into layers of information at that point on the map of the watershed to give a better understanding of the runoff hydrology at that point. The technique of Digital Elevation Model (DEM) helps obtain the digital representation of the topographic surfaces as a regular grid of spot heights. In our case, elevation data for the field was used to obtain the flow routing in ArcGIS 9 and validate our visual observations regarding the effectiveness of the buffer strip. Elevations at equal and uniform intervals in the watershed were interpolated from the collected data (which was at unequal spacing) through a method called Kriging.

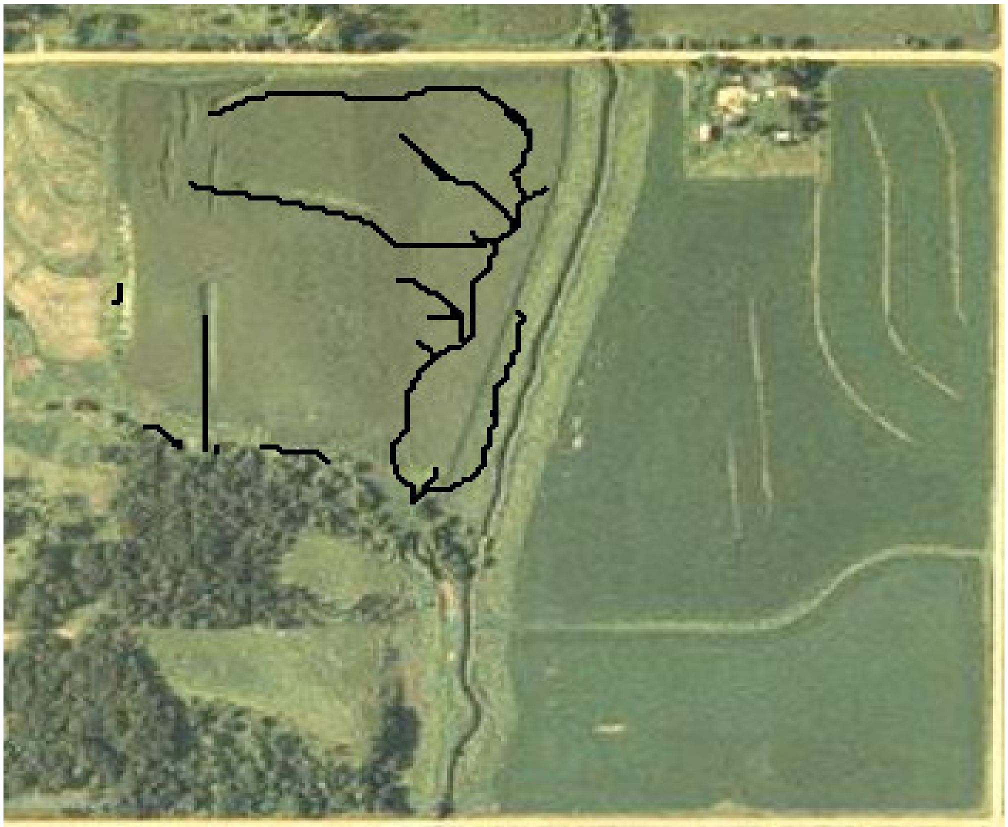

Fig. 2. Stream network of a subwatershed in Rock Creek watershed field

DEM helped identify the sinks in the drainage area, generate flow accumulation and drainage/stream network of the watershed for the collected data through ArcView 3.3. The stream network delineated by the software for a subwatershed of Rock Creek watershed field, shown in Figure 2 above, was found in congruence with the visual observation, which lies as a major objective of this study.

As per the objectives, stream network for four different resolutions of DEM, namely 5X5, 10X10, 20X20 and 30X30 was also delineated. The stream network for a subwatershed has been delineated for all four datasets at a resolution of 5X5 of the collected data. These layers were laid on top of one another, as shown in Figure 3 below. We can clearly see the difference in the stream network delineation for four datasets at the same resolution. We aim to quantify these differences in terms of area ratios and percent of flow along each stream segment for four different sizes of the dataset.

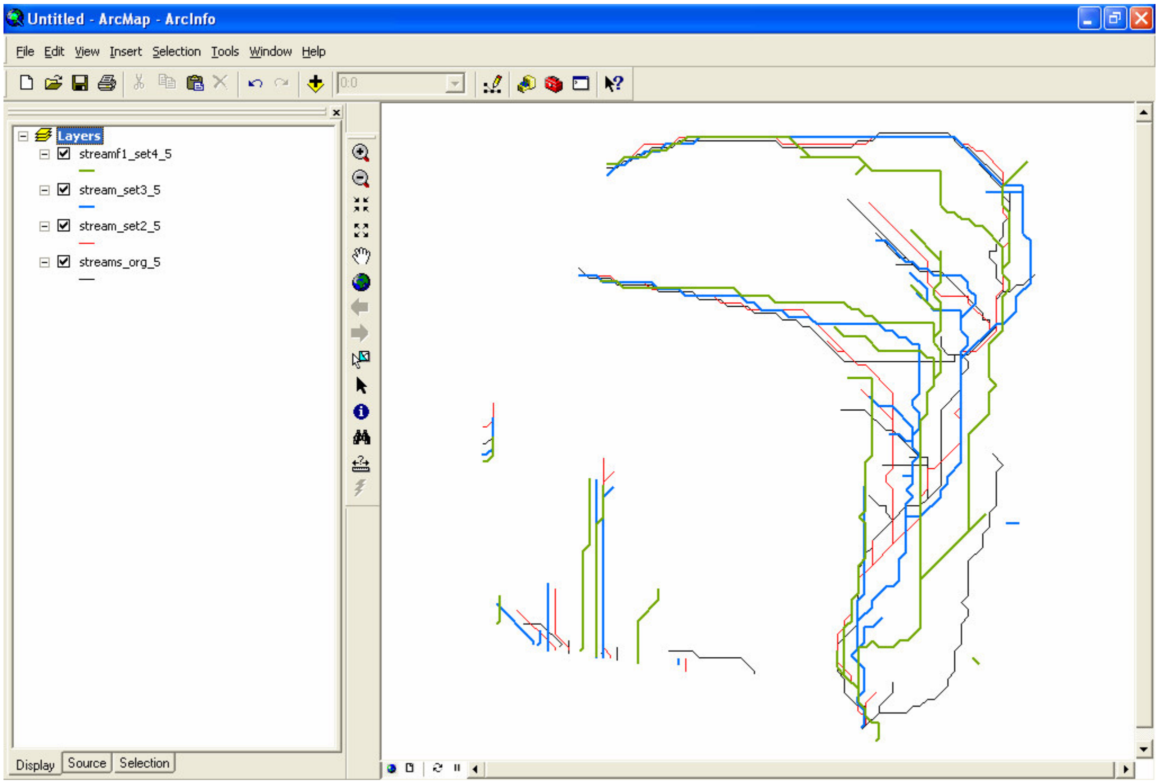

Fig. 3. Stream network for all four datasets at 5X5 resolution

Another objective aimed at comparison of the flow routing for USGS 7.5 Quad Angle values and spatial analysis of the elevation data of the field at a resolution of 30X30. The DEM of USGS 7.5 Quad Angle 30X30 values were extracted from Iowa data found at ftp://gis.iastate.edu and the stream network was delineated in ArcView 3.3. Flow routing for this showed well-organized streams heading towards the buffer before draining into streams. This was laid on top of 30X30 resolution of the collected data, as shown in Figure 4. Apparently, delineated stream network for collected data was far different from that of USGS data at resolution of 30X30. But it was seen that the delineated stream network for dataset 4, the smallest data size, at resolution of 30X30 was close to that of USGS 30X30, as shown in Figure 4. This clearly implies that the number of data points for USGS 30X30 lies close to the number of data points in Set 4.

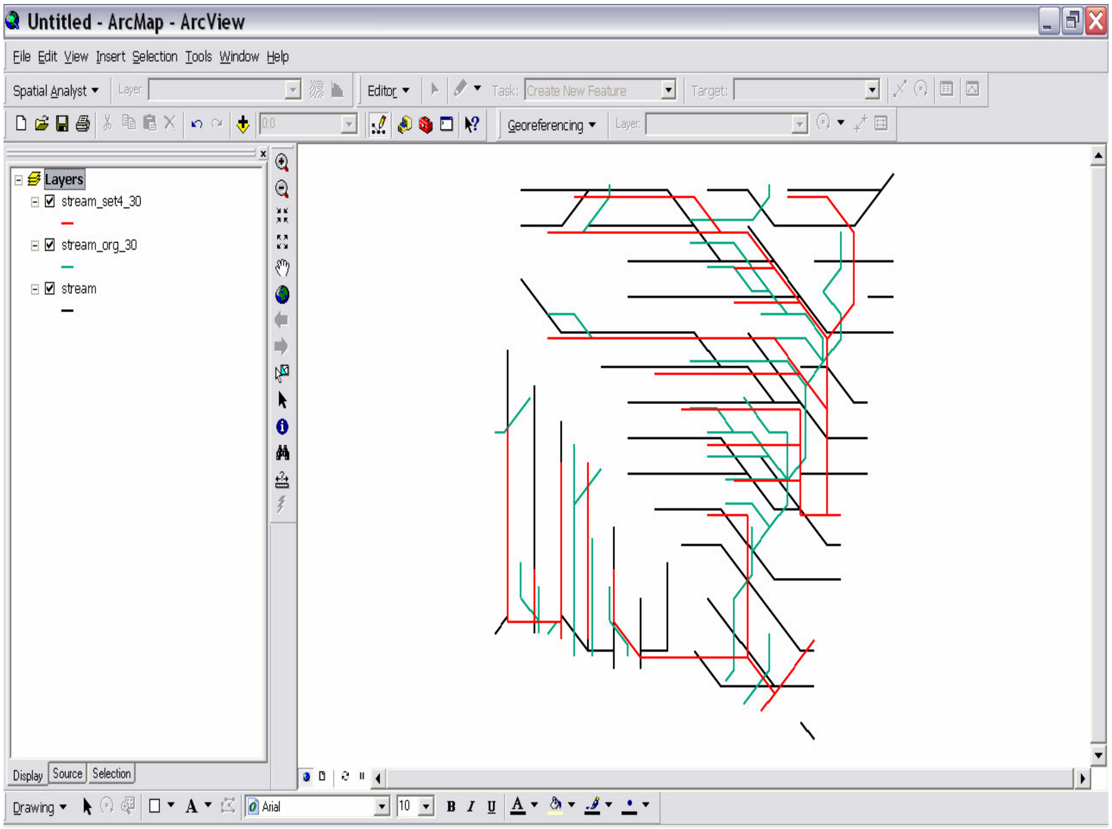

Fig. 4. Overlay of USGS, Dataset 1 and Dataset 4 stream network at 30X30 resolution

The differences in the flow accumulation at various resolutions (5X5, 10X10, 20X20 and 30X30) of the different survey data sets for the field will be quantified in terms of area ratios and percentage of flow along each stream segment. From the spatial analysis of the different datasets of one subwatershed, it has been concluded that the smallest dataset is least accurate in terms of flow routing, with results being far different from visual observation. In addition to this it was seen that the resolution of 5X5 gives good estimate even with the smallest dataset.

Principal Findings and Significance

The majority of the year 2004 has been spent on reviewing the key literature related to VFS for the purpose of designing the best on-site VFS assessment tool. Significant time was also given to developing the assessment tool, choosing the correct watershed, collecting in-field survey data, and setting up collaboration with the Newton educators. The following section summarized the findings from the literature review.

The transport of the sediments and the range of applied agrochemicals from the agricultural fields into the surface water bodies is one of the major environmental threats. This transport is a result of heavy rainfalls or huge amounts of overland flow. Controlling the amount of the agrochemicals and the sediments available for potential loss to the environment by planting close growing vegetation or tall, stiff grasses is a significant management practice that helps reduce the transport of these substances to receiving waters. VFS are the bands of planted or indigenous vegetation situated down slope of cropland or animal production facilities to filter nutrients, sediment, organics, pathogens, and pesticides from agricultural runoff before it reaches a water system (Dillaha et al., 1989). These VFS offer important advantages where runoff concentrates. These have been considered to be effective in slowing down the runoff velocity and filtering sediment. VFS prove as an impediment to the movement of the suspended material in the runoff, hence promoting the settling of the suspended solids which are sediments and applied chemicals. Therefore, it is important to assess the effectiveness of the VFS in removal of the sediments and nutrients from the runoff. The effectiveness of the strip is dependent on the width, types of vegetation, age, level of development, and many more factors. Following is the key literature regarding various aspects related to the use of VFS, such as hydraulic characteristics, sediment/pesticide removal, and their effectiveness/ineffectiveness owing to various factors.

Hydrology & Characteristics of VFS

There are studies that quantify the effectiveness of VFS in terms of length, slope, and hydraulic characteristics. Here are some of those studies which proved to be helpful in better understanding about VFS.

Length. Gharabhaghi et al. (2001) studied the variations in sediment removal efficiency with variation in flowpath of vegetative filter strips. Lengths of 2.44, 4.88, 9.67 and 19.52 m were considered for 1.22 m wide field with a slope 5.1–7.2%. From 58 runs of experiments and 348 runoff samples, it was concluded that the first five meters play a significant role in removal of the suspended solids and aggregates greater than 40 microns in runoff. It was found that the performance of the VFS doesn't increase by appreciable margin by increasing the flowpath length beyond 10 m. High turbulence keeps finer particles in suspension which makes it difficult to remove them from runoff. However, the study pertains to the fact that infiltration is the only mechanism that helps in the removal of the smaller size sediments. The vegetative filter strip model VFSMOD was calibrated and validated using the observed data from the field experiments. The model was observed to possess high accuracy in predicting the sediment removal efficiencies of the vegetated filter strips.

Lee et al. (2003) conducted a study to determine the effectiveness of a multi-species riparian buffer in removing non-point source (NPS) pollutants carried by cropland runoff. The experimentation involved installation of three plots where each of the cropland source areas was matched with no buffer, switchgrass buffer (7 m), and a switchgrass/woody plant buffer (16.3 m). This study is a perfect example of functional differences between the long and short buffers. It is attributed to sediment trapping efficiency figures as high as >92% and >97% after passage of runoff from switchgrass and switchgrass/woody buffer respectively. It was concluded that the switchgrass is an effective measure for coarse particles unlike switchgrass/woody buffer, which is more suitable for finer particles. Sediment transported through no buffer was 13 times more than that from switchgrass/woody buffer. Sediment size distribution was found to be another significant factor that determines the performance of VFS. In this case, more than 90% of the sediment in the surface runoff from the buffered plots was in the <0.05 mm size fraction. During infiltration of nutrients, suspended fine soil particles with adsorbed chemicals also enter the profile, thus decreasing the surface runoff and sediment transport capacity. The results, therefore, indicated that the selection of buffer vegetation should take care of problems and conditions of the site.

M. Abu Zreig et al. (2004) conducted twenty field experiments to study sediment removal in VFS with variations in filter length, filter slope, and type of vegetation. Experiments were conducted with incoming sediment load of 2700 mg/l on filter lengths of 2, 5, 10, and 15 m, slopes of 2.3 and 5%, and three different types of vegetation. It was concluded that the length of the filter was the most important factor affecting the VFS sediment trapping efficiency. It was observed that increase in length of filter beyond 10m didn't increase the sediment trapping efficiency. Rather, an exponentially decreasing trend between sediment trapping efficiency and length beyond 10 m was seen. The sediment trapping efficiency was observed to increase with decrease in inflow rates and decrease in soilwater content of soil due to enhanced infiltration. Although vegetation has a secondary effect on sediment trapping efficiency, greater vegetation densities resulted in lesser erosion and lesser transport capacity of the runoff, eventually leading to greater settling of the sediments.

Hydraulic Characteristics. Infiltration is the underlying mechanism responsible for the trapping of the suspended solids and applied chemicals carried by the runoff. Infiltration is the downward entry of the water into the soil profile. Gharabhagi et al. (2001) stands by the fact that infiltration is the sole mechanism that helps the removal of smaller sized sediments. The vegetative cover helps in reducing the velocity of incoming runoff and increases the residence time, the time for water to infiltrate. Consequently, ponding occurs at the upstream end of the filter and some of the sediments and suspended solids get filtered out as the water flows through the filter and settles on the top of the filter. Stem diameter, density, stiffness, and hedge width affected the depth of ponding (Meyer et al., 1995).

Ree et al. (1949) studied the hydraulic characteristics of vegetation. It was observed that Manning's coefficient 'n' decreases as the submerged grass in a waterway bends over owing to high flow rates. Due to the bending of the grasses, there is a decrease both in the turbulence creating ability of the stems and area blocking effect. Whereas, in the case of the non-submerged channel, grass stands erect which helps retard the flow in a better way. Ree indicated that grass remains erect until submergence is complete. In other words, the study concluded that the non-submerged conditions form an ideal case for maximum flow retardation and the minimum sediment transport capacity.

Van Dijk et al. (1996) identified the use of grass vegetation as grass hedges, grass strips, buffer zones, and grass channels as an effective measure to reduce sediment transport to surface waters. This study discusses the retention of water and sediment in each of these field arrangements and concluded that the underlying mechanism is the same for all the arrangements, i.e., infiltration and sedimentation. The experiment was conducted so as to derive the comparative results regarding the sediment trapping efficiency of grasses with two different ages and agricultural management practice. It was seen that the older grass was much more effective in reducing erosion than the younger grass which was credited to frequent mowing activities. Certain differences in the water retention of two grasses were observed which were attributed to difference in the grass density at two locations. Sediment trapping efficiency of grass filters of length 1, 4-5 and10m was recorded as 50-60, 60-90, and 90-99% respectively.

M. Abu Zreig (2001) studied the factors affecting VFS performance using computer simulation by means of VFSMOD. Length of the filter was seen to have the greatest effect on sediment trapping. It was observed that sediment trapping efficiency decreases exponentially beyond 10 m. Greater vegetation densities, and therefore a greater Manning's roughness coefficient 'n' resulted in greater contact time between the runoff and vegetation resulting in less erosive power and less transport capacity of the runoff and therefore, greater trapping efficiency. Also, the effect of length of the filter was seen in combination with 'n' and it was concluded that practicality of situation lies with the fact that higher trapping efficiencies can be achieved by increasing the length of the filter than maintaining a good vegetation cover. Filter performance also depends on the size of the incoming sediment. Trapping efficiencies of 0% and 47% for clay particles over filter lengths of 1 m and 15 m respectively were observed through experimentation which implies that smaller sized particles take longer length to filter out. Different soil types have different saturated hydraulic conductivities, which have a significant effect on trapping efficiency by effecting infiltration.

Sediment & Nutrient Removal. Young et al. (1980) performed a 2-year study to evaluate the effectiveness of VFS to remove the pollutants in runoff from livestock feedlots under simulated rainfall conditions. The experiment was conducted on a field 111.25 m X 54.86 m and 6 VFS strips, each 4.06 m wide and 41.15 m long with 4% slope. Out of the length of 41.15 m, 13.72 laid within the feedlot. Cropped fields of corn, orchardgrass, sorghum-sudangrass, and oat plots were used to reduce the runoff, total solids, and nutrients. The results showed that the total nitrogen, NH_4-N, total Phosphorus, and PO_4-P were seen to have reduced by 84, 63, 83, and 76%, respectively. Suspended sediment was reduced by 86, 66, 82, and 75% for corn, orchardgrass, sorghum-sudangrass, and oats, respectively. It was seen that NO_3-N values rose, which was attributed to collection of NO_3-N by runoff from sorghum-sudangrass and oat plots. In case of corn the reductions in runoff, nutrient and suspended sediment were appreciably higher than reductions from other fields. This was credited to across the slope plantation of the corn. As the runoff passed through the vegetated buffer strips, a decrease in the indicator organisms in runoff was seen. In this case, a length of 36 m was seen to be sufficient enough to reduce nutrients, micro-organisms, and suspended solids in feedlot to acceptable levels.

Magette et al. (1989) experimented to study the effectiveness of VFS in nutrient and sediment removal. Urea-Ammonium-Nitrate, a source of N and broiler litter was applied to 22 m X 5.5 m field at the rate of 112 kg N/ha and 8.9 wet metric tons/ha, respectively. VFS of lengths 4.6 m and 10 m were employed in each set of experiment. The field soil was rich in P; therefore, no supplemental P was applied. This study assumed P movement to be dependent on total soluble solids (TSS) transport; whereas N can move in soluble form more freely. The results showed higher losses of P during UAN tests than broiler litter tests. This was attributed to the mulching effect of the litter, which eventually minimized the TSS losses. Losses of TN, TP, and TSS were seen to decrease by 0, 27, and 66%, respectively, with the use of VFS. This clearly indicated that VFS is not as effective in removing the nutrients from cropland runoff as in removing suspended soils.

Concentrated Flow. Concentrated flow or non-uniform distribution of flow limits the performance of VFS. In another study by Meyer et al., 1995, strips of tall, stiff grasses were planted across the slope in order to study their sediment trapping efficiency. It was observed that the practice of perpendicular plantation helped achieve higher trapping efficiencies by retarding the flow concentration. Flow concentration was seen to have a detrimental effect on the filtering effectiveness of the VFS. Experiments were conducted to analyze the effectiveness of the vegetative filter strips using transparent wall flumes and root boxes (grass boxes). Root boxes were placed in a pit such that the grass surface was leveled with the base of the flume. Sediment mixed with water was fed at the upper end of the flume, which passed the grass boxes. This experimentation setup helped us understand the hydrology involved in sediment removal. It was seen that the grasses retarded the flow, resulting in a hydraulic jump formation several meters upslope of the hedge, which apparently led to the deposition of the incoming sediment. The formation of the hydraulic jump and sediment deposition further helped the flow retardation and deepened the ponded flow. Sediment trapping resulted mostly from the upslope ponding due to grass hedges rather than by filtering action. It was concluded that the sediment trapping was primarily a result of sufficient settling time in the ponded flow and not because of the failure of sediment to pass through the voids in the grass. Results emphasize the effectiveness of stiff grasses as high as 80% for sand-sized sediment. This clearly implies that the trapping effectiveness largely depends upon the size distribution of the incoming sediment and we require longer path lengths for fine silt or clay-sized sediment.

Dosskey et al. (2002) conducted an assessment of the riparian buffers. It was seen that concentration of flow from agricultural fields considerably hampers the potential of the riparian buffers to remove pollutants. Concentration/non-uniform distribution of flow occurs when runoff meets only a small fraction of the gross area owing to factors like topography, flow rate, etc. The methodology employed four study farms for studying the impact of the flow on sediment trapping efficiency, evaluated with the help of a numerical model using regression equations based on the ratio of the buffer area to field runoff area. This model yielded trapping efficiencies of 99%, 67%, 59%, and 41% in contrast to 43%, 15%, 23%, and 34% for uniform and non-uniform flow conditions respectively, all other parameters held constant. It was noted that sediment trapping could only be improved by avoiding the concentrated flow, which is generally caused due to the deposition of the soils from channelization activities within the buffer zone.

Pesticide Retention

Pesticides can be applied in various ways, such as aerial spraying, incorporation or injection into the soil, or application in solution form. Similarly, these have various loss pathways such as volatilization or aerial drift, adsorption onto the soil particles, or degradation into simpler forms over a period of time. But the loss of pesticides as runoff to surface water bodies or leaching into groundwater profile is the one that is of major concern to environmentalists. The fate and transport of pesticides is largely dependent on their chemical properties, like solubility, persistence, etc. The pesticides that are highly soluble have a tendency to leach down to groundwater profile, while the ones which are highly volatile vaporize during application. There are many factors that influence the fate and transformation of the pesticides and have been described as follows:

Adsorption and Solubility. When a pesticide enters soil, some of it will stick to soil particles through a process called adsorption. Some of the pesticide will dissolve and mix with the water between soil particles. The active sites for sorption of the pesticide are mainly the clays and soil organic matter. As more water enters the soil through rain or irrigation, the adsorbed pesticide molecules may be detached from soil particles through a process called desorption. The solubility of a pesticide and its sorption on soil are inversely related, which means the more soluble the pesticide, the lesser the tendency to be adsorbed/sorbed. The pesticides that are highly soluble have an affinity to leach down through the soil to the groundwater and are referred to as *weakly adsorbed pesticides*. These can also be lost to surface waters due to high amounts of irrigation water or due to overland flow resulting from heavy rainfalls. The *strongly adsorbed pesticides* do not readily leach to the underground water but can be found bound to the soil particles. There is another type of pesticide called *moderately adsorbed*. Infiltration is the key process for retention by the buffer strips for moderately adsorbed pesticides (Arora et al., 1996). Furthermore, adsorption is also affected by various factors, as follows:

Partition Coefficient. One of the most useful indices for quantifying pesticide adsorption on soils is the partition coefficient (K_{oc}). The K_{oc} value is defined as the ratio of pesticide concentration in the adsorbed state and the solution phase. Thus, for a given amount of pesticide applied, a smaller K_{oc} value implies a greater concentration of pesticide in solution or, in other words, the more soluble the pesticide. Pesticides with small K_{oc} values are more likely to be leached compared to those with large K_{oc} value.

Persistence. Another most important factor in deciding the fate of the pesticides is persistence. This factor is commonly evaluated in terms of half-life, which is the time that it takes for a pesticide to reach half its concentration through degradation/transformation. Pesticides with longer half lives could be persistent. Pesticides are classified on the basis of their persistence:

- Non-persistent: 30 days or less
- Moderately persistent: Longer than 30 days but less than 100 days
- Persistent: Longer than 100 days

Vapor Pressure. Also, pesticides with high vapor pressures are generally not recommended for application. This is because the greater the vapor pressure, the greater is the fraction of the molecules that can escape the liquid by gaseous diffusion.

Soil Properties. Soil properties like hydraulic conductivity and organic matter content and structure are important factors to determine the fate of the pesticides. Coarse-textured soils have higher hydraulic conductivities than do fine-textured soils. The travel time of the dissolved pesticide is shorter in coarse-textured soils than in fine-textured because of the fine pores and slow permeabilities in fine-textured soils. Therefore, the chances for pesticides to leach down easily are greater in coarse-textured soils. Also, the high clay and organic matter content of fine-textured soils leads to greater sorption, thus making pesticides less susceptible to leaching in fine-textured soils. Soil structure is another factor that has a significant effect on the fate of the pesticides. Macropores or wide cracks established by earthworms or farm machinery operations help in the preferential movement of the pesticides through the soil profile to the underground water resources. In such cases, pesticides lose the opportunity to be adsorbed.

Site Conditions. From the study of Gilliam et al., 1993, it has been found that in the case of shallow vadose zone, which is prevalent mostly in humid areas, pesticides get lesser opportunity to get adsorbed. The nature of the underlying strata governs the direction and rate of chemical movement. If this stratum is a permeable layer, the leaching is much easier and the chemical generally follows in a vertical direction in contrast to hard pan or an impermeable stratum which would actually contribute to the lateral flow of shallow groundwater, hence polluting the surface waters. Sometimes cracks and fractures convey water rapidly. Warmer weather conditions accentuate the rate of chemical, biological, and physical processes involved in the fate of the pesticides such as microbial degradation, volatilization, etc.

Management Practices. The best management practices involve the use of site-specific and crop-specific pesticides. Amount and time of application needs are especially taken care of.

Baker and Laflen (1979) studied the combined effect of wheel track compaction and method of incorporation on runoff losses of herbicides, namely, propachlor, atrazine, and alachlor. A rainfall simulation study was carried out with 122 mm of rainfall on nine plots each 1.5 m X 9.1 m, Clarion sandy loam soil. The experiment was conducted both with surface applied/soil incorporated herbicide application and with/without wheel tracks to deduce results regarding the effect of two factors. The pesticide losses that were measured from plots with wheel tracks were about 3.7 times higher than those from plots without wheel tracks where the herbicides were applied to a soil surface. It was concluded that incorporation practice for herbicides is superior to surface application/broadcasting as the herbicide losses in surface applied plots were around 3.5 times larger than those from plots where herbicides were incorporated by disking.

Arora et al. (1996) carried out a study to investigate herbicide retention by VFS from runoff at the Swine Nutrition Center, Iowa State University for two years under natural rainfall conditions. Six VFS, 1.52 m wide X 20.12 m long downstream of 0.41 ha of source area were established with brome grass to study the performance of buffer strips in retaining the three herbicides, namely atrazine, metolachlor and cyanazine present in runoff. Also, the effect of drainage to buffer strip ratio of 15:1 and 30:1 on herbicide retention was another objective of the study. Herbicide concentrations associated with water was seen to fall in outflow than in inflow which indicates retention/adsorption by soil and plant surfaces. The average K (adsorption/partition coefficient) values were seen to be 22, 18, and 15 in later runoff events in contrast to 15, 10, and 8 in the first five events. Values ≥ 8 indicate higher herbicide concentrations as adsorbed to sediment than in solution. The results showed that herbicide concentration associated with sediment higher in outflow than inflow for metalachlor, unlike atrazine and cyanazine. This was accredited to the difference in the adsorption properties of these herbicides. Not an appreciable difference was seen in the percent retentions for different area ratios and was reasoned as the nature of moderately adsorbed herbicides, which follow similar processes of infiltration and interception-adsorption. Efficiencies of the studied buffer strips were seen to vary between 40 and 100%.

Patty et al. (1997) studied the effectiveness of buffer strips to remove pesticides, nitrate, and soluble phosphorus compounds from runoff water by conducting experiments on three research farms at Brittany, France with VFS of 6, 12, and 18 m length perpendicularly sown with rye. Pesticides Isoproturon, atrazine, Diflufenican, and lindane as pollutants were used. Isoproturon and atrazine are water-soluble and moderately adsorbed on soil in contrast to Diflufenican and lindane, which have very low solubility in water. The results showed that the pesticide losses depend on the time elapsed between the time of application and rainfall event. It is owing to the sorption of the pesticides onto the surface of organic matter, soil particles, etc which also adds to the sediment removal efficiency of the VFS. Direction of sowing was another factor that contributed towards the effectiveness of buffer strip and proved to be advantageous in removing the nutrient and sediment load in runoff. The results showed nitrate and soluble phosphorus losses reduction by 47–100% and 22–89% respectively.

Arora et al. (2003) aimed at determining the performance of vegetated buffer strips in reducing pesticide transport under simulated runoff conditions. Experiments were conducted on six *20.12m long X 1.52m wide* buffer strips to determine their retention efficiency for three pesticides of different adsorption properties, namely atrazine, metolachlor, and chlorpyrifos. In addition to trapping efficiency, the effect of area ratios, 15:1 and 30:1, on the pesticide retention of the buffer strips was evaluated. The results showed that sediment concentration in outflow was reduced by 60–80%. A combination of infiltration and sediment retention was observed as an active process of retention. The results showed lesser sediment adsorbed concentrations of chlorpyrifos in outflow than inflow, unlike atrazine and metolachlor. Chlorpyrifos is a strongly adsorbed pesticide, unlike atrazine and metolachlor, which are moderately adsorbed pesticides. This is explained by the fact that chlorpyrifos gets easily adsorbed onto heavier/larger particles which get trapped by VFS, while finer sediment particles are seen in outflow. These fine particles have larger specific areas, owing to which the sediment associated concentrations are found to be higher in outflow than inflow, which is the case here for atrazine and metolachlor. The effect of adsorbing properties on retention of these pesticides was clearly reflected in the results. Another noteworthy observation regarding the retention of pesticides was lower concentrations of atrazine and metolachlor in runoff outflow than inflow, unlike chlorpyrifos. This was attributed to the fact that most of the retention of moderately adsorbed pesticides, like atrazine and metolachlor, occurs through infiltration. On the other hand, pesticides like chlorpyrifos were trapped through sediment adsorption and retention by buffer strips. Although this study showed an insignificant trend of lower retention at the higher area ratio, there was no appreciable difference between the performances of buffer strips at two different area ratios.

Wu et al. (2003) conducted experiments in order to compare the effectiveness of switchgrass, tall fescue filter strips, and bare soil in removing the copper pesticide from runoff under simulated conditions. The experiment was conducted on artificially constructed beds, 0.9m X 2m and 3% slope with A-horizon of Bojac sandy loam soil. Lime and fertilizer, as a source of Cu, were packed in top 10 cm of beds. The soil used had 0.5ppm of exchangeable Cu. Two flow rates of 6 L and 2.7 L were used for this experiment. The results showed that the infiltrated amount of runoff was about 21% for no grass, 33% for switchgrass, 28% for tall fescue filter strips for 6.0 L flow rate in contrast to 77% for no grass, 97% for switchgrass, and 100% for tall fescue strips. It was found that at the slower flow rate, buffers could remove all of applied Cu, while the efficiency of removal was just 60% for the faster flow rate. The retention of Cu was attributed to the phenomenon of adsorption to soil. Thus, it implies that this metal has a very small potential for contaminating the groundwater. The copper adsorbed by the soil was calculated as the difference between the initial concentration and the equilibrium concentration. In the case of tall fescue grasses, results emphasized major Cu retention in the first one-third of the filter, which implies that relatively smaller filter lengths are required for tall fescue grasses. The study recommended use of two filter strips according to the flow rate, i.e., tall fescue is more suitable for removal in areas where runoff is not expected to travel at a fast flow rate.

Boyd et al. (2005) aimed at determination of the effectiveness of brome grass VFS for sediment and pesticide retention from subsurface drainage and runoff. The study conducted experiments with pesticides, namely atrazine, acetochlor, and chlorpyrifos in central Iowa under natural runoff conditions. Infiltration and adsorption of pesticide onto sediment particles were found to be predominant processes in retention. The results substantiate the similarities in the partitioning properties of acetochlor and atrazine, categorized as moderately adsorbed pesticides. Their fate is governed by infiltration of runoff as it was observed that the major portion of these pesticides moved within the water phase. Chlorpyrifos, a strongly adsorbed pesticide, unlike acetochlor and atrazine, was highly adsorbed to the sediment which resulted in its higher sediment retention by buffer strips. In addition to the study of pesticide retention, the effect of area ratios, 15:1 and 45:1, was studied on the pesticide retention in the buffer strips. It was also concluded that higher area ratios led to higher flow rates and easy saturation of the VFS, reducing its removal efficiency. Not a very significant difference was seen in the performance of buffer strips with difference in area ratios. The results showed considerable concentrations of moderately adsorbed pesticides in tile flow.

Geographic Information Systems

For a long time, scientists and engineers have studied the world as maps and models. But as time passed, a need arose for models beyond maps and globe that could also serve as tools of analysis. Geographic Information Systems(GIS) is one such sophisticated model that is capable of developing, using, visualizing, and analyzing the geospatial data. Various tasks that are performed using this software are:

- Input
- Manipulation
- Management
- Query & Analysis
- Visualization

Nowadays, many fields are employing GIS for data analysis, such as Natural Resources, Land Use Planning, Landscape Architecture, Transportation, Real Estate and Property Taxation, etc. The advantage of this model on other models is that a part of the data can be ripped from another for suitable analysis. In GIS, the geographic data is in the form of *layers*. For example, to study the world map, data would be in distinct layers of oceans, continents, countries, states, rivers, etc. For example, it is possible to study the geography of world and TMDL of the rivers in India separately in spite of them being one geospatial dataset. The object that a particular layer depicts is called a *feature,* having a set of attributes. Features have particular shapes and sizes. All the geographic objects in GIS can be represented as one of only three shapes—point, line, or polygon. Data in the form of these shapes is called *Vector* data. But geospatial data has properties like slope, temperature, and elevation which can't be represented as one of the above shapes. This type of data is represented in the form of *Raster* or, in other words, represented as surfaces. Data in raster form has numeric values rather than shapes. The numeric values represent the intensity of that particular property in geography. The higher the numeric value assigned to a particular point on the map, the higher would be the property, like temperature, slope, etc. The boundaries of rasters are depicted by squared cells of the same dimension, and each cell carries a numeric value. Every point on a GIS map is referred to in the form of (x, y) coordinates, which is relative to the origin of that particular coordinate system.

Some of the literature has been reviewed to emphasize the importance of GIS in the field of environment protection, planning, and management and also to enhance the understanding of various tools of GIS that are used for projects dealing with watershed analysis.

GIS and Non-Point Source Pollution. Subra and Water (1996) employed GIS modeling technique in the identification of areas contributing towards NPS pollution in a 20 x 20 mile section of Calcasieu River Basin, Southwest Louisiana. This study also quantified and prioritized the areas that were of importance in regard to water contamination through NPS pollution. ERDAS Imagine Spatial Modeler helped in the selection of layers that were of importance to this project, such as hydrography, distance to water, slope, and soil permeability. The watershed boundaries and data for the layers were digitized from sources like water quality management basins map (Water Resources, Louisiana), United States Geologic Survey DLG data, and General Soil maps (Department of Agriculture, USDA). USGS Digital Elevation model was not available; therefore, contours were digitized from 1:62,500 quad sheets. The maximum distance to water was set to 254 pixels. Soil permeability divided soils in 4 categories : poorly to moderately well drained soils, poorly drained soils, very poorly drained soils, and open water. It was concluded that major pollution was a result of industrial and commercial services. This paper recommends the use of GIS for setting up of industries and commercial facilities at apt locations and employing specific management practices in order to prevent pollution.

GIS and Stormwater Runoff Management. Sieker and Klein (1998) studied the case of water quality of Rummelsberg Lake, Berlin, Germany. Emissions from a nearby catchment called MHG, spreading on an area of 22 km^2 were seen to have a detrimental effect on the water quality of the lake. The soil of the catchment was of low infiltration capacity except a part where the soil was of high infiltration capacity but simultaneously high groundwater levels. Various measures like central/decentral stormwater treatment plants and their pros and cons with regard to factors like groundwater level, infiltration capacity, soil contamination, slope of ground, etc. were evaluated, but a large scale model known as KOSIM was found to be the best for simulation of the settling processes in pollution load transport. An arrangement of central and semi-central stormwater management measures were found to be working the best for the current situation in the watershed. GIS was used as a tool to explore the possibilities of enforcing decentralized stormwater treatment plant to this area.

GIS and Pesticide Contamination. Dabrowski et al. (2002) conducted sampling over a 3-yr period with pesticides, namely azinphos-methyl (AZP), chlorpyrifos (CPF), and endosulphan (END) in the Lourens River watershed, South Africa, spread on an area of 44 km^2, consisting of eight subwatersheds. The study employed a GIS-based runoff model to validate the results of pesticide contamination. All the activities occurring in the upstream areas cause pollution in tributaries that drains each of the subwatersheds and joins the main stream. Data for watershed boundary, land use patterns, slope and contours were digitized and converted to shapefiles for use as layers in ArcView 3.1 GIS. The advantage of using a GIS-based runoff model was that it could predict the contamination with consideration of catchment variables (i.e., slope of the land, soil type, etc.), pesticide properties (i.e., adsorption, solubility, etc.), etc. for each of the subcatchments, while any other mathematical model employed numerous variables at a time and is not that accurate in prediction. A positive correlation was seen between the predicted and observed values. Pesticide application in the months October to February in the growing area of 4 km^2 was considered responsible for the contamination of waters of Lourens River. It was concluded that the lack of best management practices in the watershed was one reason for pollution in the river.

GIS and Waste-water Management Planning. Apfel et al.(2004) presented a GIS planning tool for McHEnry County, Illinois. As this county lies northwest of Chicago, it is experiencing tremendous pressure of growth, hence leading to exploitation of natural resources like groundwater. Glacial activities in this region further added to it by disturbing the geophysical conditions in that region, like increasing the permeability of soil, creating expansive wetlands, etc. Therefore, the wastewater management and planning was a significant contemplation to help conserve the groundwater resources, for which GIS was used as a tool of planning. The study emphasizes that for an onsite risk assessment, what is necessary to protect the environment should be the guiding principle rather than the use of traditional technology, where GIS proves to be a helpful and useful effort. Parameters that were used as input to the project were soil types, wetlands inventory, municipal boundaries, municipal sewer service areas, transportation tracks, surface waters and groundwater aquifer maps. It was concluded that GIS is the best tool as it provides a visual format with efficient graphics to the map which is very useful for public settings. In addition to this, it was seen that resource information in GIS provided very efficient analysis.

The work is under progress.

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Rubric: Assessment tool for Vegetative Filter Strips

Design Parameters	*Excellent*	*Good to Fair*	*Poor*
a ***Ratio of drainage area to buffer area***	1:1–8:1	8:1–40:1	* > 40:1
b* ***Length of the filter (ft)	> 20	20	< 20
c, d ***Density of buffer vegetation (stems / m^2)(approx.)*** - Bermuda grass - Grass	Thick cover 9000 3600	Average cover 7100–3600 2900–1450	Sparse cover 1800 700
Soil Type	Sandy Loam	Loam	Clay Loam

* It forms the most viable situation practically, but the effectiveness has not been validated by research studies.
** Filter strip flow length shall be based on the field slope percent and length, and on the filter strip slope percent, erosion rate, amount and particle size distribution of sediment delivered to the filter strip, density and height of the filter strip vegetation, and runoff volume associated with erosion producing events. The quoted flow length is based on the recommended values for field slope area of 1–10%.

Design Parameters	*Excellent*	*Good to Fair*	*Poor*
a *Maintenance*	1) Frequent inspections 2) Absence of erosion channels. 3) A uniform vegetation cover. 4) Mow height of 6-inch, upright vegetation. 5) Good health of plants. 6) No evidence of unwanted trees, bushes, and noxious weeds. 7) No evidence of animal traffic.	1) Not very frequent number of inspections but, an instantaneous inspection after intense rains or long runoff events. 2) No evidence of gullies and rills, but small diversions paths or channels that are not very deep. 3) Good cover of vegetation with some patches of no or less cover. 4) Sward height a little more or less than 6 inches. Little evidence of bent over grasses at spots. 5) Diminutive yellow colored patches on vegetation. 6) Little evidence of weeds and unwanted plants. 7) Traffic with minimum damage due to grazing.	1) A few inspections in a year. 2) Evidence of rills and small channels that hinder the sheet flow. 3) Inadequate/sparse ground cover. 4) Uneven height of the grasses that necessitates mowing. Significant evidence of bent over grasses due to heavy runoff or vehicular traffic. 5) Unhealthy plants with broken, burnt and rotten leaves, brown in color. 6) Significant indication of weeds and unwanted bushes. 7) Evidence of livestock traffic, damage to vegetation due to overgrazing.

Design Parameters	*Excellent*	*Good to Fair*	*Poor*
e ***Wildlife Evidence***	1) No evidence of animal foot, grazing patches etc.	1) A few voids in cover that indicate grazing.	1) Large patches in cover that indicate grazing. 2) Evidence of wildlife traffic due to presence of animal hooves at many spots in vegetation. 3) Loss in filter width over years due to encroachment.
E ***Vehicular Traffic in VFS***	1) No damage to the VFS vegetation due to vehicular traffic. No evidences seen.	1) Little evidences seen due to loss of grasses along a path.	1) Evidence of diverted flow pattern because of vehicular traffic. 2) Established pathway seen in the VFS which indicates pedestrian or two-wheeler traffic.

Design Parameters	*Excellent*	*Good to Fair*	*Poor*
f *Type of flow*	*Sheet Flow* 1) Shallow, uniform flow all along the length of the filter.	Uniform flow for most of the distance but mild evidence of channel or gully formation.	*Concentrated flow* 1) Significant evidence of deep gullies and rills. 2) Tendency of runoff to flow into topographic swales before entry into buffers.
Number of concentrated passes	No concentrated flow passes observed.	Single or few concentrated flow passes seen.	Multiple concentrated flow passes seen along the width of the VFS.

Design Parameters	*Excellent*	*Good to Fair*	*Poor*
e *Drainage*	1) Efficient drainage along downstream VFS. 2) Even topography with no hills and depressions. 3) No inundation seen throughout the VFS. 4) Effective in sediment removal and nutrient reduction with efficient drainage.	1) A few, small height depressions and hills found in topography. 2) Little evidence of inundation with established drainageways.	1) Accumulation of surface runoff in natural drainage ways within fields before it reached the VFS. 2) Runoff from the drainage ways crossed the VFS, totally inundating the filters and rendering them ineffective for sediment and nutrient reduction. 3) Undesirable topography which hinders the proper drainage.

VEGETATIVE FILTER STRIP (VFS) ASSESSMENT FORM

Assessed By ______________________ Date _______________

Location of investigation (County / State) __________________________

Adjacent water body type __________________________

Quantitative Analysis :

1. Ratio of drainage area to VFS area ____________________
2. Length of the VFS (ft) ____________________
3. Slope of the drainage area upslope of VFS (%) ____________________
4. Density of vegetation in VFS (stems/m^2) ____________________
5. Slope of the VFS (%) ____________________
6. Filter Cover (Bare, Warm Season, Cool Season) ____________________

Qualitative Analysis :

	Poor	Good- Fair	Excellent
1. Ratio of drainage area to VFS area	☐	☐	☐
2. Length of the VFS (ft)	☐	☐	☐
3. Density of vegetation in VFS	☐	☐	☐
4. Soil type in VFS	☐	☐	☐
5. Maintenance of VFS	☐	☐	☐
6. Wildlife Evidence in VFS	☐	☐	☐
7. Vehicular Traffic in VFS	☐	☐	☐
8. Type of flow	☐	☐	☐
9. Number of concentrated passes	☐	☐	☐
10. Drainage	☐	☐	☐

NOTE: If you find more than 3 poor parameters on the VFS, then the VFS should be considered for modification in design.

Hydrologic Modeling of Subsurface Drainage for Predicting Drainage Outflow

Basic Information

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Principal Investigators:	Matthew J. Helmers

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1. Singh, R., M.J. Helmers, and Z. Qi, 2006. Calibration and validation of DRAINMOD to design subsurface drainage systems for Iowa's tile landscapes, Agricultural Water Management. (Accepted for publication 5/19/06)
2. Qi, Z., M. Helmers, and R. Singh, 2006. Evaluating a drainage model using soil hydraulic parameters derived from various methods. ASAE Meeting Paper No. 062318. St. Joseph, Mich.:ASAE.
3. Singh, R. and M.J. Helmers, 2006. Subsurface drainage and its management in the upper Midwest tile landscape, In Proceedings of the EWRI Congress, ASCE.
4. Helmers, M.J., P. Lawlor, J.L. Baker, S. Melvin, and D. Lemke, 2005. Temporal subsurface flow patterns from fifteen years in north-central Iowa. ASAE Meeting Paper No. 052234. St. Joseph, Mich.: ASAE.

Hydrologic Modeling of Subsurface Drainage for Predicting Drainage Outflow

Matthew J. Helmers

Problem and Research Objectives

Movement of water and nutrients through subsurface drainage systems is a concern in many midwestern agricultural watersheds, including the Des Moines Lobe of Iowa. Although subsurface drainage has its benefits—it improves the productivity of croplands and generally reduces surface water runoff—these systems result in a greater volume of subsurface drainage flow to downstream water bodies, thereby increasing nitrate-nitrogen movement to the same. In order to reduce excess water movement and nitrate-nitrogen movement in these watersheds, hydraulic modifications of drainage systems are being considered as water-quality management practices. At the Iowa Water Summit held at Iowa State University on November 24, 2003, three of the five work groups (Nonpoint Sources, Nutrients, and Impaired Water Restoration) identified the need for assessment and demonstration of hydrologic modifications as a new way of addressing water quality concerns, particularly nitrate-nitrogen leaching. Two hydrologic modifications commonly proposed are shallow drain tube installation and controlled drainage. Shallow drainage consists of placing conventional tile drains at shallow depths (e.g., at 24–30" rather than at 48–60"). Controlled drainage raises the outlet of the drainage system at certain times to raise the water table. These modifications to the drainage system are expected to have a direct effect on the volume of subsurface flow and nitrate-nitrogen concentrations and loading from subsurface flow.

However, to evaluate effectively the performance of tile-drained landscapes and potential impacts of modifications, water and nutrient outflow in the system must be accurately estimated or predicted under different scenarios. Use of hydrologic models affords one the opportunity to evaluate the impact of different management strategies on water quantity and quality in subsurface drainage systems; but in order to have confidence in the modeling results, the models should be calibrated and validated. Through calibration and validation the impact of parameters that affect drainage volume—specifically, soil hydraulic properties and climate conditions—can be better understood. With this information in hand, researchers gain confidence in the models' ability to predict subsurface flows and ultimately make use of them in management decisions.

Soil hydraulic parameters are required for running hydrological models. However, it is time-consuming and costly to obtain detailed soil hydraulic parameters in most cases. An alternative way to get these parameters is to use pedotransfer functions (PTFs). Based on artificial neural networks, ROSETTA (Schaap et al., 2001) conducts PTFs to derive van Genuchten soil hydraulic parameters from soil textural data only, combining them with bulk density, or combining soil textural data, bulk density with water content at one or two pressure points (33kPa or 1500kPa).

Van Genuchten soil hydraulic parameters for hydrological modeling could be obtained at different levels when using ROSETTA. One method is to find the soil name, the textural data and bulk density in the maps and tables included in a Soil Survey, then input them into ROSETTA. Another method is to analyze the particle size in the laboratory as well as organic matter content, then to calculate the bulk density and extrapolate θ_{33kPa}, $\theta_{1500kPa}$

by the formula and triangles offered by Rawls (1983) and Rawls and Brakensiek (1983), and input this information into ROSETTA. ROSETTA can use these two levels of information as raw materials and output the van Genuchten soil hydraulic parameters for each data set. Besides those two methods, calibrating the model using non-linear parameter estimation software (PEST) to optimize the initial input hydraulic parameters could be considered as the third level to obtain reliable van Genuchten parameters.

Controlled drainage raises the outlet of the drainage system at certain times to raise the water table and restrict outflow. These modifications have the potential to reduce subsurface drainage volumes, thereby decreasing the export of nutrients and other pollutants from agricultural landscapes. Studies have shown a reduction of 25 to 44% in subsurface drainage through shallow or controlled drainage practices (Evans et al., 1995; Cooke et al., 2002; Sands et al., 2003; Burchell et al., 2003; Fausey, 2004). Since the hydrology of tile landscapes change from one region to another region with variations in weather, soil, and crop cultivation, there is a need to investigate the impact of shallow and controlled drainage considering the local ecohydrological conditions. To better understand the performance of these systems under the climatic and soil conditions present in much of the upper Midwest, a first step is to understand drainage flow patterns over a range of climatic conditions. In addition, for integrating in-field management practices and downstream practices, an understanding of the temporal patterns of drainage flow is useful.

DRAINMOD (Skaggs, 1980), which has been widely applied to modeling the hydrological process of conventional and controlled drainage in the areas with relative high water table, includes the output of ROSETTA as an input of soil hydraulic information. The objectives of this study are to determine which level of soil information would be sufficient to use with DRAINMOD in predicting subsurface drainage volume and to evaluate controlled drainage in reducing drainage over an extended period (1945–2004), using the predicted outflow with a set of soil hydraulic parameters that was proved to be sufficient. This research has applicability in addressing the suitability of models for predicting subsurface drainage and the level of input data required to make accurate predictions. This research is focusing on the drainage outflow, with possible future research in this area to focus on the ability to predict nitrate-nitrogen leaching.

Methodology

Subsurface drainage volumes in five Webster soil plots for 14 years from 1990 to 2003 were simulated using DRAINMOD. The field experimental plots were located near Gilmore City, in Pocahontas County, IA. Drain tiles have been laid at a depth of 1.06 m with a spacing of 7.6 m. The flow rates of all 78 plots and onsite weather have been monitored in the period from April to November since 1989.

DRAINMOD inputs are aggregated into 4 groups: soil, weather, crops, and drainage design. Three different methods were used in preparing the soil hydraulic parameters for the model: 1) determining the soil texture and bulk density(BD) data from the Iowa Soil Properties And Interpretations Database (ISPAID, Version 7.1, 2004), then inputting them into a pedotransfer model (ROSETTA) to determine soil hydraulic properties

(SP_1); 2) analyzing the soil texture and organic matter (OM) content in the laboratory, calculating the BD through the formula offered by Rawls (1983) and extrapolating θ33kpa, and θ1500kpa from the triangle offered by Rawls and Brakensiek (1983), then inputting them into ROSETTA (SP_2); and 3) model calibration with PEST to optimize the soil hydraulic parameters from SP_1 using observed monthly drainage volume from 1990 to 1993 (SP_3).

The detailed description of weather, crop, drainage system design, and monthly ET factors is included in Singh et al (2006). DRAINMOD was run 3 times with soil hydraulic parameters derived from the 3 levels of method and the same weather, crops, and drainage design information. Daily, monthly, and yearly subsurface drainage volumes were collected from DRAINMOD output files. Four statistical measures, as shown in the following, were employed to evaluate the fit of the predicted drainage with the observed data.

$$\text{Root Mean Square Error, } RMSE = \sqrt{\frac{1}{N}\sum_{i=1}^{N}\left(P_i - O_i\right)^2} \tag{2}$$

$$\text{Coefficient of Mass Residual, } CRM = \frac{\sum_{i=1}^{N} P_i - \sum_{i=1}^{N} O_i}{\sum_{i=1}^{N} O_i} \tag{3}$$

$$\text{Index of Agreement, } IoA = 1 - \frac{\sum_{i=1}^{N}\left(P_i - O_i\right)^2}{\sum_{i=1}^{N}\left(\left|O_i - \overline{O}\right| + \left|P_i - \overline{O}\right|\right)^2} \tag{4}$$

$$\text{Model Efficiency, } EF = \frac{\sum_{i=1}^{N}\left(O_i - \overline{O}\right)^2 - \sum_{i=1}^{N}\left(P_i - O_i\right)^2}{\sum_{i=1}^{N}\left(O_i - \overline{O}\right)^2} \tag{5}$$

Simulations of drainage in Webster continuous corn plots with a tile spacing of 30 m and a drain depth of 1.2 m are used to study the drainage patterns over 60 years (1945–2004), using DRAINMOD with a set of soil hydraulic parameters that proved to be sufficient.

Principal Findings and Significance

Soil Hydraulic Parameters

The Webster soil textural data, bulk density, and soil water contents at 2 pressure points are shown in the upper part of Table 1. Since the ISPAID 7.1 only offers the data for the surface soil, the silt and clay content are higher than those obtained from laboratory analysis. However, the bulk density 1.42 g cm^{-3} found in ISPAID is higher than those calculated from Equation (1). Sand content SP_1 is 20% while it is 38%, 33%, and 44% in the top, middle, and bottom layers of Webster soil from laboratorial analysis. In SP_2, the soil in the bottom layer retained higher volumetric soil water content at the two pressure points.

The soil hydraulic parameters for van Genuchten Equation derived from soil parameter input SP_1 and SP_2 by ROSETTA were included in the middle part of Table 1. The difference of these parameters is small except for the saturated hydraulic conductivity and lateral saturated hydraulic conductivity (K_{sat} and LK_{sat}). The K_{sat} in SP_1 is much lower than those in SP_2, which were consistent with bulk density and sand content.

Included in the bottom of Table 1, denoted as Calibration Output SP_3, are the optimized soil hydraulic parameters. After the optimization, K_{sat}, LK_{sat} and α increased to the extent of 1.5 to 2 times while other parameters kept unchanged.

Table 1 Input Soil Parameters and Output Soil Hydraulic Parameters for the 3 Soil Parameter Sets

Input/ Output	Soil Para-meter Set	Soil Properties							
		Soil depth	Texture			Bulk density	Θ_{33kPa}	$\Theta_{1500kPa}$	
		(cm)	Silt (%)	Clay (%)	Class	(g cm^{-3})	(cm^3 cm^{-3})	(cm^3 cm^{-3})	
Rosetta Input	SP_1	0-390	49	31	Clay Loam	1.42	-	-	
	SP_2	0-25	33	29	Clay Loam	1.16	0.34	0.18	
		25-50	37	30	Clay Loam	1.19	0.34	0.18	
		50-390	29	27	Loam	1.38	0.35	0.21	
					Soil Hydraulic Parameters				
			q_r	q_s	K_{sat}	LK_{sat} *	■	n	l
			(cm^3 cm^{-3})	(cm^3 cm^{-3})	(cm d^{-1})	(cm hr^{-1})	(cm^{-1})	(-)	(-)
Rosetta Output	SP_1	0-390	0.08	0.43	8.76	0.55	0.008	1.51	-0.24
	SP_2	0-25	0.07	0.49	40.72	2.55	0.022	1.30	-1.11
		25-50	0.07	0.48	32.08	2.01	0.017	1.32	-0.79
		50-390	0.06	0.43	20.5	1.28	0.016	1.33	-0.79
Calibration Output	SP_3	0-390	0.08	0.43	12.96	0.77	0.018	1.51	-0.24

* Lateral saturated hydraulic conductivity is 1.5×Ksat.

14-year Drainage Prediction

Yearly observed and predicted subsurface drainage volume and the statistical measures are included in Table 2. The total predicted drainage in the 14 years with SP_1, SP_2, and SP_3 were 174.5, 176.2, and 182.8 cm, and the total observed drainage volume in Webster soil plots was 179.6 cm. The predicted drainage with parameters from all the 3 methods fitted well with the observed data, and the statistical measures showed little difference. In the years of 1991, 1995, 1997, 1999, and 2001, the drainage flow was over-predicted while it was under-predicted in the years of 1990, 1992, 1996, 1998, and 2002 with any soil parameter set. The predicted flow in SP_3 is higher than SP_1, SP_2, and the observed in general. Although the drainage flow was underestimated by DRAINMOD with SP_1 and SP_2, there is little difference among the 4 statistical measures in the 3 parameter sets. However, it can be concluded that DRAINMOD performed better with data set SP_2 than it did with other sets from the ranking of the statistical measures over the entire 14-year record.

Table 2 Measured and predicted annual drainage with the 3 levels of method and the statistical measures

Year	Measured (cm)	Predicted (cm)		
		SP_1	SP_2	SP_3
1990	27.24	20.94	23.87	22.73
1991	23.60	25.95	26.04	25.75
1992	17.01	13.33	11.85	13.37
1993	32.80	29.49	31.29	34.72
1994	3.31	4.07	2.64	2.60
1995	3.61	6.40	6.26	6.61
1996	15.37	11.91	11.29	11.03
1997	0.15	3.76	3.64	3.98
1998	8.23	6.71	6.68	6.80
1999	2.03	5.25	5.13	5.87
2000	1.82	2.29	1.58	0.49
2001	13.25	16.38	17.59	18.72
2002	10.34	8.52	6.36	6.32
2003	20.85	19.47	22.02	23.77
Sum	**179.6**	**174.4**	**176.2**	**182.8**
RMSE		3.059	3.055	3.362
Rank		2	1	3
CRM		-0.029	-0.019	0.018
Rank		3	2	1
IoA		0.972	0.975	0.971
Rank		3	1	2
EF		0.907	0.907	0.888
Rank		1	1	2

Statistical measures, based on the monthly predicted and observed drainage during the drainage season, and their ranks are summarized in Table 3. The difference of each statistical measure is small. The calibrated hydraulic parameters (SP_3) performed the best among all three levels of soil parameter sets in predicting drainage volume from 1990 to 1993, since the observed data in these 4 years were used for calibration of SP_3. It was indicated that PEST optimized the hydraulic parameters and gave the best output. However, in the randomly selected validation years of 1994, 1996, 2001, 2002, and 2003, the overall statistical measures for SP_1 and SP_2 were better than those for SP_3.

Although the differences were small among statistical measures for monthly drainage with the 3 sets of soil information, soil parameter set SP_2 performed slightly better than either SP_1 or SP_3 because of its higher stability. In SP_1, five RMSE values in the years of 1990, 1993, 1996, 2001, and 2003 are greater than 2, and three RMSE in the years of 1991, 1994, and 2002 are less than 1; while in SP_2, only three RMSE are greater than 2 and two are less than 1.

The predicted drainage with 3 parameter sets had little difference. In 1991 and 2001, the drainage volume was overestimated. In the wet year 1993, DRAINMOD performs the best with SP_2 in predicting monthly flow. The predicted drainage with SP_3 was consistently higher than that with SP_2 or SP_1 in the year of 1993 and 2001 with a ranking order of drainage prediction of SP_3>SP_2>SP_1.

Table 3 Statistical measures and ranks for monthly predicted and observed drainage volume

Year	N	SP_1				SP_2				SP_3			
		RMSE	CRM	IoA	EF	RMSE	CRM	IoA	EF	RMSE	CRM	IoA	EF
1990	4	2.58	-0.23	0.94	0.78	1.91	-0.12	0.97	0.88	1.62	-0.17	0.98	0.91
1991	7	0.95	0.10	0.99	0.94	0.95	0.10	0.99	0.94	0.76	0.09	0.99	0.96
1992	9	1.42	-0.22	0.91	0.62	1.59	-0.30	0.88	0.52	1.36	-0.21	0.91	0.65
1993	7	2.04	-0.10	0.90	0.69	1.63	-0.05	0.94	0.80	1.58	0.06	0.95	0.81
Overall	**27**	**1.72**	**0.16 ***	**0.95**	**0.82**	**1.52**	**0.14 ***	**0.96**	**0.86**	**1.34**	**0.13 ***	**0.97**	**0.89**
Rank		**3**	**3**	**3**	**3**	**2**	**2**	**2**	**2**	**1**	**1**	**1**	**1**
1994	6	0.51	0.23	0.95	0.83	0.92	-0.20	0.74	0.45	1.16	-0.22	0.54	0.12
1996	6	2.30	-0.23	0.69	0.37	2.47	-0.27	0.64	0.27	2.59	-0.28	0.59	0.19
2001	5	2.08	0.24	0.93	0.71	2.26	0.33	0.92	0.66	2.47	0.41	0.91	0.59
2002	6	0.70	-0.18	0.96	0.87	1.19	-0.38	0.86	0.62	1.15	-0.39	0.89	0.64
2003	4	2.23	-0.07	0.85	0.65	2.60	0.06	0.79	0.53	2.55	0.14	0.82	0.54
Overall	**27**	**1.56**	**0.19 ***	**0.88**	**0.68**	**1.89**	**0.25 ***	**0.79**	**0.50**	**1.99**	**0.29 ***	**0.75**	**0.42**
Rank		**1**	**1**	**1**	**1**	**2**	**2**	**2**	**2**	**3**	**3**	**3**	**3**

* Average over the absolute value of CRM in each year.

An identical ranking order of predicted drainage volume with SP_3 > SP_2 > SP_1 is shown by the daily cumulative drainage volume in Figure 1. The total predicted drainage in the 14 years with SP_1, SP_2, and SP_3 were 174.5, 176.2, and 182.8 cm and the total observed drainage volume in Webster soil plots was 179.6 cm.

In summary, all the 3 levels of data set are proved to be sufficient to run the model. The difference between the drainage outputs is small. It is indicated that ROSETTA in combination with Soil Survey offers a quick and easy way to derive the soil hydraulic parameters. The results also showed that the combination of field soil textural measurements plus ROSETTA (SP_2) performed the best in yearly, monthly, and daily drainage volume prediction though the drainage output differences between SP_2 and two other methods, soil data from soil survey plus ROSETTA (SP_1) and model calibration (SP_3), are small. DRAINMOD showed a higher stability of statistical measures in predicting drainage with soil hydraulic parameters SP_2 than it did with SP_1 or SP_3. SP_2 included more accurate soil textural information, so it achieved a better output than SP_1 did. Even though the soil hydraulic parameters in SP_1 were calibrated by mathematical optimization for obtaining parameters for SP_3, this method (SP_3) did not perform better than the other two methods (SP_1 and SP_2) during the validation years. The procedure of comparing measured and simulated drainage for different levels of soil input information should be performed at other sites to evaluate the level of input soil information that is required to produce reliable predictions of drainage volume. This would be important for using a drainage model for sites where site-specific soil properties may not be available.

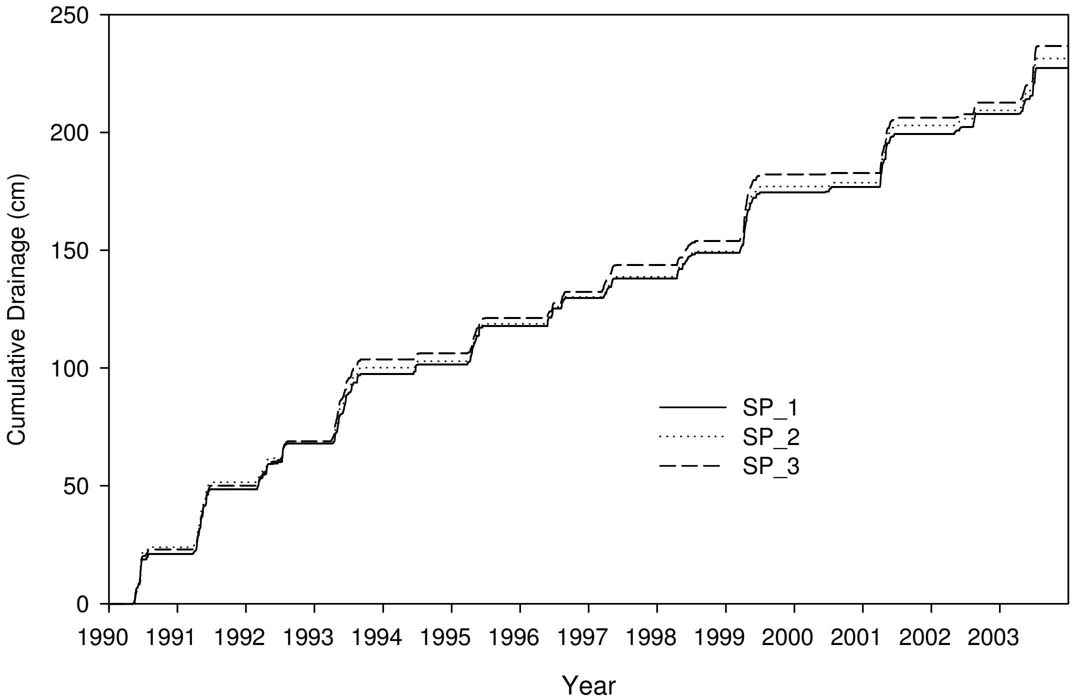

Fig. 1. Daily cumulative drainage volume predicted by DRAINMOD with soil parameter set SP_1, SP_2 and SP_3.

60-Year Drainage Prediction

Soil hydraulic parameters from SP_3 were adopted to simulate the conventional and controlled drainage over the 60-year period. As expected, precipitation and subsurface drainage show variability over the period used in this study. The highest precipitation (119.5 cm) was in 1993 and this also produced the highest subsurface drainage (44.2 cm). While precipitation patterns have a great influence on subsurface drainage, in general there is a correlation between annual precipitation and subsurface drainage (Figure 2). When reviewing the simulation results for this 60-yr period there is a 20% probability of exceeding approximately 20 cm of annual subsurface drainage when the drain spacing is 30 m (Figure 3), and, if the drain spacing was reduced, the drainage volumes are expected to increase.

While Figures 2 and 3 provide an indication of how annual precipitation affects subsurface drainage, precipitation and subsurface drainage patterns throughout the year are important for understanding how drainage management practices or other in-field management practices may be used to reduce the volume of subsurface drainage and subsequently NO_3-N export. In north-central Iowa, the higher precipitation months are from April through August. The higher subsurface drainage months are April and May (Figure 4). While the months of June, July, and August also had similar precipitation as April and May, the crop water use is more in the later months so, as expected, these months have lower volumes of subsurface drainage (Figure 5). This highlights that the periods of higher subsurface drainage occur when there is little vegetative growth on the landscape in a corn-soybean agricultural system. This also coincides with periods when subsurface drainage is important for trafficability and early crop growth. By the end of May, on average, approximately 60% of the annual subsurface drainage has occurred and by the end of June nearly 80% of the annual subsurface drainage has occurred. The months of April and May alone account for nearly 45% of the annual subsurface drainage. Reviewing just the months of April, May, and June there is a 20% probability of exceeding 5 cm of subsurface drainage in each of these months (Figure 6).

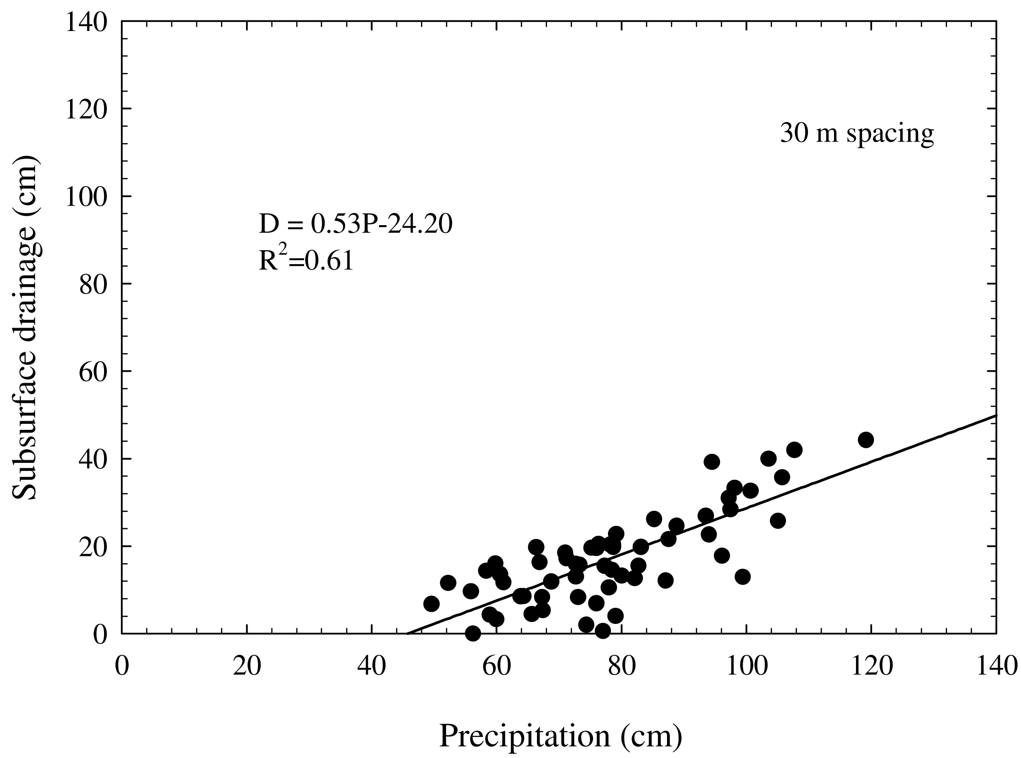

Fig. 2. Correlation between annual precipitation (cm) and simulated subsurface drainage (cm) over the 60 years (1945–2004).

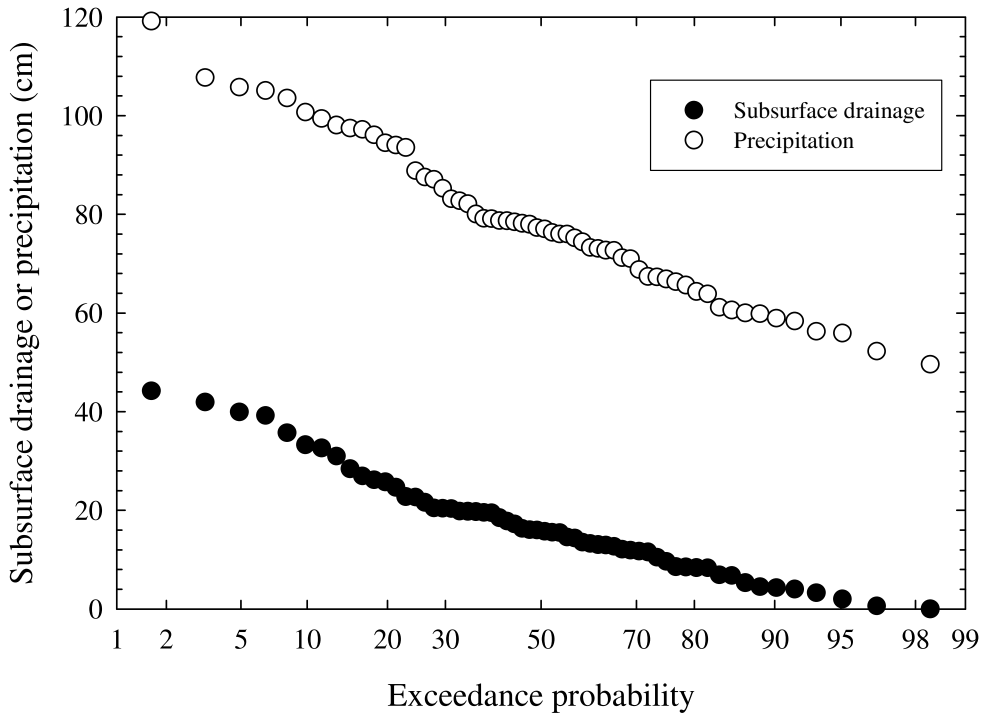

Fig. 3. Exceedance probability of annual precipitation (cm) and subsurface drainage (cm) over the 60 years (1945–2004).

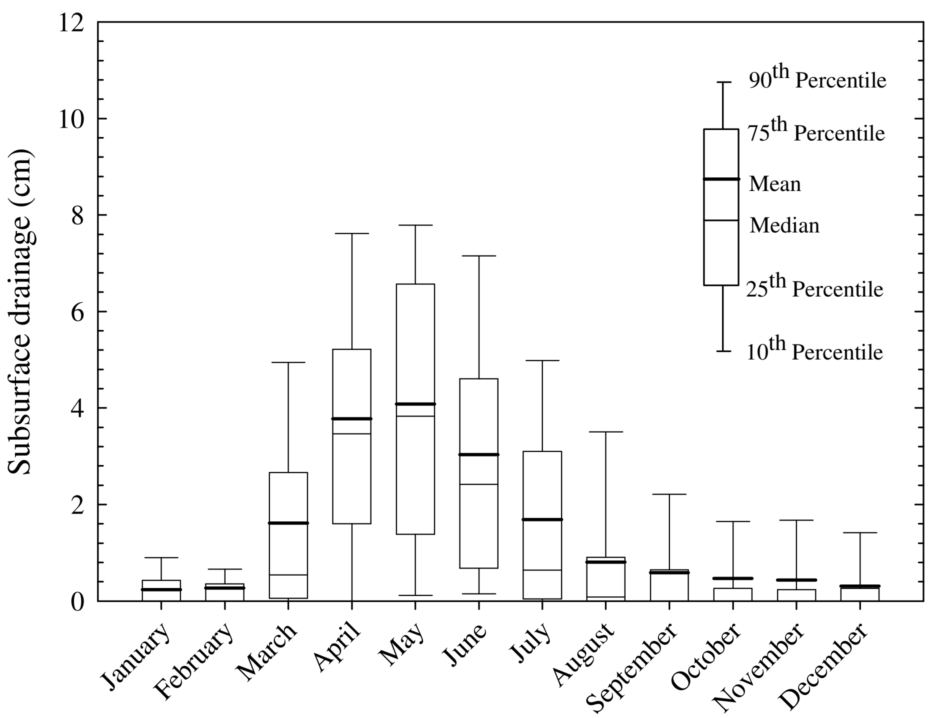

Fig. 4. Distribution of monthly simulated subsurface drainage (cm) over the 60 years (1945–2004).

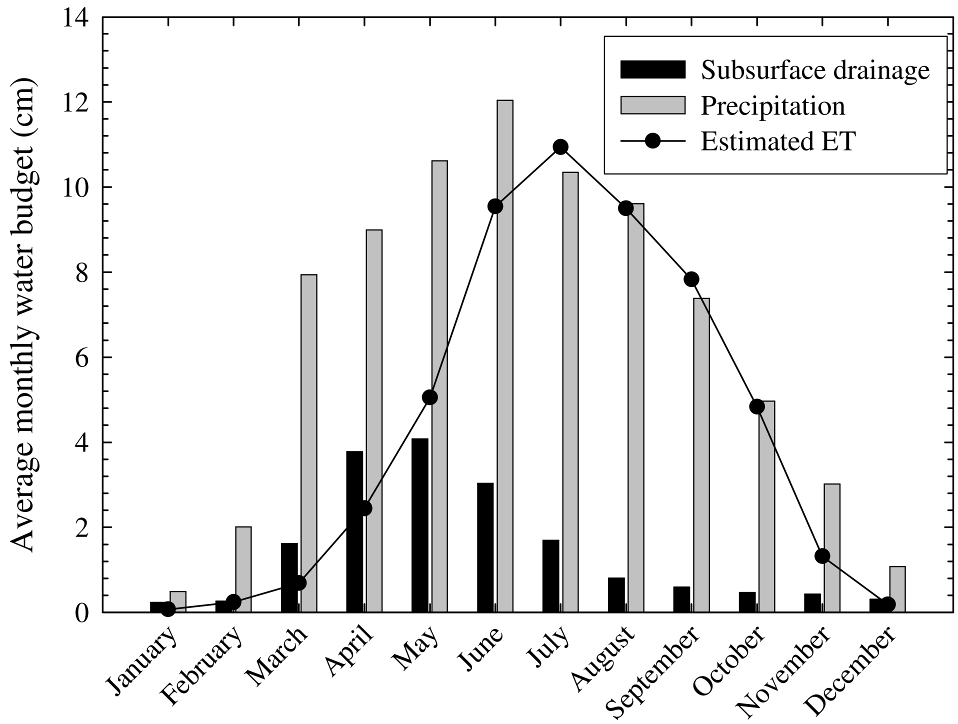

Fig. 5. Average monthly water budget (cm) over the 60 years (1945–2004).

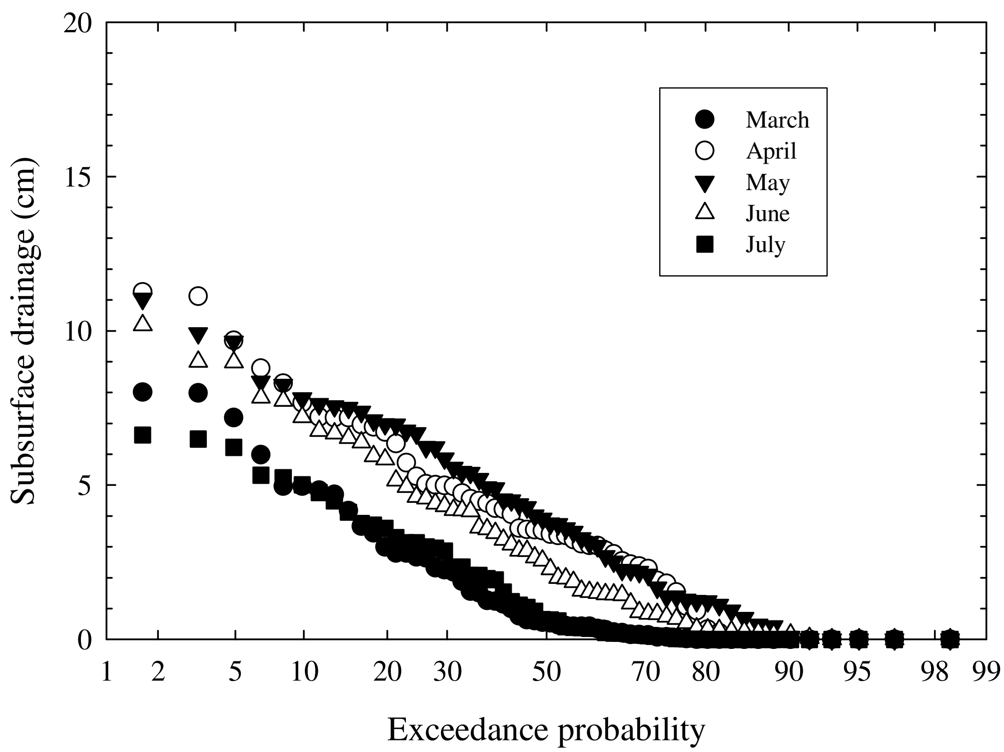

Fig. 6. Exceedance probability of average monthly subsurface drainage (cm) for specific months (from March to July) over the 60 years (1945–2004).

One drainage management practice being considered for reducing the volume of subsurface drainage is controlled drainage where the outflow of the subsurface drainage system is controlled during specific months of the year when maximum drainage is not required. Maximum drainage is required for trafficability during crop sowing (April and May) and harvesting (September and October) months. Using DRAINMOD, controlled drainage was investigated by controlling subsurface drainage outflow during the months of November through March and then again from June through August. During these months the drain outflow was restricted to maintain a depth of 60 cm below the ground surface, while free drainage at a normal outflow depth of 120 cm was used during the months of April, May, September, and October. Simulated average monthly subsurface drainage and surface runoff for conventional (free drainage) and controlled drainage over the 60 years (1945–2004) are shown in Figure 7. Since the months of April and May are time periods when both the conventional and controlled drainage systems have free drainage outflow at the normal outflow depth of 120 cm, there is no reduction in subsurface drainage in these months for the controlled drainage system. In fact, since water has been stored in the soil profile during the winter months and then released when the outflow level is lowered to free drainage level, there is an increase in subsurface drainage in April and May under the controlled drainage system. During the months when the outflow level is controlled to 60 cm there is a reduction in subsurface drainage with the controlled drainage management. However, much of this reduction is reflected in increased surface runoff (Figure 7).

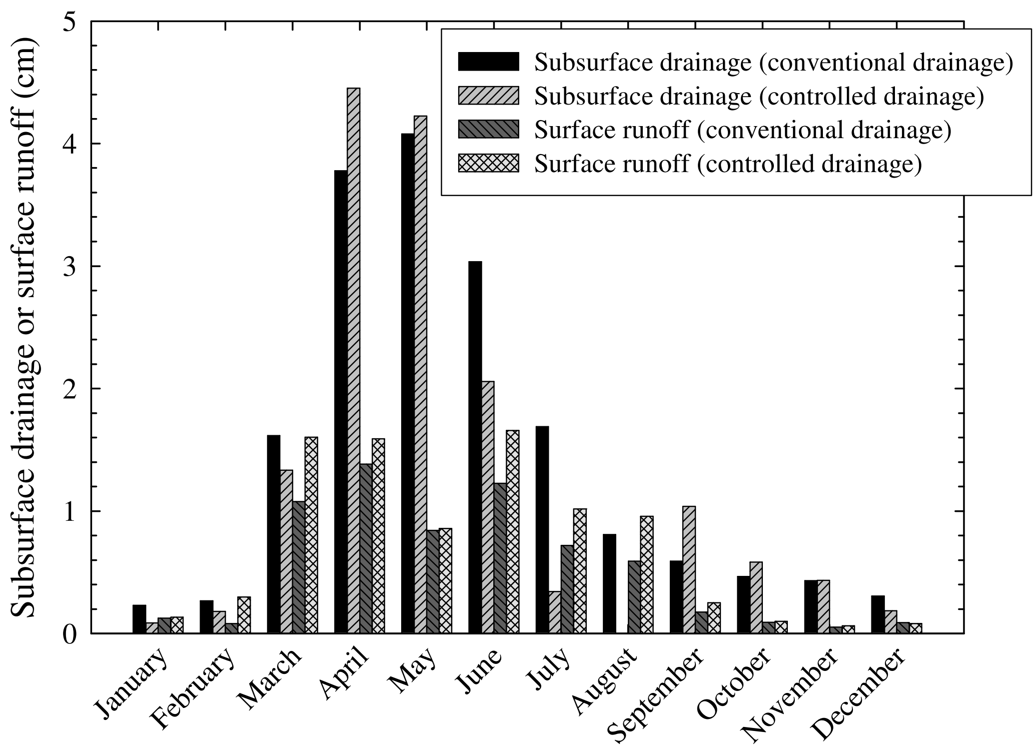

Fig. 7. Simulated average monthly subsurface drainage (cm) and surface runoff (cm) for conventional (free drainage) and controlled drainage systems over the 60 years (1945–2004).

The simulated annual subsurface drainage is reduced when using a controlled drainage management system (Table 4). However, as mentioned above, most of this reduction is reflected in increased surface runoff. There is approximately a 16% reduction in subsurface drainage and a 33% increase in surface runoff. When reviewing the exceedance probability for subsurface drainage and surface runoff for the controlled and conventional drainage, the controlled drainage consistently reduces the volume of subsurface drainage while increasing the volume of surface runoff (Figure 8). It must be noted that these simulations accounted for little vertical seepage as would be expected in north-central Iowa. From an intensive groundwater modeling study, Ella et al. (2002) estimated 2.3 to 4.3% of the annual precipitation as groundwater recharge in the glaciated region of the Des Moines Lobe, IA. Also, the simulations did not include lateral seepage that may occur in some controlled drainage situations so there is the possibility that there could be an increased reduction in subsurface drainage and less increase in surface runoff from controlled drainage when accounting for lateral seepage. The simulations showed little change in the estimated ET (Figure 9) but there is the possibility that ET could be increased under certain controlled drainage scenarios if there is water stored within the soil profile during the summer months. This possibility warrants additional investigations. These simulations show smaller reductions in the volume of subsurface drainage than many field investigations in other parts of the U.S have shown (Evans et al., 1995; Cooke et al., 2002; Sands et al., 2003; Burchell et al., 2003; Fausey, 2004). From this there is a need to study the performance of controlled drainage on a field-scale in Iowa and specifically account for the pathways of water movement.

Table 4 Simulated average annual subsurface drainage (cm) and surface runoff (cm) for conventional (free drainage) and controlled drainage systems over the 60 years (1945–2004).

	Subsurface drainage (cm)	% Reduction	Runoff (cm)	% Reduction
Conventional drainage	17.29		6.45	
Controlled drainage	14.54	15.9	8.59	-33.2

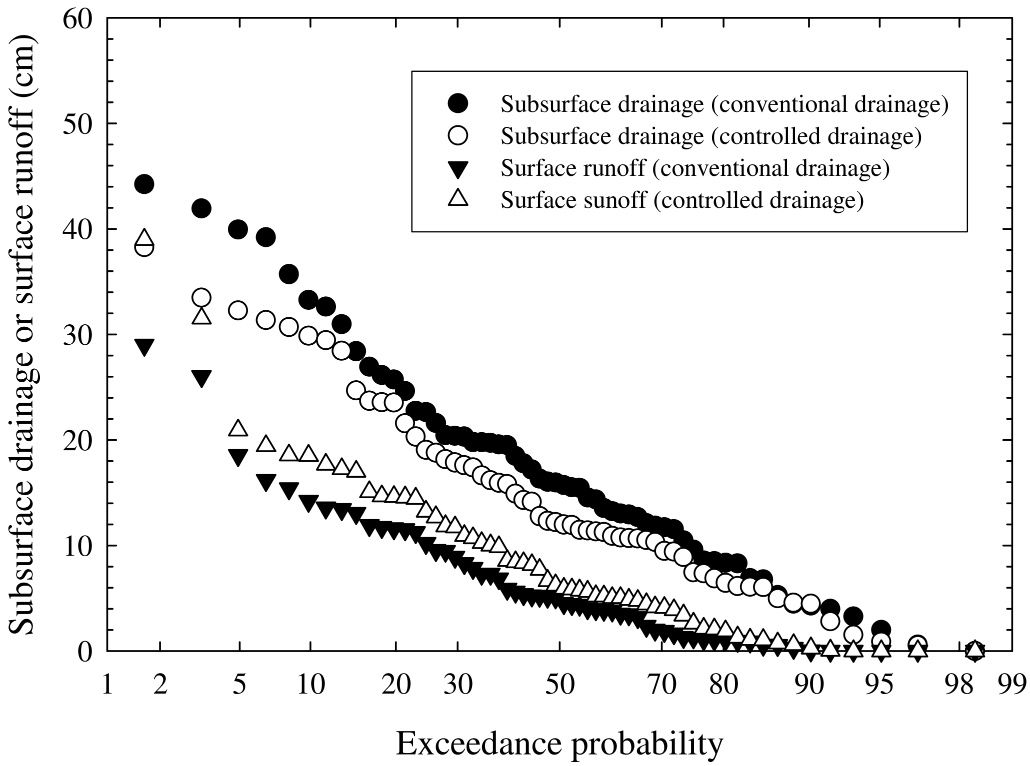

Fig. 8. Exceedance probability of simulated annual subsurface drainage (cm) and surface runoff (cm) for conventional (free drainage) and controlled drainage systems over the 60 years (1945–2004).

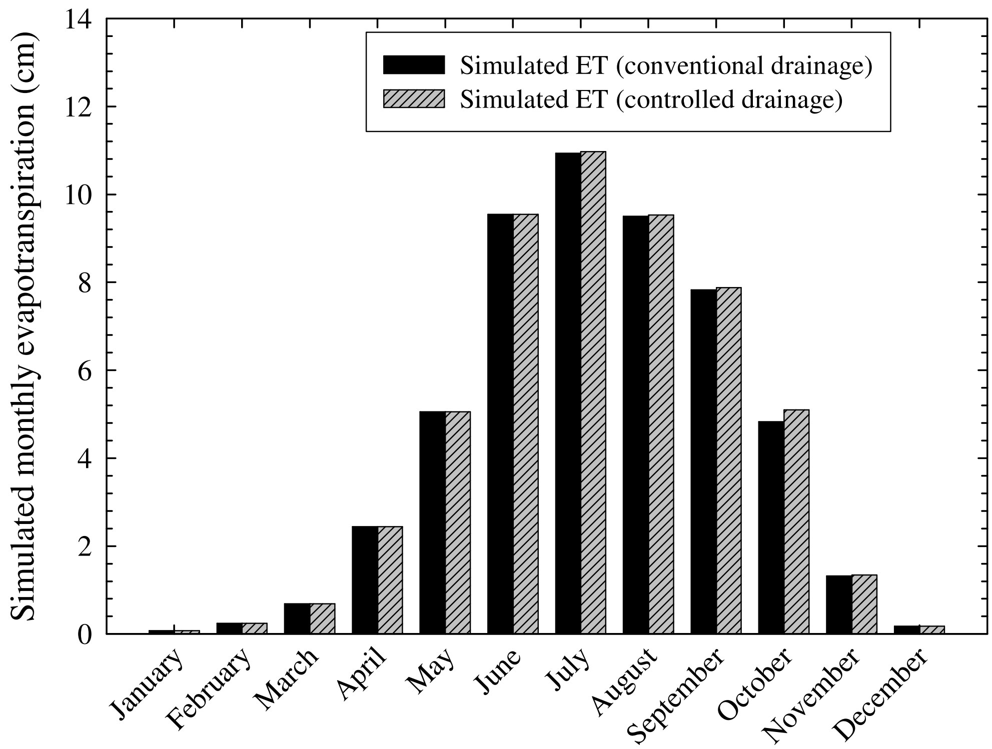

Fig. 9. Simulated average monthly evapotranspiration (cm) for conventional (free drainage) and controlled drainage systems over the 60 years (1945–2004).

From the analysis of precipitation, conventional and controlled drainage prediction, we found that approximately 45% of the annual subsurface drainage occurs in the months of April and May and approximately 80% of the annual subsurface drainage has occurred by the end of June. This coincides with the time when maximum drainage is required for trafficability and crop development specifically during the months of April and May. This coincident may limit the effectiveness of drainage management practices such as controlled drainage to reduce subsurface drainage in north central Iowa. When simulating controlled drainage where the outflow was controlled during the winter months (November to March) and the summer months (June to August), there was approximately a 16% reduction in the volume of annual subsurface drainage but most of this reduction was reflected in increased surface runoff. So, while controlled drainage has some potential to reduce subsurface drainage it would need to be managed so that any negative impacts of potential increases in surface runoff are considered. In addition to drainage water management practices there is a need to consider and study other management practices that could be used to reduce subsurface drainage specifically during the early spring months when there is little water use by the common corn-soybean agricultural system. Alternative cropping practices such as cover crops and living mulches are two potential practices that might increase water use during this time period. These practices may also use some of the available nitrate within the soil profile so that there is less risk of nitrate loss. There is a need for further study of subsurface drainage water management practices in the tile-drained landscapes of Iowa and the upper Midwest of the U.S. along with studies that investigate the potential for using cropping practices that may also have a positive effect on subsurface drainage and nitrate export.

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Improving water quality in Iowa rivers: cost-benefit analysis of adopting new conservation practices and changing agricultural land use

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3. Feng, Hongli, Manoj Jha, and Phil W. Gassman. Allocating Nutrient Load Reduction across a Watershed: Implications of Different Principles, Economics Working Paper #06013, Iowa State University, March 2006.

Improving Water Quality in Iowa Rivers: Cost-Benefit Analysis of Adopting New Conservation Practices and Changing Agricultural Land Use

C.L. Kling, L.A. Kurkalova, S. Secchi, H. Feng, P.W. Gassman

Problem and Research Objectives

This research project built upon the research that CARD conducted through an Iowa Department of Natural Resources (IDNR) funded project. The IDNR project assessed the in-stream water quality benefits and monetary costs of a single conservation policy for agricultural land throughout the State as part of the Environmental Protection Agency National Needs Assessment to determine the needs of states to address nonpoint source water quality problems. As a natural extension of this assessment, this project considered the sensitivity of the water quality improvements and costs of the policy under several alternative scenarios, thus evaluating cost-efficiency of alternative conservation plans. The water quality measures of interest are sediment, nitrogen and phosphorus. Nonpoint source pollution due to agricultural activities is a vital issue for the State. This project would provide a first assessment of the overall impact on in-stream water quality of a large scale conservation policy that includes several practices simultaneously. For the Raccoon watershed, where the calibration and validation of the Soil and Water Assessment Tool (SWAT) is focused, we conducted an additional study to assess alternative targeting policies.

Methodology

Micro-unit-based economic models and data on land use and conservation practices are combined with the watershed-based hydrological model (SWAT) to estimate the costs of obtaining water quality changes from the hypothetical placement of conservation practices. The main procedures involved in the project include: identify the location of a conservation practice, calculate the cost of alternative conservation policies, estimate the water quality consequences of the policies, and, finally, assess the cost and benefit of the policies. For more detail, please see Secchi et al. attached working paper.

For the Raccoon River watershed, we use SWAT, along with transfer coefficients, to assess alternative principles of allocating nutrient load reduction in the Raccoon River watershed in central Iowa. Four principles are examined for their cost-effectiveness and impacts on water quality: absolute equity, equity based on ability, critical area targeting, and geographic proximity. Based on SWAT simulation results, transfer coefficients are calculated for the effects of nitrogen application reduction. For more detail, please see Feng et al. attached working paper.

Principal Findings and Significance

During the first year of the proposal, we have finished investigating the costs and benefits of improving water quality in Iowa under two different policy objectives. We have found that conservation policies can achieve quite different results if the policy focus is shifted.

Specifically, we examined two scenarios, estimating for both of them in-stream sediment, nutrient loadings, and carbon sequestration. In the first scenario we assessed the impact of a program designed to improve water quality in Iowa on carbon sequestration, and in the second scenario we calculated the water quality impact of a program aimed at maximizing carbon sequestration. Our results indicate that the amount of benefits depends on which indicators are used to measure water quality. Our results also suggest that, because various water quality indicators respond differently to conservation efforts, the term "improving water quality" can be ambiguous. For a more detailed analysis, please see the attached working paper.

For the study on the Raccoon River watershed, we find both critical area targeting and downstream focus (an example of geographic proximity) can be more expensive than equal allocation, a manifestation of absolute equity. Unless abatement costs are quite heterogeneous across the subwatersheds, the least-cost allocation (an application of the principle of equity based on ability) has a potential of cost savings of about 10% compared to equal allocation. We also find that the gap between nitrogen loading estimated from transfer coefficients and nitrogen loading predicted by SWAT simulation is small (in general less than 5%). This suggests that transfer coefficients can be a useful tool for watershed nutrient planning. Sensitivity analyses suggest that these results are robust with respect to different degrees of nitrogen reduction and how much other conservation practices are used.

The designation of co-benefits and its implication for policy: water quality versus carbon sequestration in agricultural soils

by

Silvia Secchi, Manoj Jha, Lyubov Kurkalova, Hongli Feng, Philip Gassman, and Catherine Kling

The designation of co-benefits and its implication for policy: water quality versus carbon sequestration in agricultural soils

The design of policies to induce the adoption or maintenance of conservation practices is often complicated by the fact that many conservation practices produce multiple benefits. For example, conservation tillage has the potential to sequester carbon as well as to reduce soil erosion. Likewise, land set aside and planted with grass or trees can improve wildlife habititat and stop or reduce nutrient runoff in addition to storing carbon and reducing erosion. If the social value of each benefit is known, then the total value from the multiple benefits can be summarized in a comprehensive index and policy design would be straightforward: sound policy would aim to maximize the value of this index for any given budget.

However, it is difficult to assess the value of benefits from conservation practices. Most of these goods are non-market and so monetary values are not always available, even if we know the environmental improvements in physical quantities. Water quality provides an apt example. There are many studies that estimate the value of improved water quality through contingent valuation methods or travel cost models, nonetheless it is a challenging task to connect the "water quality" used in these studies (which is often represented by criteria such as whether a lake or river is swimmable, fishable, or boatable, etc.) to the "water quality" represented by the reduction of nutrient or sediment loading, the criterion typically used in actual measurement or water quality simulation models. Moreover, even the size of environmental improvements in physical quantities is a major task to assess whether by costly direct measurement or complex and uncertain model simulations.

There is a large body of literature which focuses on a single benefit from conservation practices. For example, there are many studies that investigate the cost of carbon sequestration without considering other benefits. Likewise, the least cost way of improving water quality or nutrient loading in waterways has also been examined extensively (e.g., Ribaudo et al. 2001), again largely without consideration for the other benefits of the practices that would implement the policy such as carbon sequestration. More recently there has been some attention paid to the multiple benefits of conservation practices. Often in this literature one benefit is treated as the primary benefit and others as co-benefits (Kurkalova et al. 2004), although the benefits have also been considered as a bundle.

The views in the literature are mirrored in the historical development of at least some conservation programs. For example, in the early days of the Conservation Reserve Program (CRP), the primary land set aside program in the United States, erosion reduction was the main focus. However, the current CRP takes into account a number of benefits through an index which gives weights to a variety of environmental factors. As reflected in the index, water quality receives more weight than carbon sequestration, but carbon is also included

In this paper, we investigate the implications of treating different environmental benefits as the primary target of policy design for the entire bundle of environmental benefits that society values. That is, when we consider other environmental goods as co-benefits resulting from policies that sequester carbon rather than the focus of the programs, it implies that programs will target more heavily carbon sequestration. In contrast, if water quality is the focus of the program, it implies that the program will not heavily target carbon. We undertake an empirical study to demonstrate the importance of this issue in the context of a vital policy debate occurring in much of the Midwestern United States related to the policies and conservation practices that will be necessary to improve water quality in local rivers and streams. In particular, we focus on two issues: (a) the carbon sequestration co-benefits of currently proposed and/or implemented programs designed to improve water quality and (b) the value of various bundles of water quality and carbon benefits associated with the programs and alternatives to the programs that focus more on carbon than water quality.

It is a vital public policy issue to understand the carbon sequestration co-benefits of current programs and the degree to which changes in program design could increase carbon storage with relatively small tradeoffs in other environmental benefits. Further, we perceive something of a mismatch between the focus that the academic literature has placed on carbon sequestration and its co-benefits and the policy arena where programs are more often being considered with a focus on water quality (making carbon the co-benefit). There is a large amount of expenditures on conservation in agriculture, much of which is meant for water quality, and many evaluation studies have shown that people are willing to pay for water quality improvement (for example, Lipton 2004, and Desvousges et. al 1987). According to Ribaudo (1989), water quality benefits for the Corn Belt were greater than $80.00 per acre; the number was twice as high in the Lake States. In addition to the potential benefits, the National Needs Survey indicates that a large amount of funding may be needed to meet the water quality needs. This implies that the nation might be poised to devote a large amount of resources to improving water quality. Undoubtedly, a co-benefit of these activities will be an increase in carbon sequestration, and in this paper we will examine empirically the potential magnitude of carbon that can be achieved through programs mainly intended for water quality.

Specifically, we study a set of policies designed to support the Clean Watersheds Needs Survey for the state of Iowa. This survey is required as part of the Clean Water Act and it requests states to identify the financial resources necessary to meet water quality goals from non-point source reductions (primarily agriculture). As part of this effort we are performing an analysis of the costs of improving Iowa's in-stream water quality by linking an economic model of farmers' choices to a watershed-based hydrologic model, the Soil and Water Assessment Tool (SWAT).

The remainder of the paper proceeds as follows. In the next section, we describe the water conservation "policy" developed to support the needs assessment for the state of Iowa and two additional policies focused on carbon sequestration. These form the basis for our comparisons and assessment of the magnitude of co-benefits. In section II, we describe the estimation procedures and data we employ to estimate the costs and instream water quality benefits associated with these three policies. In section III, basic results of the simulations are presented and interpreted. In section IV, we undertake a simple breakeven type analysis to examine the implicit "price" of nutrient and erosion reductions (measured at the edge of the field) that would be necessary to make these policies pass a simple benefit cost test. Conclusions and final thoughts follow.

I. Conservation policies for water quality and carbon sequestration

The U.S. Environmental Protection Agency (EPA) is required to perform a periodic national Clean Watersheds Needs Survey in response to directives that were established in the 1972 U.S. Clean Water Act. The purpose of the survey is to identify all existing water quality or public health problems. As interpreted by the 2000 EPA National Needs Assessment (USEPA 2003), the process of determining the needs of states to address nonpoint source water quality problems consists of two steps. The first is the identification of the set of conservation practices and land use changes that should be placed on the landscape, and the second is the estimation of the costs of those practices. As part of this process for the State of Iowa, in consultation with the Iowa Department of Natural Resources (IDNR), we determined the location of the practices based on the potential environmental impact as captured by several indicators, such as proximity to a stream, an erodibility index, and slope. Several practices were included in this exercise. Here, we consider a key one: land set-aside. We choose this practice to demonstrate the importance of the issues we raise because land set-aside is a major change in land use: it is both costly and land well suited for the sequestration of carbon is not necessarily well suited to the reduction of sediment and nutrient loadings to waterways. Thus, there is the possibility of very different efficient configurations of a policy designed to focus on carbon sequestration relative to water quality.

The water quality improvement scenario studied here assumes that all land within 100 feet of a waterway will be placed out of production and planted with perennials. This area amounts to about 251,600 acres. Additional land is retired on the basis of the Erodibility Index (EI), until 10% of total cropland is retired from production. This amounts to an additional 2,172,100 acres. The EI indicates the potential of a soil to erode on the basis of both climatic factors and the properties of the soil. A higher index indicates higher erodibility potential. The rational for this choice is that choosing land closer to waterways to retire means that it can filter sediment and pollutants from upstream farms. The highly erodible land, on the other hand, is highly correlated with elevated levels of sediment loss. The 10% cropland cap was chosen in concert with the IDNR, taking into account both the high level of impairment of Iowa's waters and the issues linked with taking large areas of farmland out of production[1].

We also consider carbon focused policies by ranking each piece of land on the basis of its ability to sequester carbon when retired from production. We then "enroll" land into the program based on the highest carbon benefit. To maintain comparability with

[1] Note that this area is in addition to the already existing CRP area.

the water quality scenario, we consider two cases: in the first, we keep constant the amount of acres enrolled in the set-aside program (i.e., we enroll acreage until a total of 10% of the cropland in the state is retired). In the second, we keep constant the cost of the program. That is, we enroll acreage into the program until the expenditure total is the same as the amount spent under the hypothetical water quality scenario.

II. Data and models

To develop our models, we draw heavily from the National Resource Inventory (NRI) (USDA-NRCS 1997) to provide data on the land use, cropping history, and farming practices in the state of Iowa. The NRI is the most comprehensive data set on land use in the United States and we use data on the 14,472 physical points in Iowa that represent cropland (Nusser and Goebel 1997). Conceptually, our data and models are based on individual producer and farm level behavior and we treat an NRI point as a producer with a farm size equal to the number of acres represented by the point (the expansion factor provided by the survey). Figure 1 illustrates the 35 watersheds corresponding to the United States Geological Survey 8-digit Hydrologic Cataloging Units that are largely contained in the state and which are modeled in this study.

The costs of enrolling into the land set-aside program are estimated using the model developed in Feng et al. (2004), in which the opportunity cost of land retirement is determined by using the cropland cash rental rate. The average cost of land retirement in the sample is $123 per acre with a standard deviation of $21 per acre. The rental rates are the highest in the central and northern parts of Iowa, averaging as high as almost $140 per acre in subwatersheds of Iowa and Wapsipinicon rivers (Figure 1). The least expensive land to retire from crop production is in the southern portions of the Des Moines River as well as in the Nishnabotna and Nodaway river watersheds where the average land retirement cost is as low as $105 per acre.

To compute the amount of carbon sequestered when an NRI point is retired from cropland, we rely on estimates from the Environmental Policy Integrated Climate (EPIC) model version 3060 (Izaurralde et al. 2001). The current version of the EPIC model features enhanced carbon cycling routines (Izaurralde et al. 2001) that are based on the approach used in the Century model (http://www.nrel.colostate.edu/projects/century5

/reference/index.htm; Parton et al. 1994). More generally, EPIC is a model that estimates the changes in erosion, carbon sequestration, and nutrient runoff measured at the edge of the field from changing farming practices. Inputs into the model include weather, soil, landscape, crop rotation, and management system parameters. When land is retired from crop production, we assume that annual grasses are planted and maintained on the land and we run a 30-year simulation with EPIC to predict the carbon sequestration level associated with this change. For each NRI point, we also calculate the soil erosion reduction at the edge of the field.

In addition to EPIC, we also rely on estimates from a watershed-based model to assess the conservation policies. Unlike carbon sequestration, a key concern in the design and implementation of policies to improve water quality is to recognize that there are critical interactions between land uses within a watershed that affect the total water quality level obtained. Thus, for otherwise identical tracts of land, more water quality improvement may occur from retiring a piece of land from production that is located downstream from numerous other cropped points. The potential filtering effect is just one example of the physical processes that a fully integrated watershed hydrological model should capture. So that we can study in-stream water quality changes, we employ the Soil and Water Assessment Tool (SWAT) to estimate changes in nitrogen, phosphorous, and sediment loads from retiring a particular set of parcels from production within a watershed. The model represents hydrology, plant growth, erosion, fate and transport, and various management practices (Arnold et al. 1998 and Neitsch et al. 2002). SWAT model achieves a high level of spatial differentiation by dividing a large watershed into a number of subwatersheds and then further divided into Hydrologic Response Units (HRUs). Each HRU represents homogeneous land use, management, and soil characteristics. To estimate the in-stream water quality consequences of the increase in land set aside, we have calibrated the SWAT model for each of the watersheds identified in Figure 1 to baseline levels[2]. By running the model at the set-aside levels "after" the policy, we can compute the changes in water quality attributable to the increase in land set aside.

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[2] The details of the calibration and validation process for SWAT can be found in Jha et al. 2005 and Gassman et al. 2005.

Given that political boundaries and watershed boundaries do not perfectly correspond in Iowa, there is something of a geographic mismatch between our study regions on the cost side and on the water quality side.

The watersheds studied correspond to 13 outlets, at which the in-stream water quality is measured. The water quality measures of interest are sediment, nitrogen, and phosphorus. For the cost analysis we consider placing the identified set of practices all across the state and exclude the pieces of the watersheds that fall outside of the state boundaries (for example the section of the Des Moines River watershed that falls in Minnesota). Thus, the costs and water quality benefits we report are not quite consummate: one represents a political boundary (the statewide costs), the other represents a natural system boundary. Direct comparisons between the aggregate cost and water quality benefit comparisons may be misleading, although the unit costs and benefits (per acre costs and/or per outlet of the watershed measures) can still be appropriately compared.

III. Results

The three policies considered are quite similar in terms of the acreage enrolled, as illustrated by Table 1. However, Figures 2, 3, and 4 show that the location of those acres is very different. The water quality policy would enroll more acres along the Mississippi and Missouri, in the more hilly parts of the state. The carbon policies, on the other hand, would focus on the central part of Iowa, in the ecoregion known as the Des Moines Lobe, a flat area, with very productive agriculture and particularly suited for carbon sequestration. Note the similarities between Figures 3 and 4. Because the criterion used to enroll land is the same, the only difference is on the additional amount of land allowed in the equal area scenario.

The policies are extremely different in the levels and location of environmental benefits. The carbon-based policies sequester about ten times as much carbon as the water quality policy. Similarly, the water quality policy is around four times more effective than the carbon-based policies in reducing soil erosion at the edge of field as calculated with EPIC.

Figures 5 and 6 illustrate the location of the carbon sequestered across the state in the water quality policy and in the carbon policy that keeps costs constant,[3] and Figures 7 and 8 show the location of the edge-of-field soil erosion reduction across the watersheds studied. As with the levels of the benefits, we find the location of the benefits to be quite different across the policy designs. The extreme differences are at least partly driven by the fact that either policy is only affecting a relatively small percentage of the cropland. More substantial overlap of benefits could occur in policies applying to more extensive areas, as it was found in the case of conservation tillage adoption in Iowa (Kurkalova et al. 2004).

The policy design also substantially alters the in-stream water quality impacts reported in Tables 2 and 3. For a Water Quality policy (Table 2), with only one exception, sediment is reduced across all the watersheds from the baseline[4]. Since the enrollment criterion was the EI, this is an expected result. If we sum the sediment loads across the watersheds, the total reduction is 11%. The higher reductions are along the Mississippi and in southwest Iowa, reflecting the watersheds where more land is put out of production. Reductions in phosphorus loads are somewhat smaller, but still significant, as phosphorus loads are linked to sediment. The policy, however, is not effective in reducing nitrates, and, since nitrates are the most important form of nitrogen in surface water, total nitrogen is not reduced by much.

In contrast, for a Carbon Policy (equal cost) (Table 3), we find that in several watersheds, particularly in eastern Iowa, the sediment loadings actually increase. Further investigations on this result are needed, but the likely reason for this finding is the fact that the NRI points selected for carbon sequestration purposes in these areas were farmed with effective conservation practices such as filter strips and grassed waterways that actually were more effective at capturing sediment in our modeling system. On the other hand, it is interesting to note that the carbon-based policies are better than the water quality policy in reducing nitrates. The reason is that the land taken out of production in the carbon-based policies is prime agricultural land, heavily fertilized. The higher reductions are in the central watersheds part of the Des Moines Lobe, where most of the acres of set-aside in

[3] The two carbon policies are very similar, therefore only the results for the equal cost case are shown.
[4] See Appendix 1 for the baseline loads.

the carbon-based scenarios are located. These results suggest that even a water quality based policy may have to deal with trade offs, depending on which measure of water quality is used. Implicit in our setup was a heavier weight on the importance of sediment and phosphorus loadings reduction, since the NRI points were selected on the basis of the land erodibility.

IV. Indirect monetization and lessons for policy design

As noted in the introduction, it is not straightforward to assign monetary values to the nonmarket goods of reduced nutrients or soil erosion. It is also not straightforward to assign a monetary value associated with sequestered carbon, although a number of analyses have been undertaken that suggest likely values of the price of carbon in an efficient and fully-functioning trading program. In this section, we undertake a simple breakeven type analysis to consider the marginal value of erosion and nutrient reductions that would have to hold under a range of likely carbon prices to make the alternative policies socially efficient, i.e., to be assured that they would pass a simple cost-benefit test. We are implicitly assuming in this exercise that the hypothetical carbon prices would reflect the social value of carbon reductions.

First, we estimate the minimum value of erosion reduction that would have to be held by society to justify the policy outlay. This is calculated assuming that the total cost of the policy must be equal to the monetary valuation of the benefits from the policy. Two benefits are considered at the time, carbon sequestration and erosion reduction. The monetary valuation of the benefits is equal to the monetary valuation of carbon sequestration, C, multiplied by its price p^C plus the monetary valuation of erosion reduction, E, multiplied by its price p^E. Thus, for the policy to "break even", we need $C \cdot p^C + E \cdot p^E = Budget$. For the scenarios considered, we have the $Budget$, C, and E. The literature provides several estimates of p^C. We use these pieces of information to find p^E from the equation above. Specifically, we consider 3 prices for carbon, p^C = 10, 50, and 100, and get 3 prices for erosion, p^E for each of the scenarios considered. The results are reported in Table 4. At high enough carbon prices, carbon benefits are more than enough to justify these carbon-based policies. On the other hand, if the goal of the policy is to improve water quality/reduce erosion, increasing the price of carbon will not affect the value of erosion reduction. The reason is that the water quality policy reduces erosion by 16 million metric tons, versus 0.3 million tons of carbon sequestered. Carbon prices would have to rise much higher than $100/metric ton to affect the break-even price of erosion. Pimentel et al. (1995) estimate an off-site[5] cost of erosion of about $3 a ton in 1992 dollars. This estimate assumes that more than half of the costs are caused by wind erosion, and does not consider biological impacts such as the effect on biodiversity. As the authors of the study note, this is likely to underestimate the impact of soil erosion. However, even tripling the number calculated by Pimentel et al. would produce prices lower than the lowest implicit break-even prices from our scenarios. This suggests that a land set-aside policy is unlikely to be justified on the grounds of soil erosion alone.

The calculation of break-even prices for in-stream pollutant load reductions is more complex to interpret for several reasons. First, there is less precedent on how the reduction should be measured, and therefore there is no consensus on how the benefits should be compared. We are using reductions in metric tons of the loads at the outlet of each watershed, but, as we noted above, this does not directly correlate with measures such as swimmability and transparency. The units of analysis are not immediately apparent either. For example, Table 5 reports the break-even prices for total phosphorus load reductions, using metric tons of reduction as the common unit. Because the edge of field values are much higher than the reductions in stream, the price of phosphorus load reductions is extremely high compared to the price of carbon.

An alternative way to assess the trade offs is to present the budget outlays for each benefit, either in dollar terms or in percentages, as illustrated in Table 6. This allows a comparison of the trade offs involved without having to decide the appropriate units of comparison. Clearly, this is an issue that needs further investigation, and additional empirical analysis of various categories of co-benefits.

Additional analysis is needed also because, as our results show, viewing different benefit indicators as the primary benefits and other indicators as co-benefits in policy making would produce substantially different policy scenarios. In particular, there should

[5] We do not consider the on-site impacts of soil erosion because they directly affect productivity and should be accounted for in the farmer's profit maximizing problem.

be careful consideration of the environmental goals of conservation policies implemented on only a fraction of cropland, because the trade-offs for such programs are likely to be more extreme. Policies included in this category are likely to be, besides land set aside, wetland conversion and adoption of no till, as opposed to reduced/mulch till, which is likely to be more widespread.

Though our results are limited to the State of Iowa, and to the case of set aside for water quality or carbon sequestration purposes, it is apparent that at our scale of analysis, the amount of co-benefits depend on which indicators are used to measure water quality. Local water quality issues are usually linked with phosphorus, since phosphorus tends to be the limiting factor in freshwater bodies. However, at least until recently, there was a consensus that nitrogen was the main cause of hypoxia in the Gulf of Mexico. In general, this study shows that improving "water quality" in the sense of reducing nutrient or sediment loadings is too vague. Even if it is taken to refer to in-stream nutrients, because the responses of nitrogen and phosphorus to conservation efforts are not well correlated, this terminology may not provide much guidance. In the case of the CRP's environmental benefits index, for example, the water quality benefits refer both to a generic impairment that includes nutrients and also specifically to sediment (USDA - FSA 2004).

We have not considered co-benefits such as wildlife habitat or biodiversity, which are harder to assess because they only have indirect links to "hard" environmental indicators such as water quality, and depend on a large number of factors. However, the scenario results provide some indication of the issues involved with including these co-benefits. For example, the GAP analysis for Iowa suggests that species richness for mammals, birds, amphibians, and reptiles is higher along the Mississippi and the Missouri Rivers and in southern Iowa (Kane et al.). The Des Moines Lobe area, where it would be more efficient to retire land for carbon sequestration purposes, has lower biodiversity since the land is intensely cropped, and there are few forests, grasslands, and rivers. This indicates that a land set-aside program similar in size to those analyzed here, and designed to preserve biodiversity, would focus largely on the same areas as our water quality scenario.

In terms of policy design, our results support the use of different instruments (programs) to achieve different environmental goals. A weighted index could also be used.

In such a case, because the overlap of benefits can be modest, the weights would essentially reflect the relative priority of each environmental improvement category and its share of the budget.

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Table 1 Simulation results

Policy	Budget, $ million	Area enrolled, Million acres	Carbon sequestered, million metric tons	Erosion reduction (edge of field), million metric tons	Sediment load reduction (in stream), million metric tons	Nitrates load reduction (in stream), thousands metric tons	Total P load reduction (in stream), thousands metric tons
Water quality	242.1	2.2	0.3	16.0	2.2	2.47	1.49
Carbon, equal area	287.0	2.2	3.1	4.0	0.0	10.05	0.81
Carbon, equal cost	242.1	1.9	2.7	3.5	0.0	8.96	0.57

Note that the carbon and edge of field reductions are calculated for the State, while the in-stream water quality measures are the sum of the reductions across the 13 watersheds of Figure 1. Therefore, the numbers are not exactly commensurate.

Table 2 Percentage reduction in sediment, N and P in stream for CRP Water Quality policy

		Sediment	Nitrate	Org N	Total N	Org P	Min P	Total P
1	Floyd	4	-2	3	-1	1	-4	-3
2	Monona	6	-2	10	0	11	7	8
3	Little Sioux	-2	-9	8	-6	13	6	7
4	Boyer	17	0	19	5	19	17	17
5	Nishnabotna	25	0	15	3	15	18	18
6	Nodaway	21	6	20	10	20	19	19
7	Des Moines	3	3	8	5	9	6	7
8	Skunk	20	7	15	8	14	15	15
9	Iowa	1	4	0	2	0	5	1
10	Wapsipinicon	4	-2	-4	-2	-2	-11	-9
11	Maquoketa	11	-2	11	0	9	9	9
12	Turkey	19	1	14	3	14	13	13
13	Upper Iowa	13	2	6	3	0	3	2
	Total	**11**	**1**	**5**	**2**	**4**	**7**	**6**

Table 3 Percentage reduction in sediment, N and P in stream for CRP Carbon (equal cost) policy

		Sediment	Nitrate	Org N	Total N	Org P	Min P	Total P
1	Floyd	0	2	-1	1	-3	-8	-7
2	Monona	0	1	-1	1	0	-3	-2
3	Little Sioux	0	-1	1	0	0	2	2
4	Boyer	2	2	1	2	1	1	1
5	Nishnabotna	-1	-3	0	-2	1	-1	0
6	Nodaway	12	3	12	6	12	12	12
7	Des Moines	0	8	9	9	8	9	9
8	Skunk	5	4	4	4	4	4	4
9	Iowa	1	9	3	6	2	2	2
10	Wapsipinicon	-7	-1	-15	-3	-15	-17	-16
11	Maquoketa	-3	1	-12	-1	-11	-11	-11
12	Turkey	-8	1	-4	0	-4	-4	-4
13	Upper Iowa	1	-1	-1	-1	0	-3	-2
	Total	**0**	**4**	**3**	**4**	**3**	**2**	**2**

Table 4 Simulation results - Break-even price of erosion, $ per metric ton

Policy	Carbon price of 10$ per metric ton	Carbon price of 50$ per metric ton	Carbon price of 100$ per metric ton
Water quality	14.9	14.2	13.3
Carbon, equal area	64.5	33.6	-5.0
Carbon, equal cost	61.4	31.1	-6.8

Table 5 Simulation results- Break-even price of Total P load reduction, $ per metric ton

Policy	Carbon price of 10$ per metric ton	Carbon price of 50$ per metric ton	Carbon price of 100$ per metric ton
Water quality	51,540.3	152,540.4	142,800.2
Carbon, equal area	60,373.0	164,781.7	-24,736.1
Carbon, equal cost	70,488.1	190,620.3	-41,870.7

Table 6 Simulation results- Budget break up (percentage values in parenthesis).

Policy		Carbon price of 10$ per metric ton	Carbon price of 50$ per metric ton	Carbon price of 100$ per metric ton
Water quality	Carbon	2,905,800 (1.2%)	14,529,000 (6.0%)	29,058,000 (12.0%)
	Co-benefit	239,162,090 (98.8.0%)	227,538,890 (94.0%)	213,009,890 (88.0%)
Carbon, equal area	Carbon	30,698,440 (10.7%)	153,492,200 (53.5%)	306,984,400 (107.0%)
	Co-benefit	256,252,000 (89.3%)	133,458,240 (46.5%)	-20,033,960 (-7.0%)
Carbon, equal cost	Carbon	26,602,270 (11.0%)	133,011,350 (54.9%)	266,022,700 (109.9%)
	Co-benefit	215,465,620 (89.0%)	109,056,540 (45.1%)	-23,954,810 (-9.9%)

Fig. 1. Study area and watershed delineations.

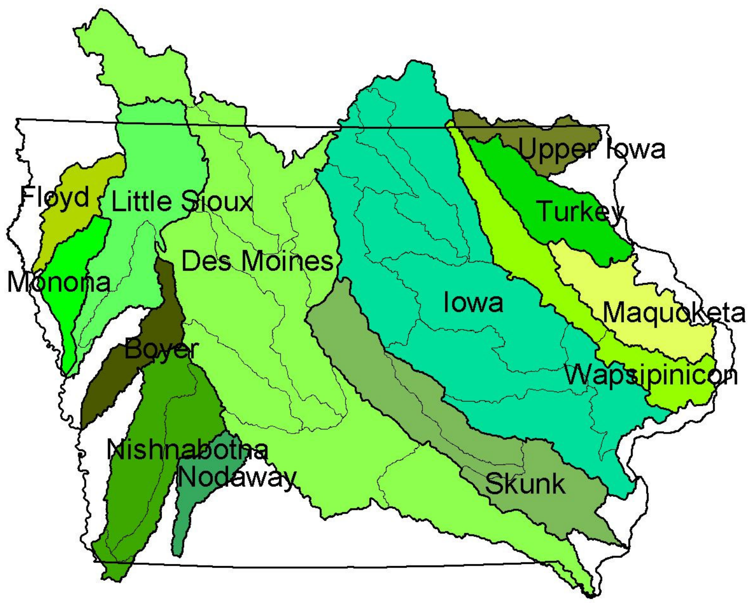


Fig. 2. Set aside acres - Water Quality Policy

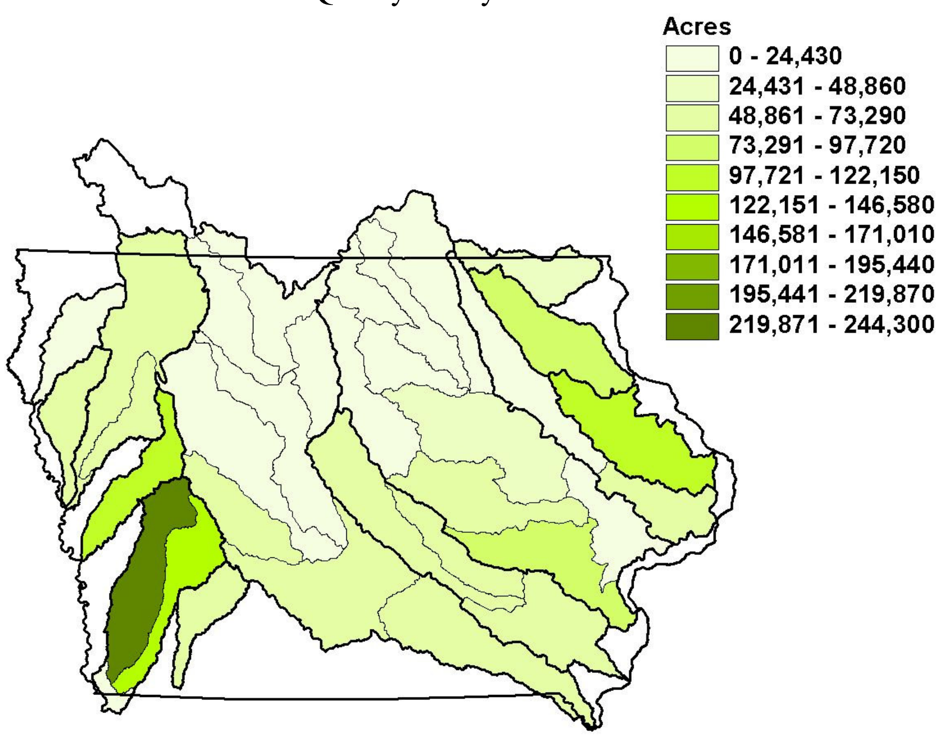


Fig. 3. Set aside acres - Carbon Policy (equal cost)

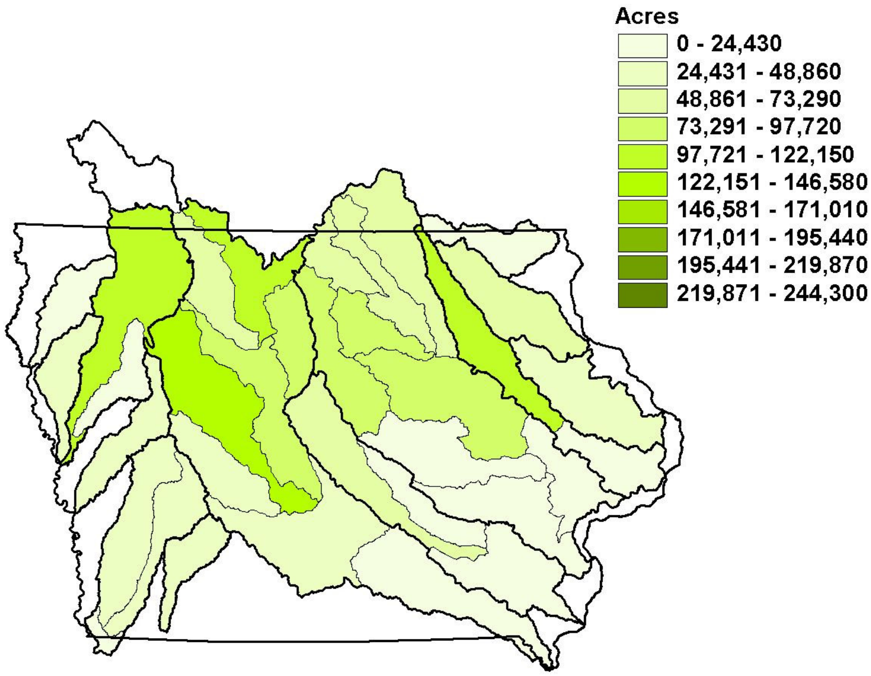


Fig. 4. Set aside acres - Carbon Policy (equal acres)

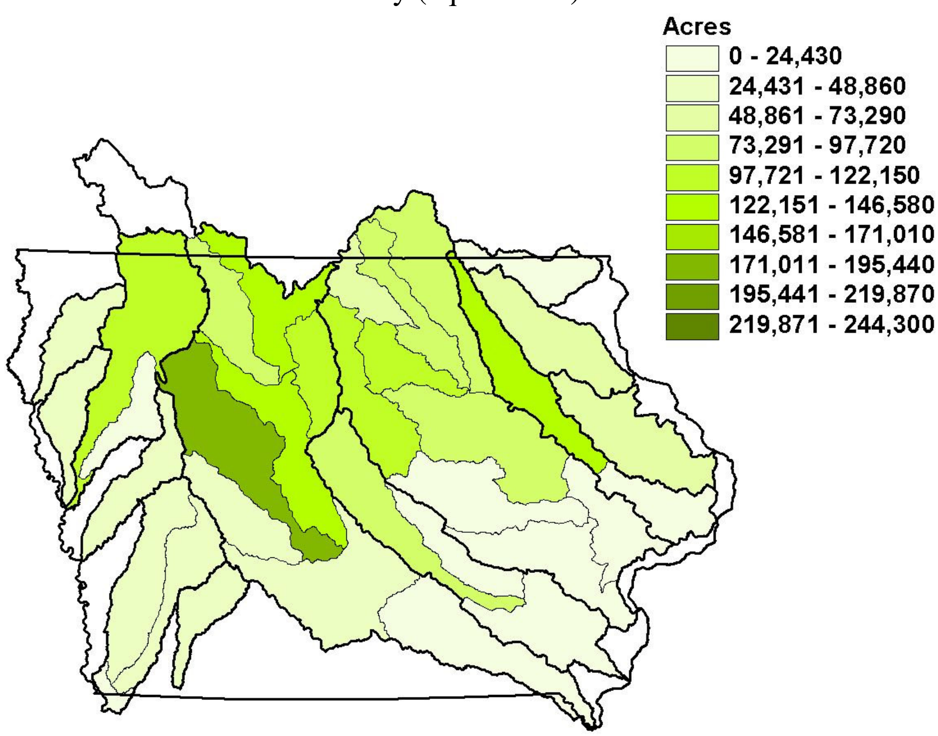


Fig. 5. Carbon sequestration from Water Quality Policy

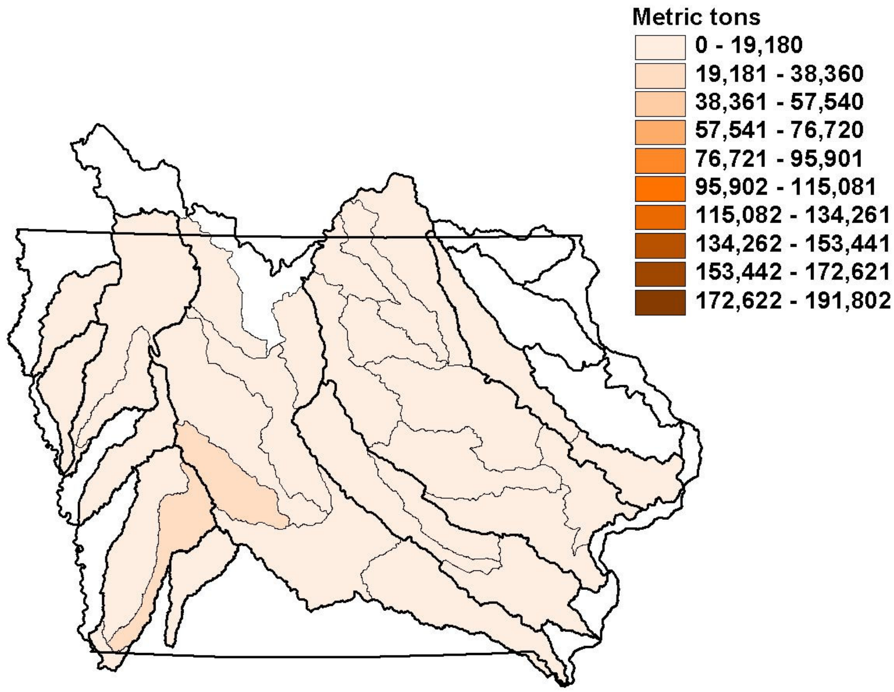


Fig. 6. Carbon sequestration from Carbon Policy (equal cost)

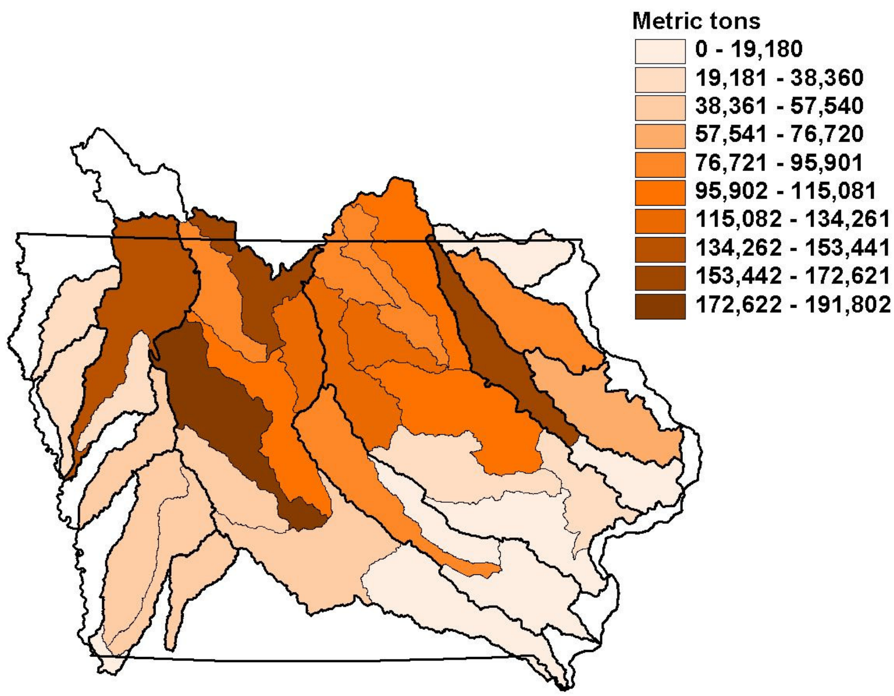


Fig. 7. Edge-of-field erosion reduction from Water Quality Policy

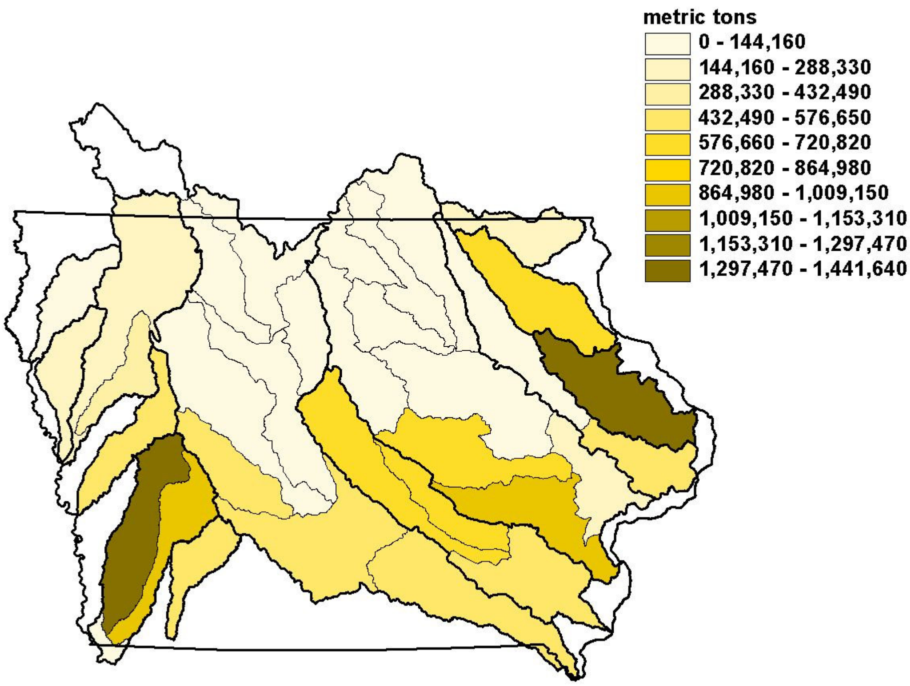
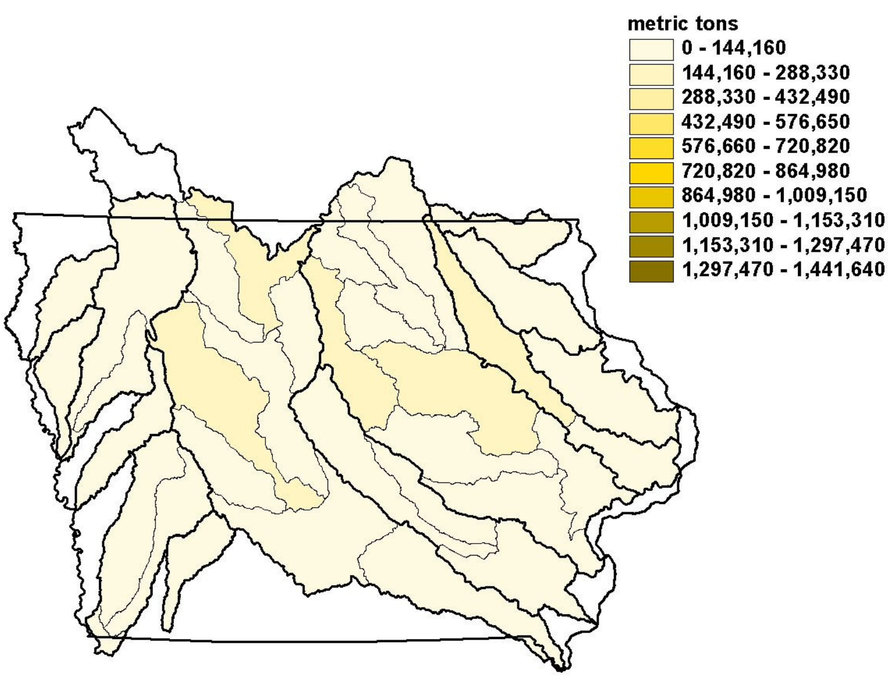


Fig. 8. Edge-of-field erosion reduction from Carbon Policy (equal cost)



Appendix 1 - Baseline loads

BASELINE		Baseline loadings - Annual average values in *Metric tons* (1980-1997)						
		Sediment	Nitrate	Org N	Total N	Org P	Min P	Total P
1	Floyd	241,423	7,125	1,281	8,406	104	301	405
2	Monona	198,589	4,847	757	5,605	58	269	327
3	Little Sioux	632,456	23,851	4,730	28,569	381	1,687	2,067
4	Boyer	777,245	4,947	2,044	6,991	161	548	709
5	Nishnabotna	1,968,399	8,257	2,399	10,656	207	794	1,001
6	Nodaway	520,045	3,312	1,344	4,656	113	405	518
7	Des Moines	2,174,303	38,252	25,098	63,349	2,411	4,664	7,075
8	Skunk	3,800,345	28,122	3,965	32,087	343	1,678	2,021
9	Iowa	3,423,237	54,050	49,688	103,738	7,000	786	7,786
10	Wapsipinicon	2,238,966	27,533	3,950	31,484	342	1,228	1,571
11	Maquoketa	1,218,739	14,781	2,965	17,746	245	674	919
12	Turkey	1,297,814	12,436	2,406	14,842	208	642	850
13	Upper Iowa	730,155	3,675	1,008	4,683	112	267	379
	Total	19,221,717	231,188	101,635	332,811	11,686	13,943	25,629

Allocating Nutrient Load Reduction across a Watershed: Implications of Different Principles

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Allocating Nutrient Load Reduction across a Watershed: Implications of Different Principles

Abstract

A watershed-based model, the Soil and Water Assessment Tool (SWAT), along with transfer coefficients is used to assess alternative principles of allocating nutrient load reduction in the Raccoon River watershed in central Iowa. Four principles are examined for their cost-effectiveness and impacts on water quality: absolute equity, equity based on ability, critical area targeting, and geographic proximity. Based on SWAT simulation results, transfer coefficients are calculated for the effects of nitrogen application reduction. We find both critical area targeting and downstream focus (an example of geographic proximity) can be more expensive than equal allocation, a manifestation of absolute equity. Unless abatement costs are quite heterogeneous across the subwatersheds, the least-cost allocation (an application of the principle of equity based on ability) has a potential of cost savings of about 10% compared to equal allocation. We also find that the gap between nitrogen loading estimated from transfer coefficients and nitrogen loading predicted by SWAT simulation is small (in general less than 5%). This suggests that transfer coefficients can be a useful tool for watershed nutrient planning. Sensitivity analyses suggest that these results are robust with respect to different degrees of nitrogen reduction and how much other conservation practices are used.

Key words: Least-cost allocation, Soil and Water Assessment Tool (SWAT), Transfer coefficients.

1. INTRODUCTION

It is increasingly recognized by the public that nonpoint source pollution is the largest remaining source of water pollution in the US. In agricultural dominated regions, nonpoint source control is primarily aimed at preventing and reducing agricultural pollutants. It is clear that the allocation of loading among nonpoint sources, which has not been a focus of the current implementation of the Total Maximum Daily Load (TMDL) program, will be a critical issue in the search for measures to achieve water quality goals at the lowest cost possible. A variety of criteria can be used to decide where conservation measures should be implemented to control agricultural nonpoint source water pollution.

The first is absolute equity, which often requires that every subwatershed makes the same percentage of load reduction per hectare or per capita. A second often used criterion is equity based on ability. According to this criterion, those with lower marginal costs of abatement are required to make bigger cuts in pollutant load. In addition to being simple, these two criteria also have direct policy relevance—absolute equity can be implemented with a command-and-control type policy, while equity based on ability is consistent with a market-based mechanism such as taxes or permit trading. Geographical proximity is another criteria often used to decide which areas should be considered to share the responsibility of improving water quality in a watershed. For example, conservation measures are sometimes assumed to be implemented in the entire county where the cropland impaired water body lies (USEPA 2006). Finally, it is not unusual that conservation measures are required only in critical areas responsible for a disproportionate share of loading or having most potential for improvement. In this paper, we examine the consequences of these four criteria in terms of water quality benefit and abatement costs.

There are practical challenges in applying some of the criteria mainly because they require the assessment of the impacts of conservation practices on the fate and transport of pollutants at a watershed scale. Nutrient loads discharged from specific source areas can be further impacted by ongoing in-stream processes including deposition or assimilation along the waterway, the input of additional nutrients via riverbank and/or riverbed erosion, or additional nutrient inputs through atmospheric deposition. Moreover, the degree of such deposition and erosion effects can often be affected by what happens in other subwatersheds across the watershed. Thus, to understand the effects of a particular conservation practice adopted in a subwatershed, it is often necessary that we understand the complex hydrologic process in the whole watershed.

Simulation models that are designed to capture such complex hydrologic and pollutant transport processes can be used to aid our understanding. These models range in complexity and can require extensive input data and a high level of expertise in order to perform a successful calibration and validation exercise for a given watershed. Besides, while useful in providing insights on the fate and transport of nutrients, it can be difficult to base policy design directly on these models often because of their complexity.

For both theoretical and empirical policy analyses, transfer coefficient models, which use simple parameters to capture the long term impacts of conservation measures, have been used for a long time. There is a broad literature of nutrient transfer coefficients by land-cover type based on decades of field-based research (Beaulac and Reckhow 1982; Johnes 1996). In this literature, transfer coefficients are usually referred to as export coefficients. As early as the 1970s, economists used transfer coefficients to study how market based mechanisms can be utilized to minimize the cost of pollutant abatement (Montgomery 1972). Recently, Hung and Shaw (2005) showed that a trading-ratio system based on transfer coefficients can achieve the least cost to reach a water quality goal. Khanna et al. (2003) found that incorporating endogeneity in transfer coefficients can have a large impact on the costs of abatement, based on an application of the Agricultural Non-Point Sources Pollution (ANGPS) water quality model to a watershed in Illinois. In the context of local versus regional water pollutants, Kling et al.(2005) examined the extent to which transfer coefficients based on the Soil and Water Assessment (SWAT) model (Arnold et al, 2000; Arnold and Forher 2005) can be used to assess nutrient trading at a large regional scale. There are other simple biophysical measures that can be used to quantify the potential of pollutant loading from a source. For example, the phosphorous index can be used to characterize the potential of a field or region to load phosphorous into surrounding waters (Johansson and Randall 2003).

In this study we use SWAT and transfer coefficients similar to those used in Kling et al. (2005) to assess alternative policy scenarios in a small watershed—the Raccoon River Watershed in Iowa. In the scenarios, the allocation of load reduction responsibility is based on the four principles discussed earlier and on the transfer coefficients derived from the output of SWAT for the Raccoon River Watershed. We then examine how the allocations differ in terms of cost-effectiveness and water quality improvement. By assessing the four principles, we provide some general guidance to researchers and watershed planners in selecting treatment areas. We also contribute to the literature by testing the validity of allocating load reduction responsibility in a small watershed based on transfer coefficients.

2. STUDY REGION

The Raccoon River Watershed drains a total area of about 9397 km^2 in west central Iowa (Figure 1). The land use in the watershed is dominated by agriculture, with about 75.3% in cropland, 16.3% in grassland, and 4.4% in forest. Urban use accounts for the remaining 4.0% of the total area. The Raccoon River and its tributaries drain all or parts of 17 counties before joining the Des Moines River in Des Moines. The watershed is the primary source of drinking water for central Iowa communities including the city of Des Moines which has a population of 200,000.

Intensive agriculture with widespread application of nitrogen fertilizer has been identified as the primary source of high nitrate concentration in the watershed, which is a major concern both locally and regionally. Since the late 1980s, the Des Moines Water Works has operated the world's largest nitrate removal facility, due to the high concentration of nitrate. Sections of the Raccoon River are included in Iowa's Federal Clean Water Act 303(d) list of impaired waters due to the high nitrate levels. Nitrates discharges from the Raccoon and other rivers in the Upper Mississippi River Basin have been implicated as a key source of the Gulf of Mexico seasonal hypoxic zone that has covered upwards of 20,000 km^2 in recent years (Rabalais et al. 2002). The Committee on Environment and Natural Resources (CENR) recommended the implementation of several on-farm practices for reducing discharges of nitrogen to streams and rivers. Among these practices is a 20% reduction in nitrogen fertilizer application (Mitsch et al. 1999).

3. MODELING FRAMEWORK

The SWAT model is a conceptual, physically based long-term continuous watershed scale simulation model that operates on a daily time step. In SWAT, a watershed is divided into multiple subwatersheds, which are then further subdivided into

Hydrologic Response Units (HRUs) that consist of homogeneous land use, management, and soil characteristics. Key components of SWAT include hydrology, plant growth, erosion, nutrient transport and transformation, pesticide transport, and management practices. Detail theoretical description of the SWAT model and its major components can be found in Neitsch et al. (2002). Outputs provided by SWAT include streamflows and in-stream loading or concentration estimates of sediment, nutrients, and pesticides. Previous applications of SWAT for streamflows and/or pollutant loadings have compared favorably with measured data for a variety of watershed scales (e.g., Arnold and Allen 1996; Arnold et al. 1999; Arnold et al. 2000; Santhi et al. 2001; Borah and Bera 2004).

This study is based on the SWAT modeling framework developed by Jha et al. (2006), who calibrated and validated SWAT for streamflow, sediment loads, and nitrogen and phosphorus losses for the Raccoon River Watershed. This framework facilitates analyses of the impacts of potential policy scenarios on flow, sediment and other water quality indicators in the region. Basic input data used to set up the SWAT simulation include topography, weather, land use, soil, and management data. A key source of land use, soil and management data was the National Resources Inventory (NRI) database (Nusser and Goebel 1997). The NRI is a statistically based survey database that contains information for the entire U.S. such as landscape features, soil type, cropping histories, tile drainage, and conservation practices for the whole nation. The climate data were obtained from the National Climatic Data Center for 10 weather stations located in and around the watershed. In the modeling framework, the watershed is delineated into 26 subwatersheds identical to the 10-digit level of Hydrologic Unit Codes. The outlet of subwatershed 25 is also the outlet of the whole Raccoon River watershed (Figure 1).

4. THEORETICAL POLICY ANALYSIS—AN APPROXIMATE LEAST-COST ALLOCATION OF NUTRIENT ABATEMENT

Suppose there is a goal of reducing nutrient loading at the watershed outlet by $\bar{N}$ kilograms for a watershed divided into J subwatersheds. Let the cost of nutrient application reduction be $C_j(N_j A_j)$, where N_j is the nutrient application reduction in kilograms per hectare, and A_j is the total hectares in subwatershed j. The effect of nutrient application reduction at all subwatersheds (i.e., the total nutrient loading reduction at the watershed outlet) is represented by a function $f(N_1A_1, N_2A_2,..., N_JA_J; \boldsymbol{w})$, where $\boldsymbol{w}$ represents other land use characteristics and natural elements such as weather. The function $f(\cdot)$ reflects the complex hydrologic process in the watershed, taking into account the possibility that the effect of nutrient application reduction at one subwatershed can depend on the characteristics of the whole watershed and the action taken at other subwatersheds. We can write the total nutrient standard as

$$f(N_1A_1, N_2A_2,..., N_JA_J; \boldsymbol{w}) \leq \bar{N} \tag{1}$$

Then the following problem can be set up to find the least-cost allocation of nutrient application reduction to meet the nutrient standard at the watershed outlet:

$$\min \sum_{j=1}^{J} C_j(N_jA_j) \tag{2}$$

subject to (1).

The solutions can be characterized as

$$\frac{\partial C_j(N_j^*A_j)/\partial N_j}{\partial f(N^*{}_1A_1, N^*{}_2A_2,..., N^*{}_JA_J; \boldsymbol{w})/\partial N_j} = \frac{\partial C_k(N_k^*A_k)/\partial N_k}{\partial f(N^*{}_1A_1, N^*{}_2A_2,..., N^*{}_JA_J; \boldsymbol{w})/\partial N_k}; \tag{3}$$

where $\partial C_j(N_jA_j)/\partial N_j$ represents the marginal cost incurred from an incremental change in N_j and $\partial f(N_1A_1, N_2A_2,..., N_JA_J; \boldsymbol{w})/\partial N_j$ represents the marginal benefit, i.e., the extra loading reduction achieved from an incremental change in N_j. Equation (3) requires that the ratio of marginal benefit over the marginal cost be equalized to achieve the least cost allocation.

It is difficult to apply equation (3) to allocate nutrient application reduction in a watershed mainly because $f(N_1A_1, N_2A_2,..., N_JA_J; \boldsymbol{w})$ represents a complex hydrological process. The denominator, $\partial f(N_1A_1, N_2A_2,..., N_JA_J; \boldsymbol{w})/\partial N_j$ can vary with N_i for any $i = 1, 2,..., J$. In this paper, we explore an approximate form of $f(N_1A_1, N_2A_2,..., N_JA_J; \boldsymbol{w})$ which is much simpler and thus has the potential of being utilized in reality. We then examine whether allocations based on the simplified version can be used to achieve water quality standards at the least cost.

Specifically, we consider a linear approximation of $f(N_1A_1, N_2A_2, ..., N_JA_J; \boldsymbol{w})$, i.e.,

(4) $$f(N_1A_1, N_2A_2, ..., N_JA_J; \boldsymbol{w}) = \sum_{j=1}^{J} d_jN_jA_j$$

where transfer coefficient (d_j) is an approximation of the amount of nutrient reduction at the watershed outlet achieved by one unit of nutrient application reduction in subwatershed *j*.

In order to explicitly solve the problem represented by expression (2), we assume that $C_j(N_jA_j) = A_jc_j(N_j)$, with

(5) $$c_j(N_j) = \alpha_0 + \gamma_j\alpha\frac{\theta}{\theta+1}N_j^{\frac{\theta+1}{\theta}}.$$

The parameters α_0, $\alpha > 0$, $\theta > 0$, and $\gamma_j > 0$, control the scale, shape, and heterogeneity of the abatement cost function in the subwatersheds. In the next section, we provide more discussion on the cost function. With (4) and (5), we can derive a closed form solution for the problem in (2) as follows:

(6) $$N_j^* = \frac{\bar{N}d_j^{\theta}\gamma_j^{-\theta}}{\sum d_i^{\theta+1}\gamma_i^{-\theta}A_i}.$$

Thus, the optimal nitrogen application reduction in subwatershed j depends on the transfer coefficients and cost parameters in all subwatersheds. The solution in equation (6) is very fortuitous for our empirical analysis in that we do not need to know the precise size of the abatement cost function in order to allocate nutrient load, because α_0 and α do not appear in equation (6). As far as abatement cost is concerned, we only need to know the shape of the cost function as represented by θ and the relative magnitude of the cost across the subwatersheds as represented by γ_j. This not only facilitates our empirical analysis but also is very important in the real world given that the exact magnitude of cost for nutrient reduction can be hard to obtain.

5. EMPIRICAL POLICY ANALYSIS

In our empirical analysis, we focus on the reduction of nitrogen fertilizer application on cropland, which as mentioned earlier is a recommended practice by the NCER. We examine the effectiveness of four strategies, which are applications of the four principles we discussed in the introduction section, to reduce the application of nitrogen fertilizer. For the first strategy, an example of absolute equity, the fertilizer reduction is reduced by the same percentage across all subwatersheds. In the second strategy, the allocation of nitrogen application reduction for the subwatersheds is based on their marginal benefits and marginal costs and is determined by equation (6). This is an application of the equity based on ability principle in the following sense: those that can reduce nutrient load at relatively low costs are required to have larger nitrogen application reductions. In the other two strategies, nitrogen application reduction is only required for roughly half of the subwatersheds which are: (a) located in the downstream (lower) reaches of the watershed, or (b) found to be the most effective treatment areas in terms of nutrient reductions as measured by higher transfer coefficients. These two strategies are manifestation of principles based on geographic proximity and critical areas. In all strategies, we assume that the total nitrate load reduction is the same as estimated from the transfer coefficients. (We choose nitrate as our nutrient indicator because it is the predominant form of nitrogen pollution in water in the study region.) We then examine the cost-effectiveness of these strategies and the nitrate loading reduction achieved based on SWAT simulations, as opposed to transfer coefficients.

In our empirical analysis, we use the following procedure to derive the transfer coefficients (d_j):

1. Conduct one SWAT run: assuming no reduction at all in the watershed, obtain the baseline nitrate loadings at the watershed outlet.
2. Conduct 26 SWAT runs: assuming *x* percent nitrogen fertilizer application reduction in subwatershed *j* and 0% reduction at all other subwatersheds. Denote the amount of nitrate loading reduction obtained at the watershed outlet as y_j.
3. Transfer coefficient d_j is then defined as

$$d_j = \frac{y_j}{x * (\text{Baseline nitrogen fertilizer application at sub-watershed j})}.$$

In addition to the baseline simulation and the simulations performed to derive the coefficients, we also perform four additional SWAT simulations for the four strategies of allocating nitrogen fertilizer application reduction: (i) reduction in each subwatershed by the same percentage, say 20%; (ii) reduction in each subwatershed based on equation (6); (iii) reduction in only 14 downstream subwatersheds; and (iv) reduction in only 13 subwatersheds with the highest transfer coefficients. For the runs in (ii)–(iv), it is assumed that the nitrate loading reduction estimated from the transfer coefficients is the same as that achieved in (i). The issue we want to explore is how much costs differ among the four strategies and whether the same nitrate load reduction will be achieved, given that the transfer coefficients are only based on approximate estimates of nitrate loading impacts.

In order to compare the cost of these scenarios, it is important that we understand the cost of nitrogen application reduction. The estimation of the yield effect is the most important and probably the most controversial and unresolved issue in costing the reduction of nitrogen fertilizer application due to a number of factors. The yield effect of a moderate reduction in nitrogen fertilizer application has been estimated to be almost none, positive, or negative. Some states still recommend more fertilizer for a higher yield goal, while others have discontinued the practice (Lory and Scharf 2003). It is difficult to estimate the impacts of fertilizer application because the effects may be masked by weather, previous crops, soil condition, etc. Moreover, the reduction of fertilizer may have an insignificant effect in the short run; however, the long run effect may be large. In addition to the issues related to yield effects, Babcock (1992) also showed that the seemingly over-application of nitrogen fertilizer is actually consistent with profit maximization, which implies that a payment will be needed for farmers to reduce their nitrogen fertilizer application.

Given the diverse opinions on the cost of nitrogen fertilizer reduction, we adopt a cost function with flexible shape and scale. In $c_j(N_j) = \alpha_0 + \gamma_j \alpha \frac{\theta}{\theta+1} N_j^{\frac{\theta+1}{\theta}}$, all of the parameters can be calibrated to the abatement cost in a particular watershed. Parameters α_0 and α determine the scale of the cost function. Since α_0 and α do not appear in (6), the optimal allocation of nutrient loading reduction does not depend on these two parameters. The parameter θ determines the curvature of the cost function—the smaller the θ, the faster the cost increases as N_j increases. For a very large θ, the cost function is approximately linear in N_j. The heterogeneity of the cost function among the 26 subwatersheds is reflected by γ_j. If $\gamma_j = 1$ for all *j* then the cost function is the same for all of the subwatersheds.

A sensitivity analysis is conducted with respect to different values of the parameters θ and γ_{j}. In addition, the transfer coefficients may also be sensitive to other conservation measures adopted in the watershed. So we also derive transfer coefficients for all of the subwatersheds assuming no-till is adopted in all cropland. To investigate the sensitivity of transfer coefficients to different amounts of nitrogen fertilizer application reduction, three reduction levels are considered: 10, 20, and 30%. In the rest of the paper, we will call these scenarios the 10% scenarios, the 20% scenarios, and the 30% scenarios, respectively. In 10% scenarios, the transfer coefficients are based on a 10% nitrogen fertilizer application reduction and the nitrate reduction goal is the nitrate loading reduction achieved by a 10% nitrogen fertilizer application reduction in all 26 subwatersheds. For the 20% and 30% scenarios, similar logic applies.

6. RESULTS

Baseline average annual nitrate loadings for each subwatershed are presented in Table 1 as a function of the total land area, corn production area, and baseline nitrogen application rates. The results also show that the loading predicted at each subwatershed outlet varies substantially with the relative location of the subwatersheds in the whole Raccoon watershed as shown in Figure 1. The average corn area accounts for about 50% of total area, which is consistent with the fact that corn-soybean is the dominant rotation in the watershed. The fertilizer application rates in the region, which are assumed based on state and county fertilizer use information, are quite homogeneous with a mean of 148 kg/ha and a standard deviation of 4.7 kg/ha.. The 24-year (1981–2003) baseline simulation run of the calibrated SWAT model resulted in an annual average nitrate load of 15,200 tons at the watershed outlet (i.e., subwatershed 25 outlet). As in the baseline, SWAT simulations for all scenarios are performed over the same period (1981–2003) and then annual average output is used for nitrate load.

6.1. The fate and transport of nitrogen in the watershed

We present a schematic diagram of the Raccoon watershed (Figure 2) to highlight the connection and interactions among the 26 subwatersheds. The dark dots and gray circles represent the subwatersheds. The seven subwatersheds represented by the gray circles receive flow and nitrate from two or more upstream subwatersheds. Of the remaining subwatersheds, two have one upstream subwatershed and the others have no upstream subwatersheds.

The transfer coefficients capture the fate and transport of the nitrate losses, which are the foundation of this study for allocating nitrogen fertilizer reduction in the Raccoon River Watershed. Thus, we first present the transfer coefficients for all three levels of nitrogen application reduction in Figure 3. The average of transfer coefficient is about 0.23 for the 10% scenario, which means that for every 1 kg of nitrogen fertilizer application reduction about 0.23 kilograms of nitrate reduction is achieved at the watershed outlet. It is clear from Figure 3 that the transfer coefficients are almost the same for the 20 and 30% scenarios. This is good news for policy design in that watershed planners can use the same set of transfer coefficients to allocate loading reduction responsibilities regardless of the degree of reduction. By examining Figures 2 and 3, we see that there is not a clear pattern as to how the transfer coefficients vary with the location of a subwatershed. Some upstream subwatersheds have relatively high transfer coefficients (e.g., subwatershed 3), whereas some downstream subwatersheds have relatively low transfer coefficients (e.g., subwatershed 23).

To examine whether the transfer coefficients are sensitive to tillage practices, we derive transfer coefficients when no till is adopted on all cropland. The average of the new transfer coefficients is 0.26, which is slightly higher than the transfer coefficients presented in Figure 3. The underlying reason is that when no till is adopted, there is more nitrate runoff. Thus, even though the fertilizer reduction would be about as effective in terms of percentage reduction as in the case with baseline tillage practices, the reduction of loading in terms of kilograms increases.

6.2. Comparison of the four principles

For downstream targeting, about half of the subwatersheds were designated as downstream subwatersheds as shown by the subwatersheds inside the gray loop in Figure 2. It is easy to see that there is no clear advantage of targeting these subwatersheds as shown by the distribution of the transfer coefficients (Figure 3). This underscores the fact that reducing nitrogen applications in those subwatersheds that lie in close proximity to the watershed outlet does not necessarily imply that more effective reductions of nitrate loading will occur at the watershed outlet. To further illustrate this point, consider a downstream targeting scenario where the target is set the same as the corresponding equal percentage reduction scenario. Then, when the abatement cost increases fast (e.g., setting $\theta = 1$), the total cost for downstream targeting can be twice as expensive as the equal percentage reduction scenario. However, when the abatement cost is closer to being linear (e.g., setting $\theta = 5$), the cost difference between the two scenarios would be reduced dramatically to a few percentage points.

For critical area targeting, we assume that those subwatersheds that have transfer coefficients greater than the median should be managed with reduced nitrogen fertilizer applications. How much more effective this criterion is compared to the equal percentage reduction scenario also depends on how fast the cost increases. If the cost function is close to being linear, then the critical area can be more cost effective than equal allocation. Concentrating reduction responsibility in a small set of subwatersheds can be more expensive, when the abatement cost increases fast. This is because this small set of subwatersheds has to make larger fertilizer application reductions which are becoming increasingly more expensive. In our simulation, for $\theta = 5$ and $\theta = 3$, critical targeting is slightly cheaper than the equal percentage reduction scenario. However, at $\theta = 1$, critical targeting is actually 34% more expensive.

By definition, the least cost criterion has the lowest cost among the four criteria. Based on this criterion, subwatersheds with higher transfer coefficients and/or lower marginal cost will require a larger reduction in nitrogen application. Figure 4 is one illustration of the least-cost nitrogen application reduction across all the subwatersheds. Even though the curves are quite flat overall in the figure, the zigzagged pattern is obvious, which is in contrast with the equal percentage reduction scenario. As in the downstream and critical area targeting scenarios, how much cost saving potential there is depends on the characteristics of abatement cost in the watershed: the curvature of the cost function and how heterogeneous cost is between the subwatersheds. The greater the heterogeneity there is and/or the more linear the cost function is, the more potential for cost saving.

In Table 2, there are three panels, each of which presents the cost and nitrate loading reduction under the least cost scenario compared to the equal percentage reduction scenario. We will discuss the numbers in the second row of the panels in the next sub-section. From panel A we see that for $\theta = 1$ (cost increases relatively fast) the cost saving is small; only about 5% for all reduction levels. However, panel B shows that for slower rising costs the potential for cost savings can be as high as about 11.5%. Such cost saving is quite modest compared to the SO2 permit trading program which has a cost saving estimated at about 40% relative to "command and control" regulations (Carlson et al. 2000). For panel B as well as panel C and the rest of this section, sometimes only one of the three percentage reduction scenarios is discussed or presented to avoid clutter. The results for other scenarios are similar.

Heterogeneity in cost and benefit is a main reason for cost savings from a least-cost program (Newell and Stavins 2003). Intuitively, if every subwatershed has the same cost and the same transfer coefficient, equal percentage reduction would achieve the least cost. Thus the heterogeneity of cost functions across the subwatersheds can have a large effect on the potential gains from implementing a least-cost program such as permit trading. We examined three scenarios in order to evaluate the effects of heterogeneity in the cost function. In the first one, there is no heterogeneity, that is every subwatershed has the same cost function (in mathematical terms, $\gamma_j = 1$ for all j). In the second scenario, there is some heterogeneity and γ_j is drawn from a transformed Beta distribution with a sample mean of about 3.5 and a standard deviation of about 0.8. In the last scenario, there is more heterogeneity—γ_j is also drawn from a similarly transformed Beta distribution with a sample mean about the same size but a standard deviation about 75% larger. Consistent with the literature, Panel C of Table 2 illustrates that as heterogeneity increases, the gain from trading also increases.

6.3. The implication of approximate transport function on nutrient load allocation

In the above scenarios, all allocations were designed to achieve the same nitrate loading reduction as the corresponding equal percentage reduction scenario. However, the loading reduction is estimated based on transfer coefficients which are only a simplified representation of a complex hydrologic process. Thus, for all of the scenarios, we use SWAT to simulate the nitrate load reduction resulting from a reduction of applied nitrogen fertilizer. We then compare the simulated loading reduction with the loading reduction based on transfer coefficients.

In the 20% case, an equal percentage reduction in nitrogen application in all subwatersheds reduces nitrate loading at the watershed outlet by 17.13%, based on the SWAT estimates. If only the downstream subwatersheds are required to reduce nitrogen applications, such that the nitrate loading reduction based on the transfer coefficients would be 17.13%, then the SWAT simulated nitrate reduction is 15.42% which is about 10% less than the impacts of the equal percentage reduction scenario. Similarly, if only subwatersheds in the critical areas are required to reduce nitrogen application, such that the nitrate loading reduction based on the transfer coefficients is 17.13%, then the actual simulated nitrate reduction at the outlet is only slightly (5.8%) lower than the achievement in the equal percentage reduction scenario. These differences seem to be relatively small, compared to the gaps among the scenario costs that we discussed in the previous subsection.

For the least cost scenarios, the numbers in the second row of the panels in Table 2 are the percentage differences of nitrate loading reduction at the watershed outlet between the equal percentage reduction scenario and the scenario indicated by the column names. It is clear that all of the least cost scenarios achieve lower nitrate loading reduction than the equal percentage reduction scenario. However, the differences are quite small, especially compared to the corresponding differences in costs.

7. CONCLUSIONS

Allocation of pollutant reduction responsibilities between subwatersheds at reasonable costs is a key factor in achieving water quality goals for TMDL and other watershed-based water quality initiatives. In this study, we report the results of SWAT simulation assessments focused on four common and important principles that can be used to allocate nitrogen application reductions within a watershed. We found that it can be more expensive to obtain similar nitrogen loading goals if only downstream subwatersheds or areas considered more effective in reducing nitrogen loading are required to reduce nitrogen fertilizer applications. We also find that, contrary to the popular belief and the experience from the sulfur permit trading program, least cost allocations do not necessarily imply significant cost savings in our study area. Large cost savings (greater than 25%) only occur when the abatement cost is sufficiently heterogeneous. This has important implications in that each watershed may have its own characteristics; what is a cost-saving plan for one watershed may actually increase costs for other watersheds when the cost of planning and implementation is taken into account.

We also tested the idea of using transfer coefficients based on SWAT simulations for allocating nutrient loading in a watershed. In our study region, we find that the loading estimates based on the coefficients are close to the model-simulated loadings with a difference of only a few percentage points in general. This result is encouraging for watershed planners and for the TMDL process. This is because watershed planning can then be based on the transfer coefficients, which indicate the relative effectiveness of a practice implemented in the subwatersheds. However, while important, generalizing this result to other watersheds needs to be carefully evaluated.

Table 1 Baseline description of Raccoon River Watershed at subwatershed level.

Subwatershed	Area (hectares)	Corn (% of total area)	Nitrogen Fertilizer (kg/ha)	Nitrate loading (1000 kg)
1	90000	50.2	148.8	1,600
2	68000	49.9	146.1	1,700
3	22000	50.3	145.6	500
4	54000	49.7	145.6	6,100
5	23000	47.7	161.1	400
6	38000	53	147.2	700
7	33000	48.2	156.3	900
8	19000	54.6	147.9	400
9	39000	50	152.4	10,600
10	42000	50.2	145.6	800
11	44000	51.2	145.6	1,200
12	35000	55.1	137.5	1,200
13	19000	47.3	152.3	600
14	18000	50	153.1	200
15	48000	49.5	147.4	11,700
16	65000	55.1	150.0	300
17	32000	49.3	148.8	300
18	30000	53.1	145.6	800
19	30000	50.9	148.0	600
20	28000	45.3	145.6	900
21	36000	48.9	145.5	300
22	37000	50.7	145.6	400
23	26000	52.1	145.6	2,800
24	17000	54.1	153.2	200
25	26000	54.4	145.6	15,200
26	21000	51.3	141.7	300

Table 2 Comparison between the approximate least-cost scenario and the equal percentage reduction scenario—sensitivity to alternative cost structures.

A. Sensitivity to different degrees of nitrogen reduction
($\theta = 1$, *same cost function for all subwatersheds*)

	10%	20%	30%
Cost difference (%)	-5.00	-5.64	-4.88
Loading difference (%)	0.00	-0.82	-0.77

B. Sensitivity to the curvature of the cost function
(*10% reduction scenario, same cost function for all subwatersheds*)

	$\theta = 1$	$\theta = 3$	$\theta = 5$
Cost difference (%)	-5.00	-8.77	-11.47
Loading difference (%)	0.00	-1.46	-2.96

C. Sensitivity to heterogeneity in the cost function
(*20% reduction scenario,* $\theta = 1$)

	No heterogeneity	Some heterogeneity	More heterogeneity
Cost difference (%)	-5.64	-10.13	-26.12
Loading difference (%)	-0.82	-3.09	-1.56

Fig. 1. Location of the Raccoon River Watershed in Iowa and the delineated subwatersheds.

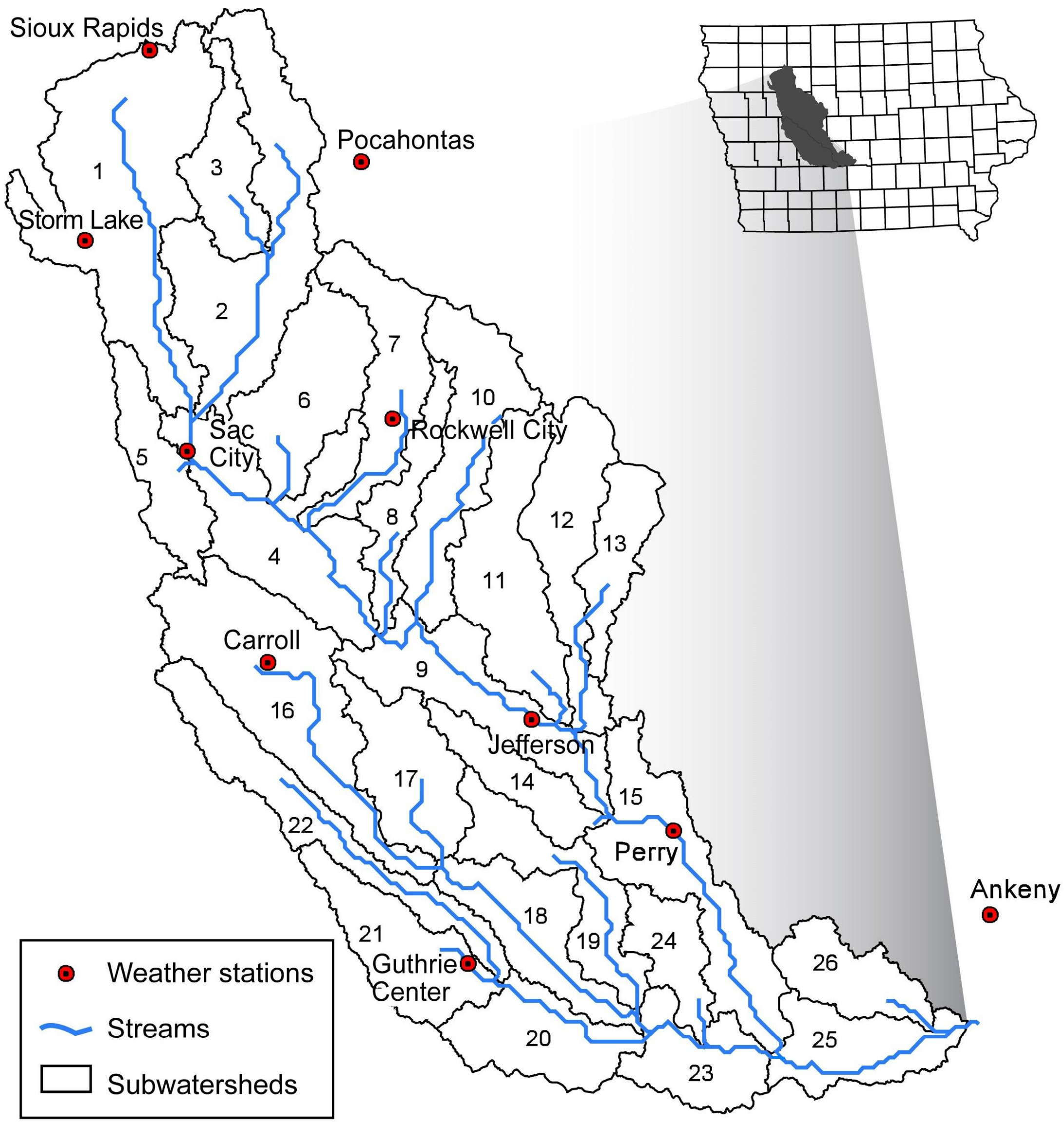


Fig. 2. A schematic diagram of the 26 subwatersheds (with designated downstream subwatersheds used in analysis)

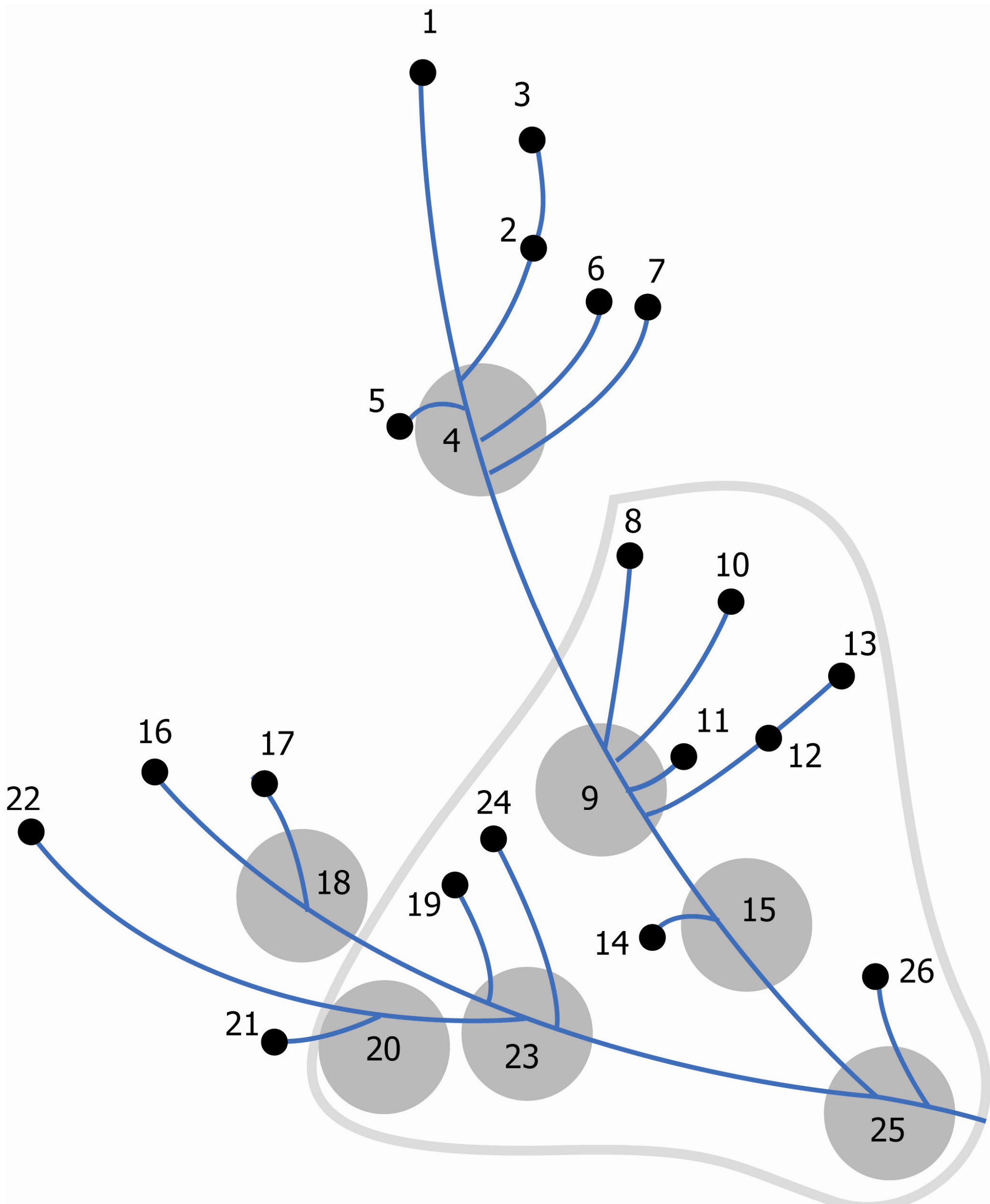


Fig. 3. Transfer coefficients by the 26 subwatersheds.

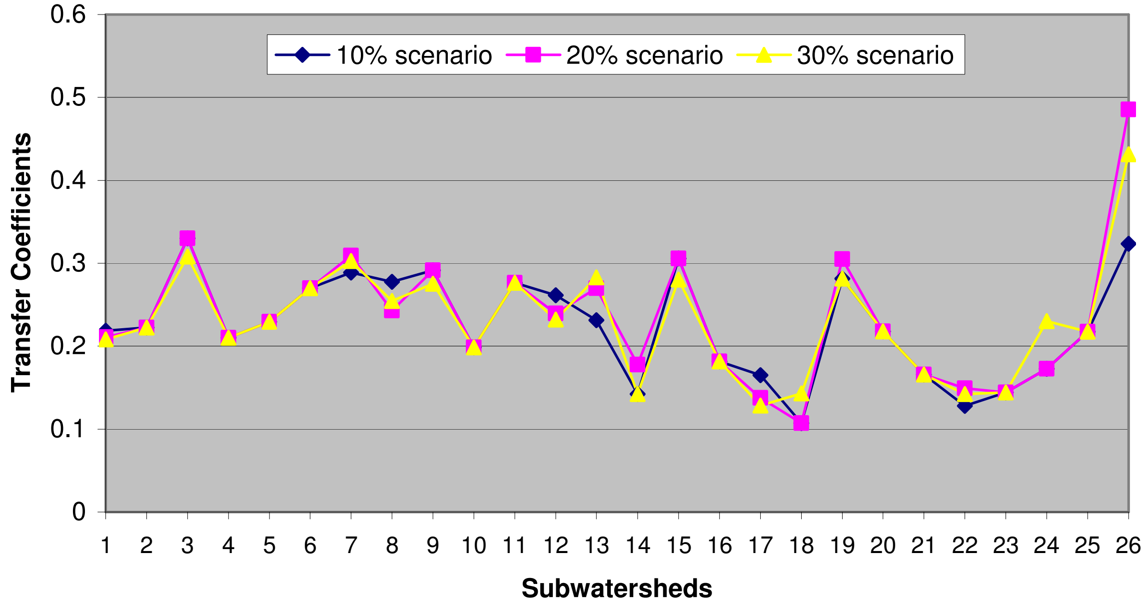
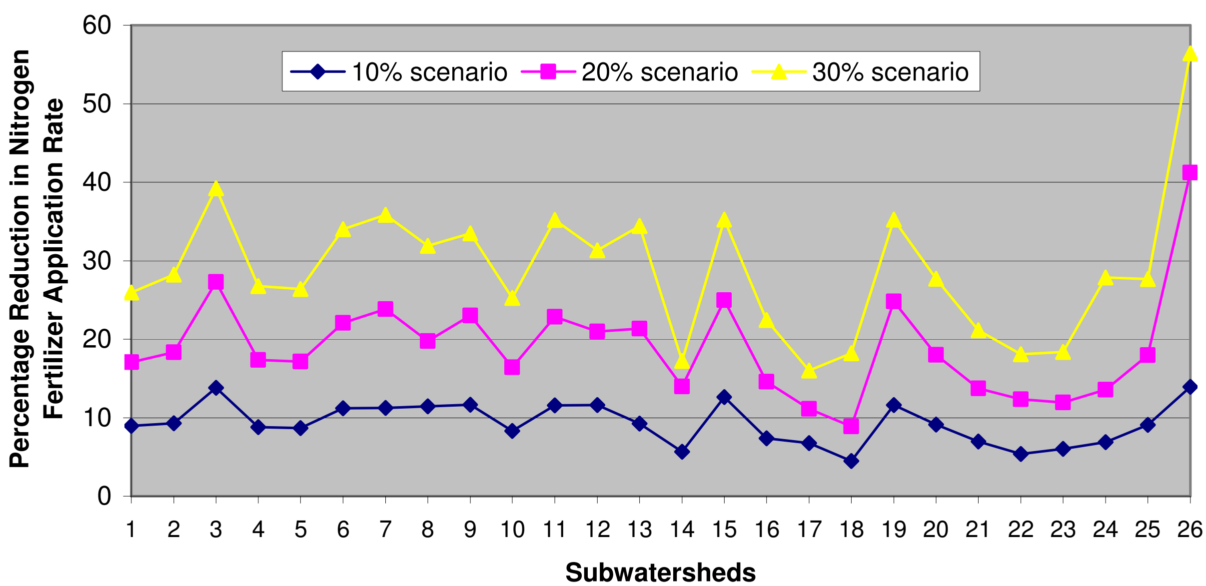


Fig. 4. The distribution of nitrogen fertilizer application reduction in an approximate least-cost scenario (For $\theta = 1$ and $\gamma = 1$).



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Sensors for CyberEngineering: Monitoring and Modeling the Iowa River for Nutrients and Sediments

Basic Information

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Sensors for CyberEngineering: Monitoring and Modeling Clear Creek Nutrients and Sediments

Jerald L. Schnoor, Professor, J.V. Loperfido

Problem and Research Objectives

The initial goal of this research is to use environmental cyberinfrastructure (sensors, servers, models and high performance computing) to construct an environmental cyberinfrastructure platform to analyze water, sediment, and nitrogen transport at the hillslope scale in an important agricultural setting, Clear Creek. The watershed which drains into Clear Creek covers approximately 267 km^2 and is subject to high amounts of erosion. It is also a highly agricultural area as 60% of the watershed is covered by beans and corn while another 19% is covered by pasture and hay. This watershed is where we are building a Clear Creek Environmental Field Facility. During the first year of the research, we deployed sediment and water quality sensors, constructed a digital database of the watershed, and we plan to integrate spatially-distributed models to improve our ability to forecast (in near real-time) critical events that transport water, sediment, and nitrate at the hillslope scale. We have begun to instrument, observe, test, and retrieve remotely-sensed information, and we will link it to numerical models of these processes. In the second year, we will include nitrate sensors (2), pressure transducers for real-time point flow measurements, and incorporate a remote data collection platform created by Dr. Anton Kruger, IIHR University of Iowa, into our monitoring network.

Another important goal of this proposal is to bring together key researchers in Environmental Engineering and the Hydrologic Sciences at the University of Iowa (UI) to create the first node of an Environmental Hydrologic Observatory (EHO) for the Upper Mississippi River Basin (UMRB). The research contributes to the understanding, analysis and modeling of water, sediment, and nitrogen transport in conjunction with Best Management Practices on row crop, barren ground, and fallow fields. Data and models are being linked in a real-time environmental cyberinfrastructure (CI) platform to test capabilities and limitations of the existing sensors for inclusion in the overall Environmental Hydrologic Observatory. As the EHO is built and grows, additional processes will be added and analyzed in real-time, and these elements of the observatory will become embedded into the backbone of the overall observatory. Likewise, new investigators and agencies will be added to enhance the research results as the scale of the processes investigated by the EHO grows beyond this proposal. Iowa Water Center funding for the second year will allow us to install new nutrient sensors and to integrate our real-time streaming data with water quality models.

Method Development

Environmental Monitoring Station System Components

The first step to creating a cyberinfrastructure is acquiring the necessary components. For the initial environmental monitoring stations we purchased three water quality data sondes from Hydrolab and two remote data collection platforms purchased from Campbell Scientific, Inc. The data sondes are used to take water quality measurements while the data collection platform is used to collect these measurements and send them to the base station computer in the lab.

The data sonde we deployed is the Hydrolab DS5X Water Quality Multiprobe (Hach, Loveland, Colorado) which has the capability to measure chlorophyll *a*, conductivity, dissolved oxygen, pH, temperature and turbidity (Fig. 1). It also contains a sweeper that removes biomass from several of the sensors. This is done immediately before a measurement is taken in order to help reduce the effects of biofouling. The data sonde is connected to the data logger by a Hydrolab SDI-12 Adapter (Hach, Loveland, Colorado, Cat. No. 013510) and a Hydrolab Calibration Cable (Hach, Loveland Colorado, Cat. No. 013470). The total cost per Hydrolab DS5X data sonde is $11,468 (see Appendix).

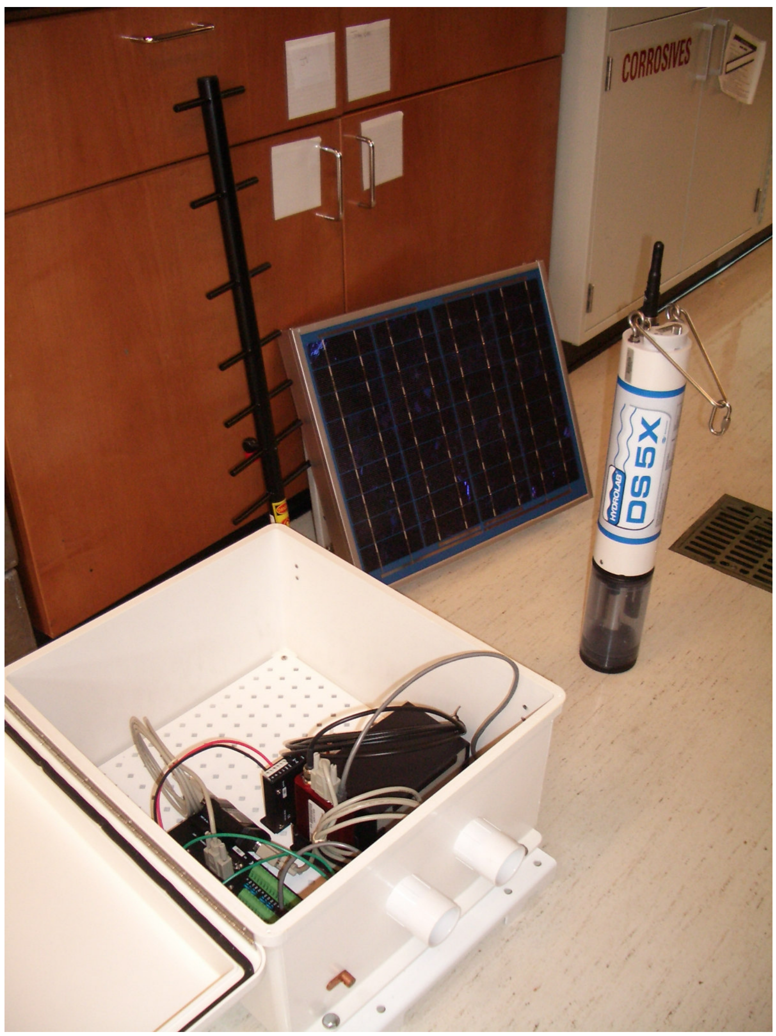

Fig. 1. Environmental monitoring station hardware components

The remote data collection platform purchased from Campbell Scientific, Inc contains components. A cost breakdown for one of these platforms is shown in the Appendix. The data logger used for this system is the CR1000 Measurement and Control System (Campbell Scientific Inc., Logan, Utah). This unit is capable of telling the sensor to take a measurement and collect that measurement data on its hard drive. The data logger will collect and store subsequent measurements until the LoggerNet software downloads it to the base station computer in the lab.

In order for data to be downloaded onto the base station computer from the datalogger, it must be transferred through the telemetry hardware in the remote collection platform. The Redwing 100 CMDA Modem will receive data from the CR1000 and send it through the Verizon Cellular Network (Verizon, Elgin, Illinois) by way of a YA Series Yagi Directional Antenna (Radiall/Larsen Antenna Technologies, Vancouver, Washington).

The data logger and modem are powered by a NP12 Rechargeable 12-Volt Lead Battery (Campbell Scientific Inc., Logan, Utah) and a BP SX20U Solar Cell (British Petroleum, London, United Kingdom) via a CH100 12 Volt Charger/Regulator (Campbell Scientific Inc., Logan, Utah). A schematic of the environmental monitoring station and how data is transferred to the base station in the lab is shown in Figure 2Fig. 2.

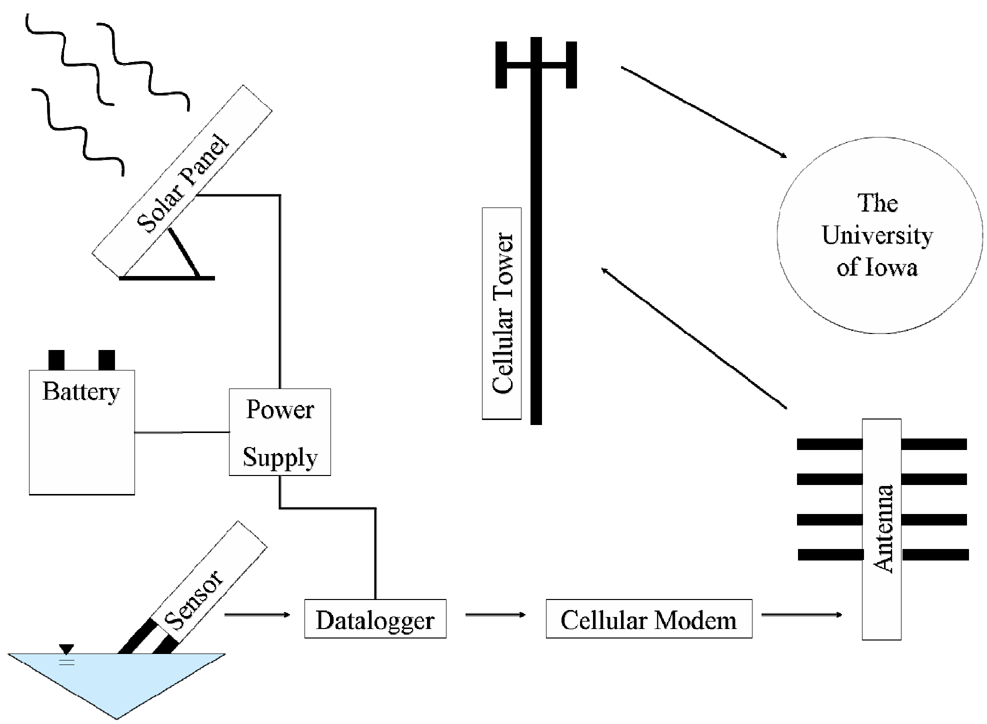

Fig. 2. Environmental monitoring network

The datalogger, cellular modem, power supply, and lead battery are all enclosed in a Stahlin non-metallic enclosure (Stahlin, Belding, Michigan, Model No. RJ1816HPL). The enclosures help to protect these components from rain and moisture. Locks are placed on the enclosure latches to discourage vandalism. Because there are holes in the bottom of the enclosure for the Hydrolab cable and the Antenna cable, the components inside are still subject to some moisture during humid conditions.

The wireless plan purchased for this system is an America's Choice II for Business 450 Plan which includes 450 minutes for a monthly price of $39.99. This plan was purchased under a business account. Because this account is for a university, Verizon was able to offer an 18% discount to bring the total monthly charge down to $32.79. The service charge is also tax exempt. The wireless plan is sufficient to meet current data transmission needs but may become too expensive when more data nodes are incorporated into the monitoring network. Therefore, an access node may be required for future work.

Hydrolab DS5X Setup
In order to take water quality measurements, we first had to learn how to operate the software associated with the Hydrolab DS5X. This software allows us to configure the data sonde, perform calibrations, and create measurement schedules. Calibration of the data sonde is required to ensure that measurements taken are both accurate and reliable. In order to calibrate the many sensors included on the data sonde, several different procedures were used.

Hydras3 LT Software. The software used to operate the Hydrolab DS5X is the Hydras3 LT software distributed by Hydrolab. This software enables the execution of basic tasks such as calibrating the sensors, configuring the data sonde clock and instrument ID number, and setting up measurement schedules which the sensor uses to collect water quality measurements. These measurement schedules are called log files and they allow the operator to deploy the sensor and obtain water quality measurements without the use of a data logger. The problem with this is that measurements may not be received in real-time. Only by using the Hydrolab DS5X data sonde in conjunction with a datalogger can real-time data be collected. A useful feature with the Hydras 3 LT software is the settings tab which can be used to set the order of SDI-12 parameters. The online monitoring function of the Hydras3 LT software allows you to obtain real-time water quality measurements by directly connecting the Hydrolab DS5X to the computer via the COM1 port. The most important function of this software is the calibration function.

Calibration Procedure. The calibration tab in the Hydras3 LT software is used to calibrate the sensors on the Hydrolab DS5X data sonde before deployment. Before performing the procedure, a calibration cup is filled with deionized water, attached to the sensor and shaken. This process is repeated several times in order to clean the sensors and remove any particles attached to the sensors. To calibrate the specific conductivity sensor, the following standards were purchased: 0.100 mS/cm Conductivity Standard Solution (Hydrolab cat. No. 013610HY) and 1.1412 mS/cm Conductivity Standard Solution (Hydrolab cat. No. 013620HY). These standards were used for a two-point calibration. Once we were comfortable with the calibrating procedure, standards were created using the procedure described in Standard Methods for the Examination of Water and Wastewater, 16th Edition. These include a 14.9 mS/cm and 1,412 mS/cm standard made by adding KCl to deionized water.

Calibration of the dissolved oxygen sensor is performed by using the water saturated air method as described in the Hydrolab DS5X user manual. In this calibration procedure the sensor is placed into the calibration cup with the open top of the cup facing up. The calibration cup is then filled with deionized water to a level just below the bottom of the dissolved oxygen sensor when the DS5X sensor is sitting on the calibration cup and not screwed into it. Any droplets on the dissolved oxygen sensor and temperature sensor were gently wiped off with a tissue. The sensor is then left unscrewed in the calibration cup for ten minutes to allow the water to saturate the air. The barometric pressure is entered in the LDO [SAT] tab of the calibration page in the Hydras 3 LT software and the calibration button is pressed completing the calibration of the LDO sensor.

Calibration of the pH sensor is performed using a two-point calibration with pH 7.00 buffer solution (Acros, New Jersey, USA, cat no. 61106-0040) and pH 10.00 buffer solution (Fisher Fair Lawn, New Jersey cat no. SB115-4). For this procedure, the 7.00 buffer solution is poured into the calibration cup to within a centimeter of the top of the cup with the DS5X screwed into the bottom of the cup. The pH value is then entered on the pH [Units] tab of the calibration page in the Hydras 3 LT software and the calibration button is pressed. The sensor is then rinsed with tap water and the procedure was repeated for the 10.00 buffer solution.

Calibration of the turbidity sensor is a two-point procedure using StablCal Standard Solutions of 100 NTU (Hydrolab cat no. 007308) and 1000 NTU (Hydrolab cat no. 007309). Before calibrating this sensor, the wiper is removed from the unit and stored in tap water. The sensor and calibration cup are then rinsed with tap water several times and dried with a lint-free cloth. The 100 NTU standard is then slowly inverted several times to re-suspend the particles in solution without introducing air bubbles into the mixture. It is then poured into the calibration cup to within a centimeter of the top of the cup with the DS5X in the cup. Thirty seconds were given to allow the NTU values to stabilize. A value of 100 NTUs is entered into the Turbidity [NTU] tab of the calibration page and the calibration button is then pressed. The cup and sensor were then rinsed with tap water several times and again dried with lint-free cloth. The calibration process was then repeated using the 1000 NTU calibration standard. The wiper is then reattached to the unit.

Remote Data Collection Platform Setup

Once we learned how to calibrate and operate the Hydrolab DS5X data sonde, the next task was to integrate it with the Campbell Scientific, Inc remote data collection platform. This required us to become proficient in configuring and programming the datalogger and cellular modem. Loggernet 3.1.5 is the software that is used to configure the CR1000 datalogger. This software can perform several tasks including creating programs used by the datalogger, communicating with the datalogger to download water quality measurements, and connecting to the datalogger to update it with new programs and set the clock.

The “Short Cut” function in the Loggernet software allows the operator to create a datalogger program. These programs tell the CR1000 when to collect measurements from the Hydrolab DS5X and what measurements to collect. The Short Cut for Windows feature first requires the user to input the frequency with which to take water quality measurements. Next, the type of sensor connected to the CR1000 must be selected. When using the Hydrolab DS5X data sonde via an SDI-12 connection, it is important to know what order your SDI-12 parameters are in. Next, the data table which will be stored by the datalogger is created. This allows you to select which measurements are collected on the datalogger (i.e. dissolved oxygen, turbidity, or pH). The reason it is important to know the order of the SDI-12 parameters is so the measurements in the data table columns can be correctly identified. The final step in creating a Short Cut program is compiling the program in a format that is usable by the datalogger. This is performed by pressing the “finish” button.

Once a program for the datalogger was created, the cellular modem was programmed to be deployment ready. First, a cellular voice plan was purchased from Verizon Wireless. When purchasing this plan, we obtained a system ID (SID). Next, the Cellset.exe program was downloaded from the Campbell Scientific, Inc website (http://www.campbellsci.com/downloads). After connecting the Redwing 100 Airlink CDMA Modem to the computer through the COM1 port and the NP12 Rechargeable Battery, the Cellset program was run. In this program, the 10-digit phone number and the SID were entered, and the baud rate was set to 9600. The “set settings” button was pressed and Cellset program configured the modem.

Following the modem configuration, Loggernet 3.1.5 was configured to be able to communicate with modem and datalogger. This was performed using the EZsetup screen. The EZSetup wizard is used to set up and configure the datalogger network. Specifically, this feature allows the user to select the type of datalogger used, the type of connection which will be used to communicate with the datalogger, and upload a datalogger program. In this setup wizard, the user will specify the phone number that the base station computer must dial in order to connect to the modem and datalogger. This wizard also enables the user to select the frequency with which water quality measurements are downloaded from the datalogger to the base station computer. The setup screen function in the Loggernet software performs the same tasks as using the EZsetup wizard with the only difference being that it is not as user friendly.

After the datalogger is set up, and deployed, the status and connection screens can be used to monitor the datalogger and interact with it. The status screen allows the user to view the communication history and the status of the communication link between the base computer and any datalogger in the field. It also displays when the last and next data download event is scheduled for. From this screen the automatic data collection schedule can be turned on or off. The connect screen allows the user to talk directly with a datalogger deployed in the field. This allows for the capability to perform a manual data download or update the time and datalogger program which is useful for daylight savings adjustments.

System Installation

With the functions of the data sonde and data collection platform understood, the next step was to assemble the system and deploy it. Included with the data collection platform were all of the cables necessary to wire the system. Once the environmental monitoring station was assembled and tested in the lab, the station was ready to be deployed into Clear Creek. This was done on March 22, 2006, and is shown in Fig. 3. Deployment began by pounding two 8-foot studded T-posts into the ground with a post driver. The Stahlin enclosure was then attached to the two T-posts using 3 U-bolts included with the enclosure. Two 8-foot studded T-posts were used to support the solar panel and were again driven into the ground using a post driver. Four U-bolts were used to attach the solar panel to the T-posts. The final 8-foot studded T-post was driven into the stream bed in the middle of Clear Creek. This post was used to attach approximately 4 feet of aircraft cable to the Hydrolab DS5X data sonde using two cable clamps. The aircraft cable was attached to the T-post by creating a loop with two cable clamps and placing a U-bolt into the loop and around the T-post. The Hydrolab calibration cable was then attached to the Hydrolab DS5X data sonde and the CR1000. The calibration cup on the Hydrolab DS5X data sonde was removed and replaced with the weighted sensor guard. The Hydrolab DS5X data sonde was then placed into the river. A large rock was placed underneath the sensor end of the data sonde in order to keep it off the soft creek bed. The power switch on the power supply was then flipped to the on position. Two desiccant bags were placed in the Stahlin enclosure and sealing putty was used to plug the cable holes on the bottom of the enclosure. A humidity indicator was also placed in the Stahlin enclosure. The enclosure door was closed and two padlocks were placed on the latches. To make sure the lab computer had communication with the environmental monitoring system, a COMM test was run to verify system connectivity. A successful COMM test signaled completion of the environmental monitoring station installation.

Fig. 3. Installation of Clear Creek monitoring station

Bi-Monthly Site Visits

The data sonde has suffered from biofouling following deployment. Thus, periodic visits to the environmental monitoring station must be made in order to swap out data sonde and perform maintenance. These visits are made to ensure that data collected by the environmental monitoring station is accurate. One method to do this is by taking independent samples and comparing those results with the Hydrolab DS5X data sonde measurements. Site visits are also used to clean components exposed to deleterious conditions. Biofouling of the sensor tips for example can be brushed away using soapy water. To maintain the Clear Creek environmental monitoring station, semi-weekly site visits are used to change the data sonde, clean system components, and collect water quality samples for QA/QC analyses.

A site visit begins with the calibration of a new Hydrolab DS5X data sonde to be deployed and the Hydrolab Quanta G sonde used for QA/QC measurements. Once calibrated, they are placed with all other supplies needed for a site visit. This includes 10 grab sample bottles, an insulated cooler for transporting water quality grab samples, chest-waders, sandals, and a field notebook. Once at the site 10 grab samples are collected from the middle of the creek (approximately 3 inches above the Hydrolab

DS5X data sonde) using a sampling rod. Ten Hydrolab Quanta G measurements are taken from the middle of the channel immediately adjacent to the Hydrolab DS5X data sonde. Care is taken to step in Clear Creek only in locations downstream of the sampling point so as to not influence Hydrolab Quanta G measurements.

Once the water quality samples are collected, maintenance on the environmental monitoring station is performed. First, any straw, grass, or biomass accumulating on the studded T-posts in the creek is removed. Next, the Hydrolab DS5X data sonde is removed from the stream. Any grass or biomass on the calibration cable and aircraft cable is removed. The calibration cable is disconnected from the data sonde, the weighted sensor guard is removed, and a calibration cup is placed on the old data sonde. Silica gel is then placed on the connection port on the top of the new data sonde. The calibration cable is then attached to the new data sonde and the sonde is placed back in the creek, making sure to place the sensor guard on the rock on the bottom of the creek. Next, any dirt or buildup is cleaned off the solar panel, and the humidity indicator in the Stahlin enclosure is checked.

The remainder of a site visit is completed in the Environmental Engineering Laboratory. The water quality grab samples are measured for dissolved oxygen using an HQ10 Portable LDO Meter (Hach, Loveland, Colorado, Cat No. 5181503), pH using a Beckman pH meter, and turbidity using a 2100 Laboratory Turbidimeter (Hach, Loveland, Colorado, Cat. No. 4700000). A light soapy water and brush are used to remove the biofilm from the sensor tips and data sonde housing. Soapy water is also used to clean the grab sample bottles, chest-waders, and sandals used during the site visit.

Results and Discussion

Since the installation of the environmental monitoring station, the Hydrolab DS5X data sonde in the stream has been replaced with a calibrated and non-biofouled data sonde four times. Table 1 shows the deployment period for each data sonde at the environmental monitoring station. Throughout the entire deployment period several water quality measurements were collected every twenty minutes. These measurements include pH, dissolved oxygen, temperature, and turbidity. In addition, the voltage in the NP12 Rechargeable Lead Battery was also measured.

Table 1 Deployment dates for Hydrolab DS5X data sondes

Deployment	Hydrolab DS5X Data Sonde Deployed	Deployment Date	Removal Date
1	1	3/22/2006	3/30/2006
2	2	3/30/2006	4/20/2006
3	1	4/20/2006	4/27/2006
4	2	4/27/2006	5/17/2006

Temperature

The first water quality parameter measured was temperature (Fig. 4). These measurements were from the first deployment in Clear Creek and a quality control sample point measured by the Hydrolab Quanta G. The diurnal rise of temperature during day and the fall of temperature at night were readily apparent. The measurements were continuous with no anomalous data points. Comparing the mean Hydrolab Quanta G measurement (taken from ten samples), 12.33º C, with the Hydrolab DS5X data point collected approximately at the same time, 12.09º C one can see that the difference is small. These differences appeared to be minimal in the second deployment as well (Fig. 5). There was one anomalous measurement during the second deployment on April 20th of 0º C. However, this was the only measurement which appeared to be inaccurate out of the almost 2100 measurements taken. Throughout the four deployments, biofouling of the temperature sensor was not a problem despite the fact that some biomass was observed on the sensor.

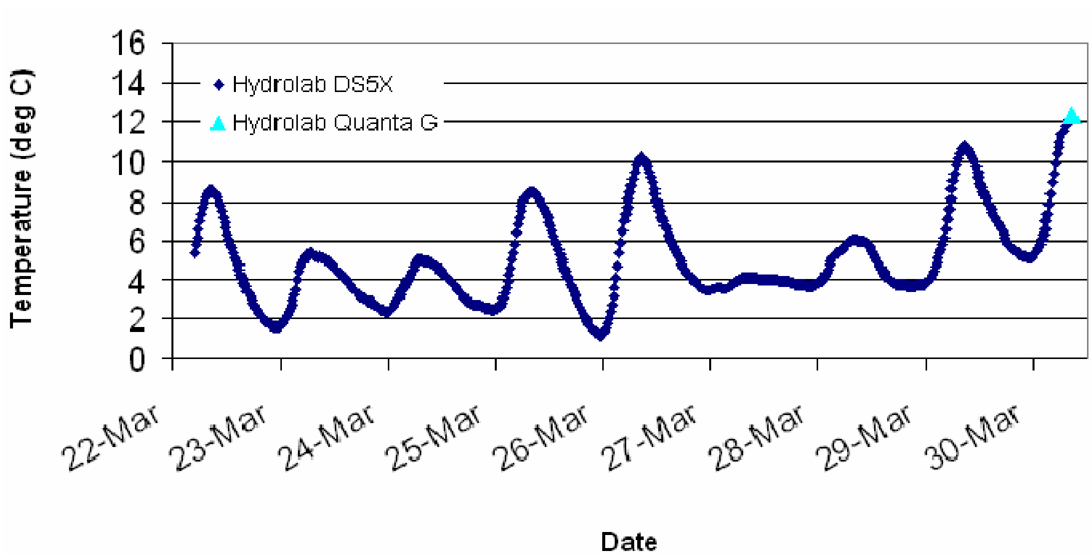

Fig. 4. Clear Creek water temperature during the first deployment

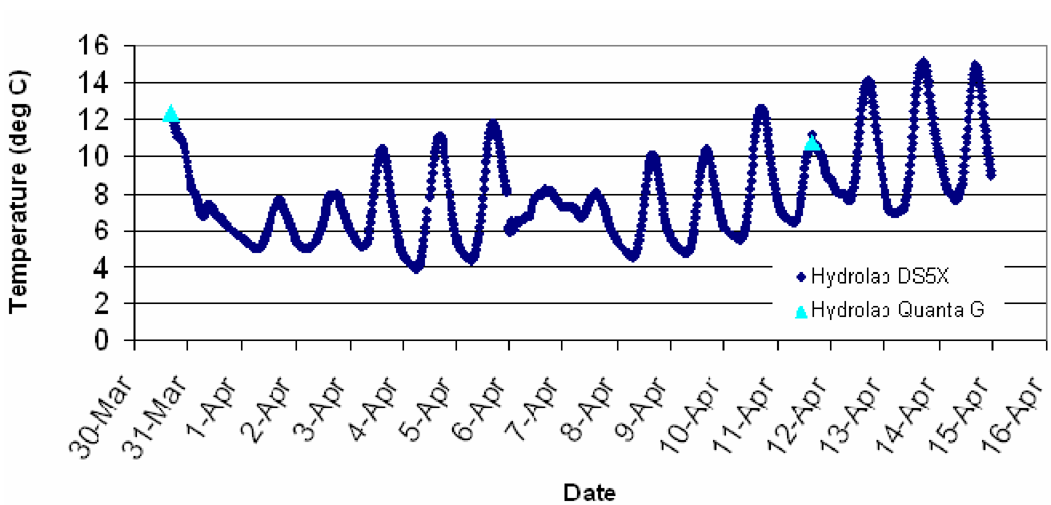

Fig. 5. Clear Creek water temperature during the second deployment

pH

The next parameter which was captured by the Hydrolab DS5X was pH. As with temperature, the Hydrolab DS5X captured a diurnal trend with pH measurements. Values of pH increased during the daytime, and decreased during the nighttime (Fig. 6 and Fig. 7). There did appear to be some discrepancies between the pH measured by the Hydrolab DS5X data sondes, the Beckman pH probe on the sample bottles, and the Hydrolab Quanta G as shown in Table 2. This may suggest several things. One, that the calibration procedure used for each of the three sensors required further improvement. Two, the Hydrolab DS5X data sonde did not warm up enough before taking a measurement. Finally, that biofouling of the Hydrolab DS5X pH probe occurred during deployment. This is supported by observations of biomass accumulation on the pH probe when a Hydrolab DS5X was removed from deployment, and that all of the Hydrolab DS5X readings were higher than the associated sample bottle and Hydrolab Quanta G measurements in Table 2. Another observation was that the range of readings for a daily period generally increases the longer the Hydrolab DS5X has been deployed. This may be another indication that biofouling occurred. A final point to consider regarding the difference in pH measurements is that the measurements were not taken at the same time. This would allow a slug of more acidic or basic water to have been captured by one measurement, but not another.

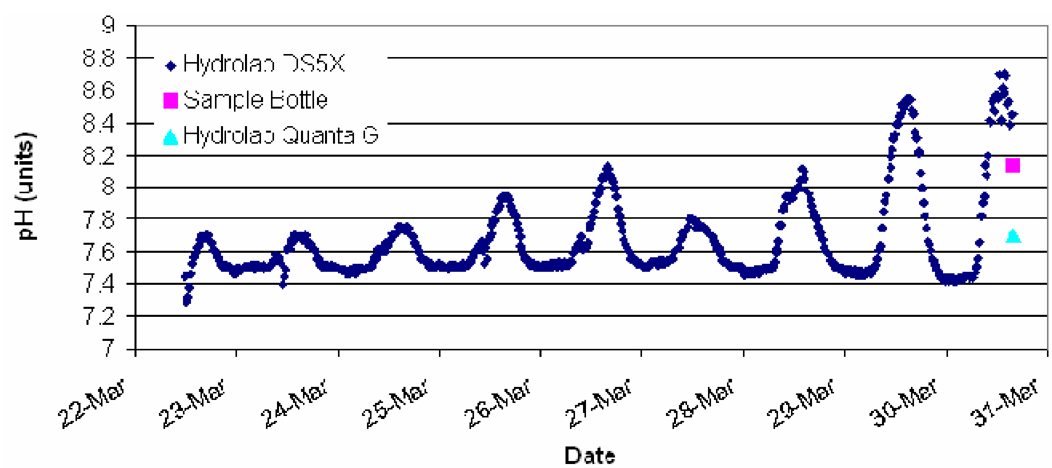

Fig. 6. Clear Creek pH during the first deployment

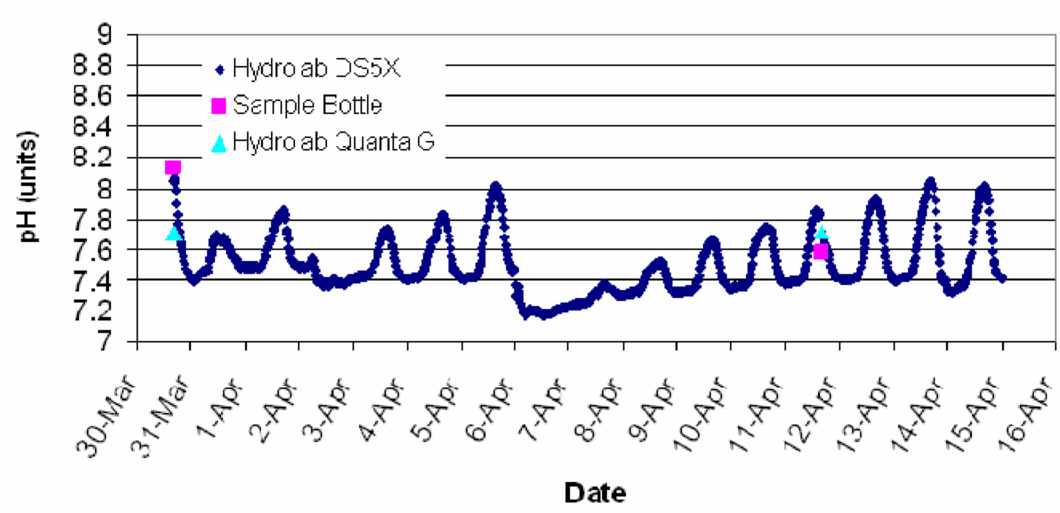

Fig. 7. Clear Creek pH during the second deployment

Table 2 Clear Creek pH measurements during site visits

Sampling Event	Time	Hydrolab DS5X Deployment	Hydrolab DS5X	Sample Bottles	Hydrolab Quanta G
1	3/30/06 4:00 PM	1	8.45	8.13	7.70
	3/30/06 4:20 PM	2	8.04		
2	4/11/06 3:40 PM	2	7.72	7.60	7.72
3	4/20/06 4:20 PM	2	8.30	7.90	8.05
	4/20/06 5:40 PM	3	7.73		
4	4/27/06 12:40 PM	3	8.08	7.88	NA
	4/27/06 1:00 PM	4	7.15		
5	5/4/06 1:30 PM			7.52	7.81
	5/4/06 1:40 PM	4	7.29		

Dissolved Oxygen

The next parameter the Hydrolab DS5X measured was dissolved oxygen. Measurements of Clear Creek dissolved oxygen taken for deployments 1 and 2 are shown in Fig. 8 and Fig. 9 respectively. The theoretical dissolved oxygen saturation concentration during these deployment periods is also shown. The saturated dissolved oxygen concentration was based on water temperature data and the chloride ion concentration was 0 mg/L. One prominent feature seen in the first deployment (Fig. 8) which was not seen in the second deployment (Fig. 9) was the disjointed data measured by the Hydrolab DS5X.. The difference between these two deployments was that in deployment 1 the Hydrolab DS5X data sonde #1 was used and in deployment #2, Hydrolab DS5X data sonde #2 was used. Data sonde #1 was used in the third deployment where the disjointed data was again seen (Fig. 10). The cause of this disjointed data is yet to be determined. Another difference seen in the first and second deployments was the difference in peak values of dissolved oxygen. In deployment 1 the peak values were roughly 6.5 mg/L and in deployment 2, the peak values were roughly 15.5 mg/L. This may be due to the particular Hydrolab data sonde deployed in each study. It should be noted that measurements taken during the second deployment were consistently above the saturation concentration during their peak periods. This may indicate that primary production was occurring during the daytime.

The diurnal trend was seen in the dissolved oxygen measurements as it was in the temperature and pH measurements. When comparing the peaks in dissolved oxygen measurements with those on the saturation curve, one can see that they are offset by about half a day. The peak dissolved oxygen values measured by the Hydrolab DS5X measurements occurred during the early afternoon. This indicated that temperature was not the main driving force in the daily dissolved oxygen cycles. One possible explanation is that primary production was the main driving force in the diurnal dissolved oxygen cycles.

Measurements taken from the Hydrolab DS5X data sondes were also compared with those taken by the Hydrolab Quanta G. Results indicated that the Hydrolab Quanta G dissolved oxygen values were significantly higher than the associated measurements taken during the first and third deployments. However, the measurements taken during the second deployment matched up well with those taken by Hydrolab DS5X data sonde #2.

Dissolved oxygen measurements have been collected from the Hydrolab DS5X data sondes, however, they are not dependable for Hydrolab DS5X data sonde #1. In the field it was observed that the sweeper on the Hydrolab DS5X did not entirely clean the top of the dissolved oxygen sensor. Thus, on about one quarter of the sensor cap there was significant amounts of biofouling. This may have contributed to erroneous dissolved oxygen readings but may not have been the only reason.

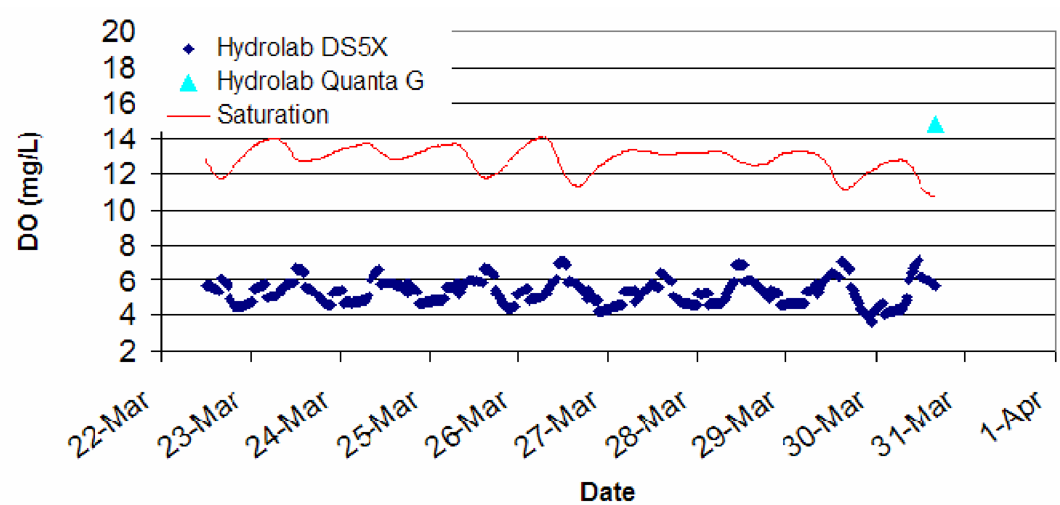

Fig. 8. Clear Creek dissolved oxygen measurements during the first deployment

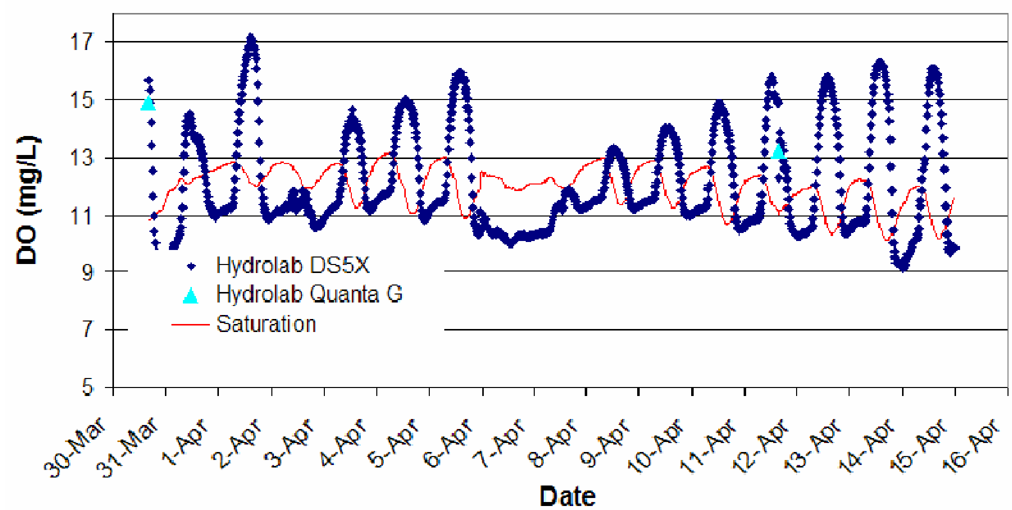

Fig. 9. Clear Creek dissolved oxygen measurements during the second deployment

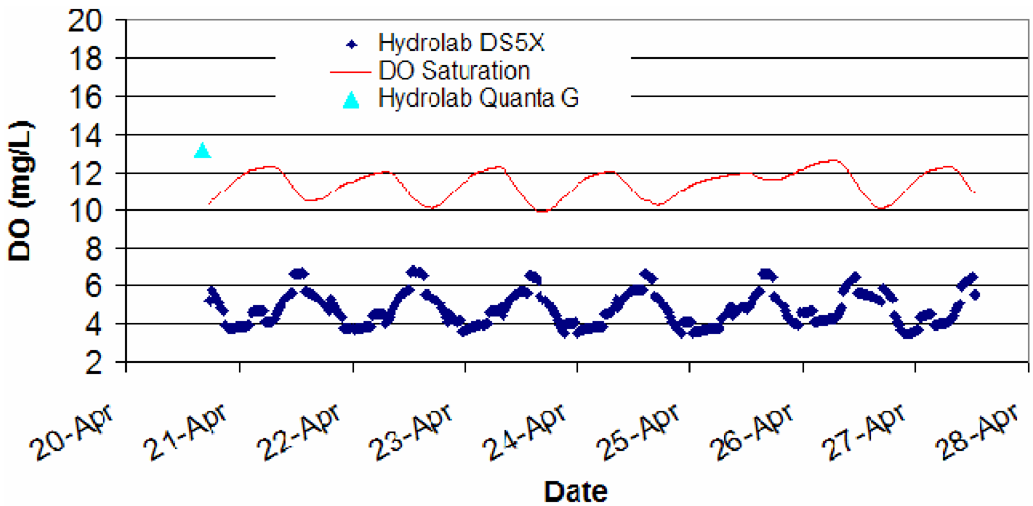

Fig. 10. Clear Creek dissolved oxygen measurements during the third deployment

Turbidity

The next measurement taken by the Hydrolab DS5X was turbidity. Measurements collected during the first and second deployments indicate that this sensor was working reasonably well except during the period of April 4th to April 11th (Fig. 11 and Fig. 12). All four sample bottle measurements appear to match the trend measured by the Hydrolab DS5X. The main noticeable feature in Fig. 12 was that a few of the readings reach 3000 NTUs during a span from April 6th to April 9th. This peak cannot be explained by rainfall as the significant rainfall events during the monitoring period occurred on April 2nd, 15th and 16th. Biofouling was most likely responsible for these high readings, but the sweeper seemed to perform relatively well in removing biomass from the turbidity sensor through all four deployments.

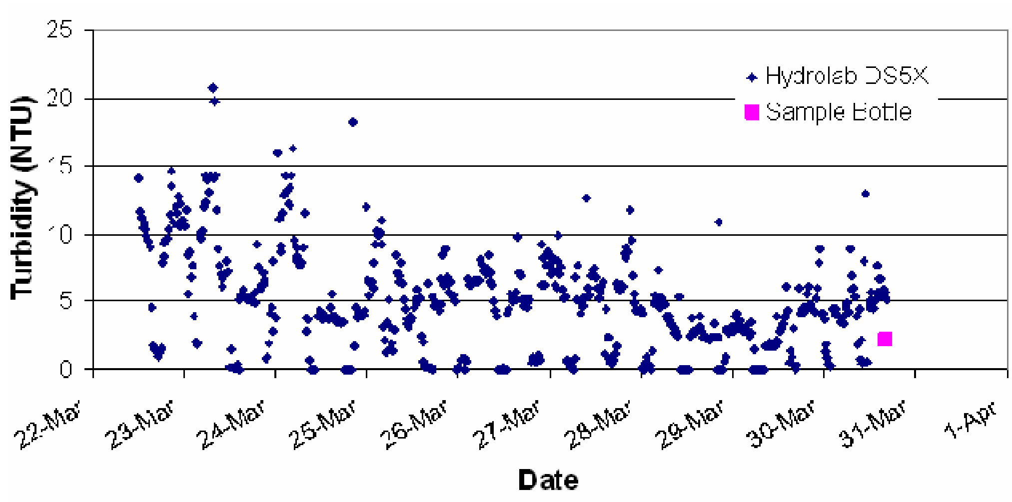

Fig. 11. Clear Creek turbidity measurements during the first deployment

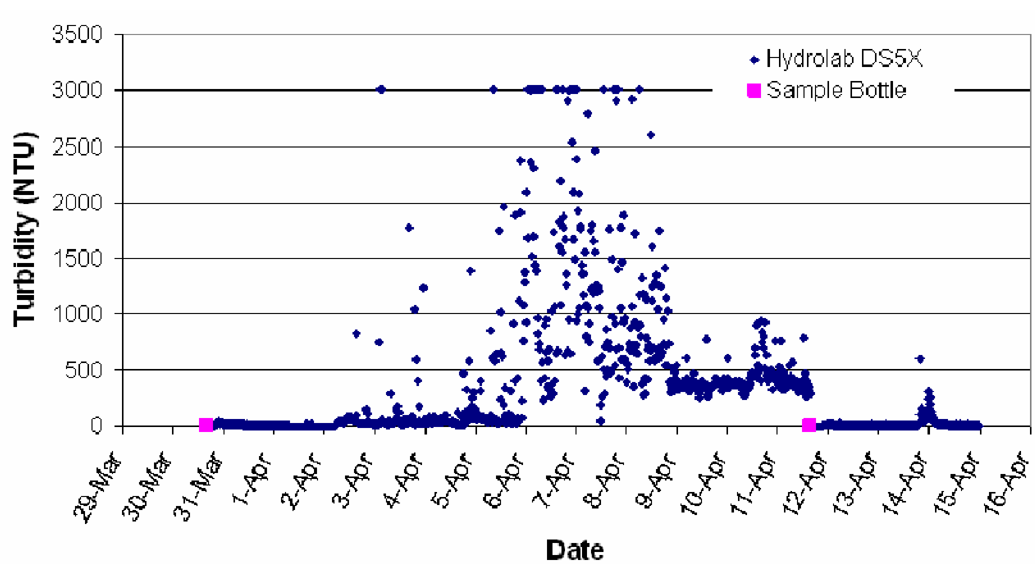

Fig. 12. Clear Creek turbidity measurements during the second deployment

Conclusions

In the past year we have made considerable progress in installing and operating a near real-time monitoring station at Clear Creek, Iowa. Initial efforts went towards the integration of hardware and software components used at the station. Specifically, Hydras3 LT software was used to enable the Hydrolab DS5X data sonde to fully communicate with the remote data collection platform purchased from Campbell Scientific. Loggernet 3.1.5 software was used to program the CR1000 datalogger to record and store measurements at a desired frequency. The environmental monitoring station was deployed on March 22, 2006. Since then we have been able to receive real-time water quality measurements from Clear Creek. Standard protocols were developed to ensure that data collected by the station was both accurate and reliable by means of discrete measurements from the Hydrolab Quanta G data sonde and grab samples in bottles. We are continuing to trouble-shoot the system with regards to dissolved oxygen and turbidity sensing. In the second year of the project, we will add nitrate sensors, pressure transducers, and a new remote data collection platform. In addition, we will introduce water quality modeling into the environmental cyberinfrastructure.

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Appendix

Table 3 Cost breakdown for the Hydrolab DS5X data sonde

Component		Unit Cost
Hydrolab DS5X Data Sonde	$	4,250.00
Hach LDO	$	1,300.00
Graphite Specific Conductance Sensor	$	350.00
DS Self Cleaning Turbidity	$	1,450.00
pH Sensor	$	475.00
Chlorophyll (Integrated)	$	2,450.00
Standard pH Reference	$	-
Internal Battery Pack	$	600.00
Chlorophyll Secondary Standard	$	320.00
Calibration Cable	$	250.00
Sub Total	$	11,445.00
Shipping	$	23.00
Total	**$**	**11,468.00**

Table 4 Cost breakdown for Campbell Scientific, Inc Remote Data Collection Platform

Component	Description	Unit Cost
CR1000	Datalogger	$ 1,350.00
Loggernet	Datalogger Software	$ 545.00
RTDM	Data Graphing Software	$ 395.00
ENC 12/14	Enclosure	$ 195.00
17716 Mounting Bracket	Enclosure Mounting Hardware	$ 35.00
CH100	Power Supply	$ 165.00
17260 Airlink Modem	Cellular Modem	$ 520.00
14394 Mounting Kit	Cellular Modem Mounting Kit	$ 20.00
14454 Antenna	Directional Antennta	$ 205.00
SC105 Interface Cable	Cellular Modem/Antenna Connecting Cable	$ 125.00
16987 - SC 105 Mounting Kit	Antenna Mounting Kit	$ 20.00
	Total	**$ 3,575.00**

Complementary Investigations for Implementation of Remote, Non-Contact Measurements of Streamflow in Riverine Environment

Basic Information

Title:	Complementary Investigations for Implementation of Remote, Non-Contact Measurements of Streamflow in Riverine Environment
Project Number:	2001IA1021G
Start Date:	9/1/2002
End Date:	3/31/2005
Funding Source:	104G
Congressional District:	Iowa 1st
Research Category:	Engineering
Focus Category:	Methods, Surface Water, Water Quantity
Descriptors:	free-surface velocity index, streamgaging stations, in-situ measurements, non-contact measurements, discharge measurements
Principal Investigators:	Marian V.I. Muste, Allen Bradley, Ralph Cheng, Anton Kruger

Publication

1. Polatel, C. 2003. Signature of Bed Characteristics on Free Surface Velocity in Open Channel Flows, International Association of Hydraulic Research Congress XXX, Thessaloniki, Greece.
2. Polatel, C., M. Muste, V.C. Patel, T. Stoesser, W. Rodi. 2005. Double-averaged velocity profile over large-scale roughness, XXXI International Association of Hydraulic Engineering and Research Congress, Seoul, Korea.
3. Polatel, C. 2005. Indexing by free surface velocity: A prospect for remote discharge estimation, XXXI International Association of Hydraulic Engineering and Research Congress, Seoul, Korea.
4. Stoesser, T., W. Rodi, C. Polatel, V.C. Patel, M. Muste. 2005. Large eddy simulations of the flow over two-dimensional dunes, XXXI International Association of Hydraulic Engineering and Research Congress, Seoul, Korea.
5. Polatel, C., T. Stoesser, M. Muste. 2005. Velocity Profile Characteristics in Open Channel Flow over Two-Dimensional Dunes with Small Relative Submergence, World Water and Environmental Resources Congress 2005, Anchorage, AK.
6. Balachandar, R., C. Polatel, B. Hyun, K. Yu, C. Lin, W. Yue, V.C. Patel. 2002. LDV, PIV and LES

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7. Polatel, C., 2006. Large-Scale Roughness Effect on Free-Surface and Flow Characteristics in Open-Channel Flows, PhD Dissertation, The University of Iowa, Iowa City, IA.
8. Polatel, C., M. Muste, V.C. Patel. 2006. Velocity indexing over large-scale roughness by free surface velocity, 7th International Conference on HydroScience & Engineering, Philadelphia, PA.
9. Polatel, C., M. Muste, V.C. Patel, T. Stosser, W. Rodi. 2006. Free surface respnse to large-scale bed roughness, 7th International Conference on HydroScience & Engineering, Philadelphia, PA.

Complementary Investigations for Implementation of Remote, Non-contact Measurements of Streamflow in Riverine Environment

Marian Muste, Allen Bradley, Anton Kruger, Ralph Cheng

Problem and Research Objectives

Estimation of flow discharges in natural streams using conventional approaches requires measurements of the flow depth, velocity distribution over the depth, and stream cross-section bathymetry. The emerging remote measurement technologies have potential to measure remotely, non-contact the free-surface velocity (radar, image-based methods, lidars) and bathymetry (ground-penetrating radars). Consequently, the need for describing the velocity distribution in the water column (or its surrogate, the velocity index, which is the ratio between the depth-averaged velocity and the free-surface velocity) for various flow regimes and bed roughness characteristics has emerged.

The main objective of this research was to establish velocity indices for flows in open channels with smooth and rough beds for a range of bulk flow velocities and depths. A secondary objective was to relate the free-surface appearance (texture) with the bulk flow velocity, bed roughness characteristics and its relative submergence. The research project was conducted to support the USGS Hydro-21 initiative for implementation of remote, non-contact, real-time estimation of stream discharges in natural streams.

Methodology

The research objectives were accomplished by conducting a series of experimental and numerical simulation tests for various open-channel flow conditions over bed roughness resembling that in natural streams. The geometry of the tested large-scale roughness elements is provided in Figure 1. The experiments were performed in a hydraulic flume with varying flow depth and bed roughness. Rectangular ribs and two-dimensional, dune-shaped obstacles were placed on the flume bottom to simulate different bed roughness conditions. Laser Doppler Velocimetry (LDV) measurements were made to obtain the vertical velocity profiles at various roughness element locations. Large-Scale Particle-Image Velocimetry (LSPIV) measurements were made to determine the velocity distribution at the free-surface. Additional experiments using image based techniques were used to capture the “signature” of the bed roughness elements on the free-surface appearance. Large-Eddy Simulation (LES) was used to numerically simulate flow over the same roughness elements to complement the experimental results for a wider range of flow conditions. Details of the numerical and experimental procedures are provided in Polatel (2006).

The local vertical velocity distributions obtained from experiments and numerical simulations were used to determine a “representative” velocity profile for each roughness element and flow case. The representative velocity profile characterizes the flow over a rough bed from a “hydraulic” point of view. Specifically, the representative velocity distribution will be the basis for calculation of the discharge in river reach where these types of roughness exist. The representative profile was obtained by spatially-averaging mean (temporal) velocity profiles over the roughness wavelength. The double-averaged velocity profile was then fitted using a two-layer, power-law model for providing an analytical form for the representative profile, and to further compare the flow over different roughness conditions as well as to establish an indexing velocity linking the free-surface velocity to the bulk flow velocity.

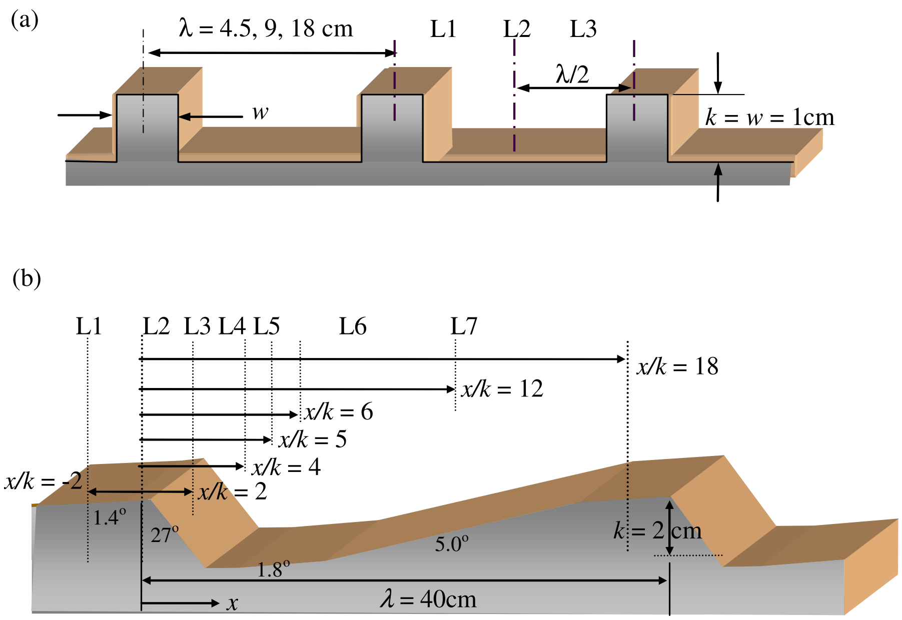


Fig. 1. Characteristics of the large-scale roughness elements investigated in the study: a) ribs; b) dunes

The flow conditions investigated experimentally in the study are summarized in Tables 1,2,3,4,5,6,7. The legend for the notations in the tables is the following: *h* is the flow depth, U_0 is the free-surface velocity, U_{Bulk} is the bulk flow velocity, U_{LDV} is the velocity measured with the LDV, *Fr* is the Froude number, *Re* is the Reynolds number (*Fr* and *Re* are based on flow depth, *d*, and bulk flow velocity), *AR* is the flow aspect ratio (the ratio of channel width to flow depth), u_* is the bed shear velocity, and S_0 is the channel slope.

Table 1 Flow conditions for shallow open-channel flows over flat bed

Code	***h* (m)**	**U_0 (m/s)**	**U_{Bulk} (m/s)**	***Fr***	***Re***	***AR***	**u_* (m/s)**	**S_0**
S01	0.025	0.044	0.029	0.06	718.4	24.38	0.0050	2.81E-4
S02	0.025	0.106	0.085	0.17	2119.8	24.38	0.0073	6.81E-4
S03	0.025	0.205	0.173	0.35	4322.4	24.38	0.0100	4.32E-2
S04	0.025	0.384	0.336	0.68	8389.5	24.38	0.0174	8.58E-2

Table 2 Flow conditions for constant velocities over flat bed experiments.

Code	***h* (m)**	**U_{LDV}**	**U_0 (m/s)**	**U_{Bulk} (m/s)**	***Fr***	***Re***	***AR***	**u_* (m/s)**	**S_0**
S06	0.06	0.506	0.490	0.445	0.50	26700	10.16	0.0213	3.83E-04
S08	0.08	0.504	0.489	0.449	0.51	35920	7.62	0.0229	1.70E-04
S10	0.10	0.496	0.485	0.455	0.46	45500	6.10	0.0247	1.70E-04

Table 2 Flow conditions for flow over ribs with λ = 0.045 m at (location L1)

Code	***h (m)***	***λ (m)***	***U_{LDV} (m/s)***	***U_0 (m/s)***	***U_{Bulk} (m/s)***	***Fr***	***Re***	***AR***	***u_* (m/s)***	***S_0***
R01	0.055	0.045	0.474	0.46	0.370	0.50	20350	11.08	0.053	0.00348
R02	0.075	0.045	0.505	0.49	0.396	0.46	29723	8.13	0.051	0.00327
R03	0.095	0.045	0.503	0.49	0.406	0.42	38526	6.42	0.049	-

Table 3 Flow conditions for flow over ribs with λ = 0.09 m (location L1).

Code	***h (m)***	***λ (m)***	***U_{LDV}***	***U_0 (m/s)***	***U_{Bulk} (m/s)***	***Fr***	***Re***	***AR***	***u_* (m/s)***	***S_0***
R04	0.055	0.09	0.350	0.34	0.289	0.39	15915	11.08	0.050	2.49E-03
R05	0.075	0.09	0.447	0.43	0.376	0.44	28200	8.13	0.055	2.55E-03
R06	0.095	0.09	0.514	0.49	0.440	0.46	41800	6.42	0.059	2.61E-03

Table 4 Flow conditions for flow over ribs λ = 0.18 m (location L1).

Code	***h (m)***	***λ (m)***	***U_{LDV} (m/s)***	***U_0 (m/s)***	***U_{Bulk} (m/s)***	***Fr***	***Re***	***AR***	***u_* (m/s)***
R07	0.055	0.18	0.359	0.35	0.307	0.42	16896	11.08	0.047
R08	0.075	0.18	0.434	0.42	0.364	0.42	27321	8.13	0.053
R09	0.095	0.18	0.512	0.49	0.434	0.45	41273	6.42	0.054

Table 5 Flow conditions for flows over dunes (location L1).

Code	***h (m)***	***U_{LDV} (m/s)***	***U_0 (m/s)***	***U_{Bulk} (m/s)***	***Fr***	***Re***	***AR***	***u_* (m/s)***
D01	0.06	0.418	0.41	0.369	0.48	22140	10.2	0.0339
D02	0.08	0.467	0.44	0.394	0.44	31520	7.6	0.0323
D03	0.10	0.477	0.475	0.425	0.43	42500	6.1	0.0299

Table 6 Flow conditions for flows over dunes roughened with sand particles and wiremesh (location L1).

Code	***h (m)***	***U_{LDV} (m/s)***	***U_0 (m/s)***	***U_{Bulk} (m/s)***	***Fr***	***Re***	***AR***	***u_* (m/s)***
Sand	0.10	0.505	0.48	0.436	0.44	43600	6.1	0.040
WM	0.10	0.525	0.515	0.466	0.47	46600	6.1	0.036

Analysis Approach

The analysis for the study is based on the following approach (see Figure 2): the detailed instantaneous flow from field obtained with numerical or experimental measurements was firstly time-averaged at characteristics locations over the roughness elements. Next, the obtained velocity profiles were averaged firstly over the roughness wavelength, then over the depth.

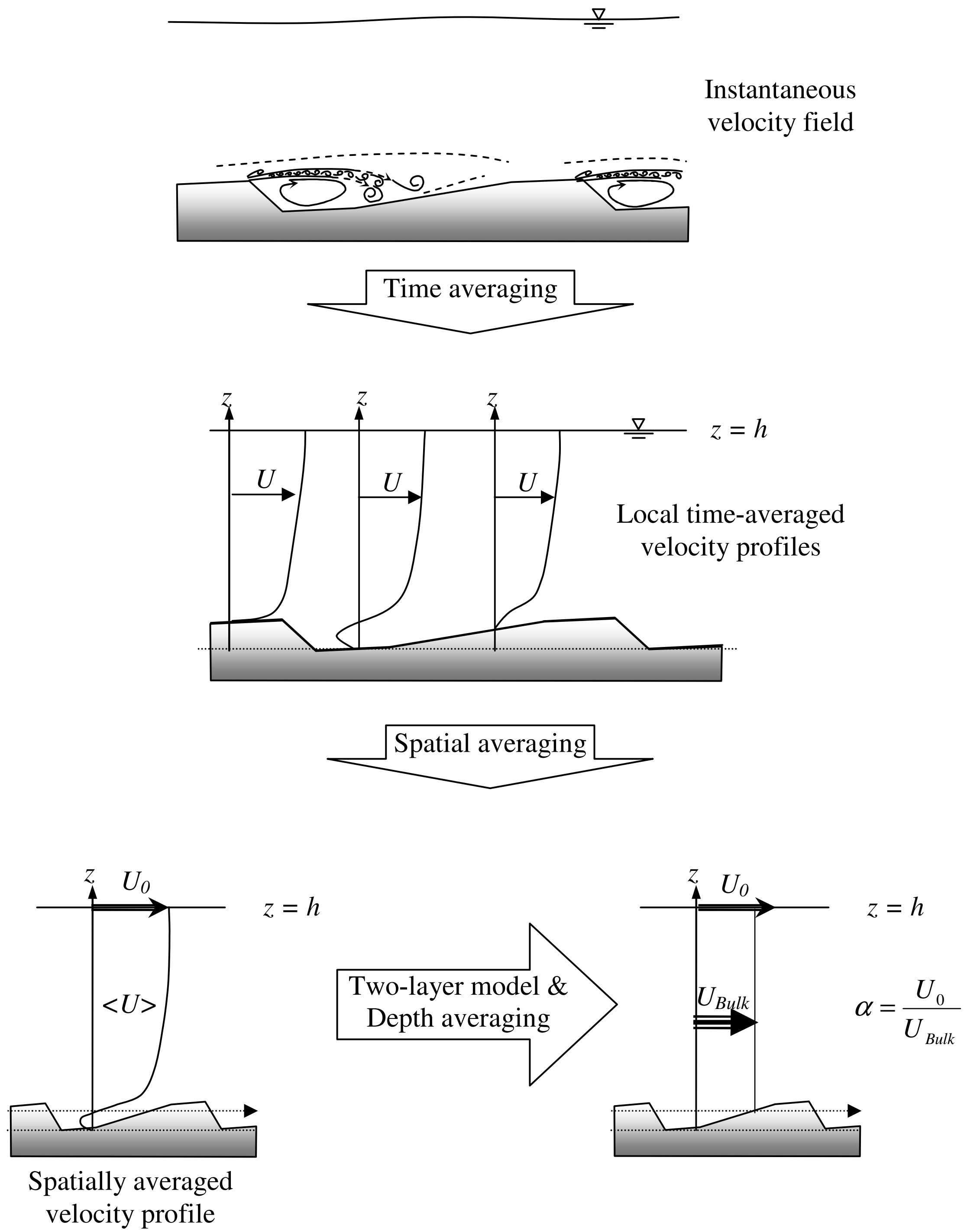


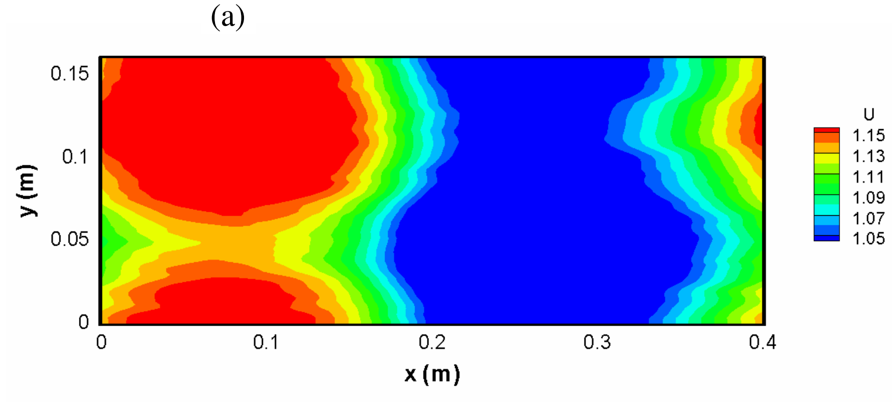
Fig. 2. Sequence of time/space averaging employed to obtain bulk flow characteristics.

Principal Findings and Significance

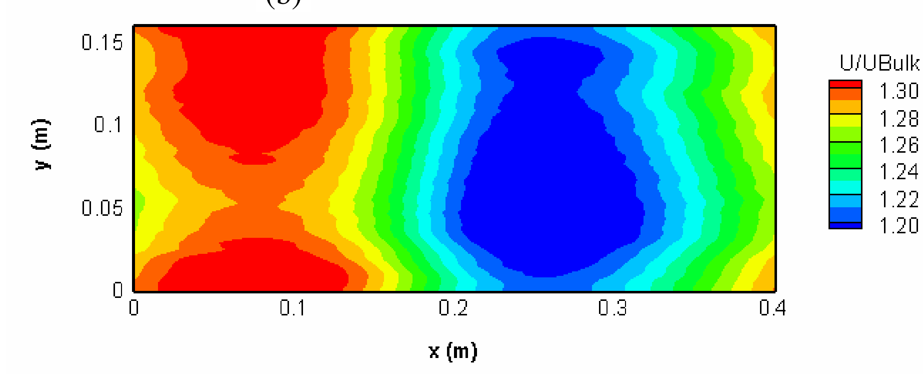
Free Surface Texture

The numerical and experimental results show that the free surface has embedded in its texture the bed roughness signature. This conclusion is substantiated by the plots in Figures 3 and 4. The signature is a function of the relative roughness submergence.

(a)

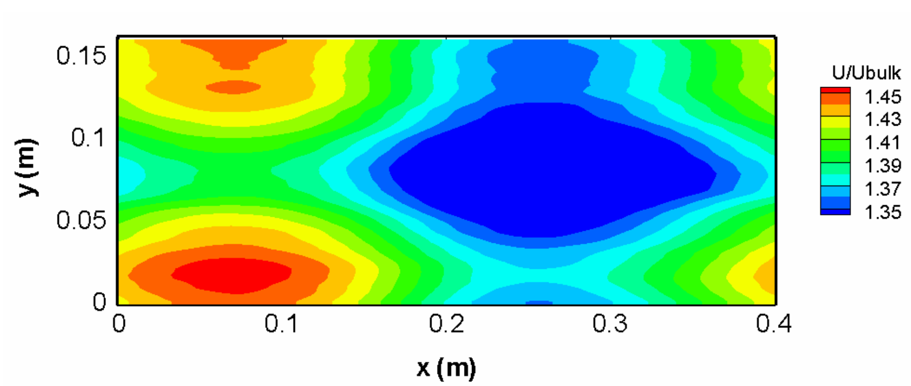


(b)



Av.	1.25
Std. Dev.	0.05
Max	1.49
Min	1.33

(c)



Av.	1.39
Std. Dev.	0.04
Max	1.46
Min	1.30

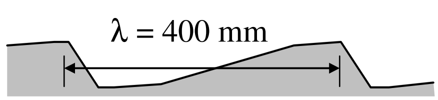


Fig. 3. LES results for normalized free-surface velocity distribution for flow over dunes with flow depth of (a) 6 cm, (b) 8 cm, (c) 10 cm.

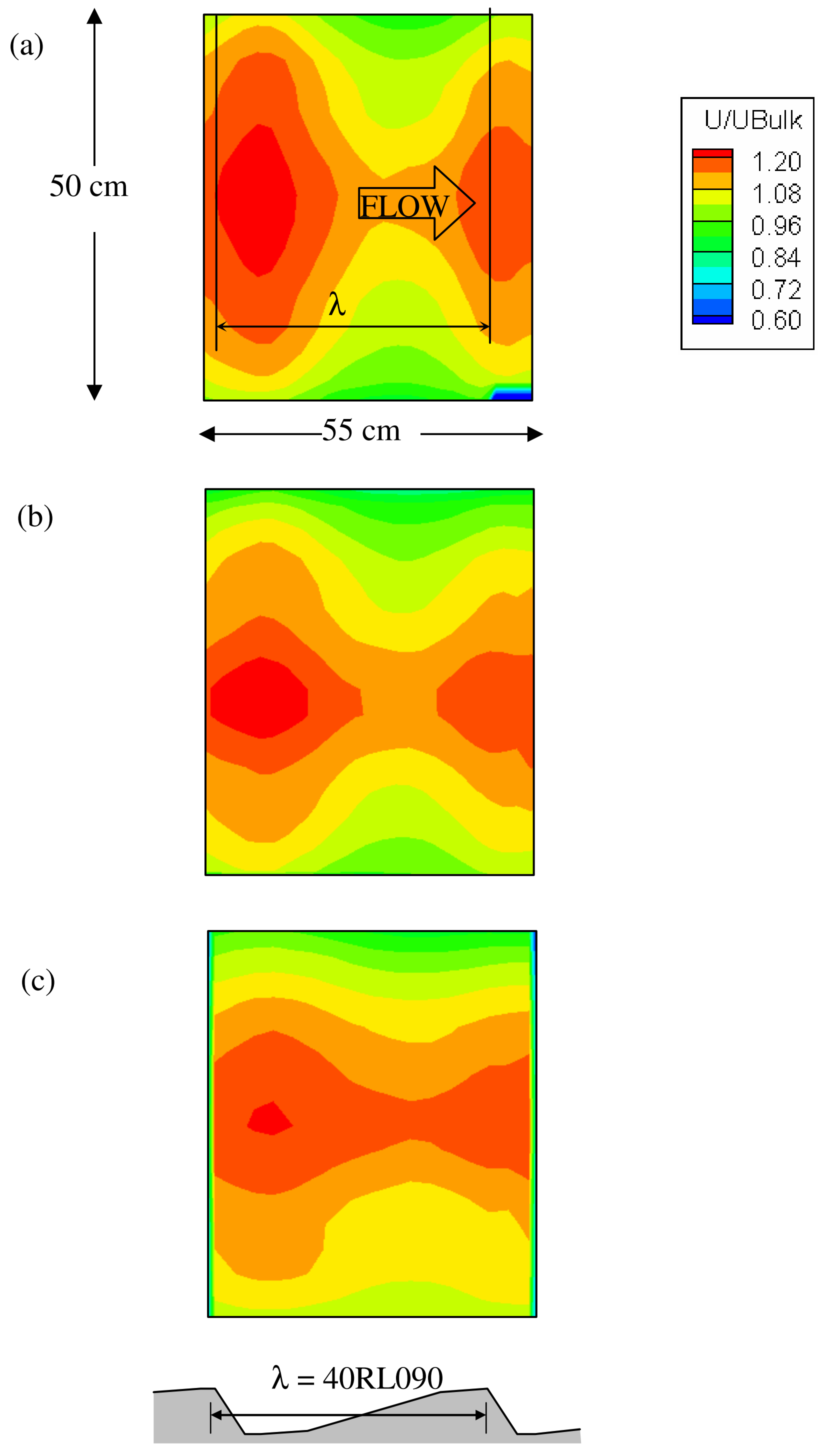


Fig. 4. LSPIV results for normalized free-surface velocity distribution for flow over dunes with flow depth of (a) 6 cm, (b) 8 cm, (c) 10 cm.

The large and small scale ripples are indicative of the flow regime and the presence of the roughness elements on the bed. The texture of the free surface formed by the ripples can be quantitatively estimated using power spectra of the free-surface waviness captured by the imaging devices (Polatel, 2006). Power spectra of this kind for flow over smooth bed and ribs are illustrated in Figures 5 and 6, respectively.

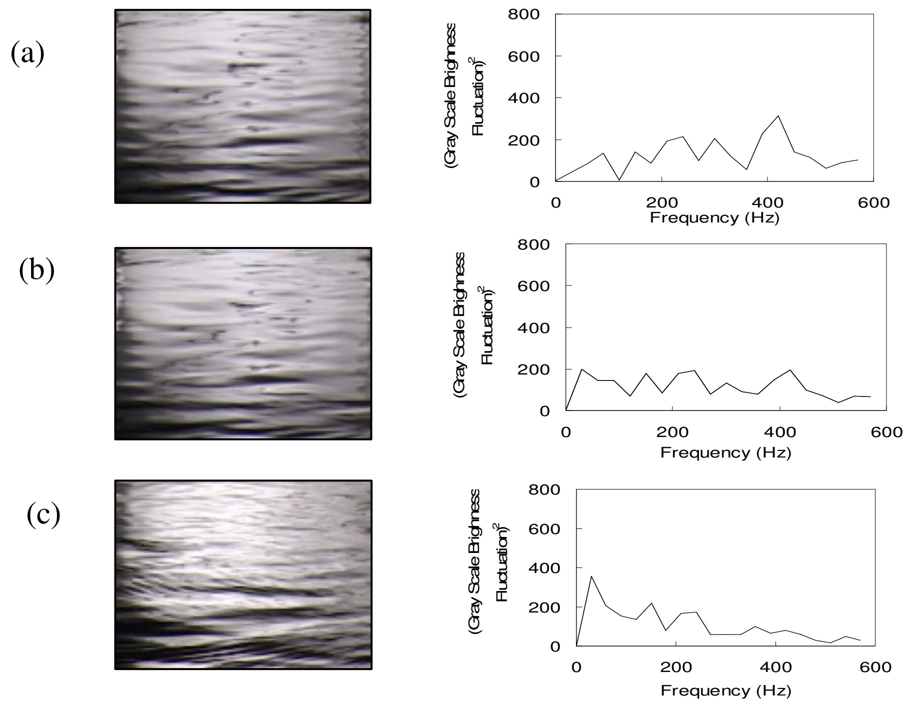


Fig. 5. Power spectra for the free-surface texture recordings for flow over smooth bed with flow depth of (a) 6 cm, (b) 8 cm, (c) 10 cm

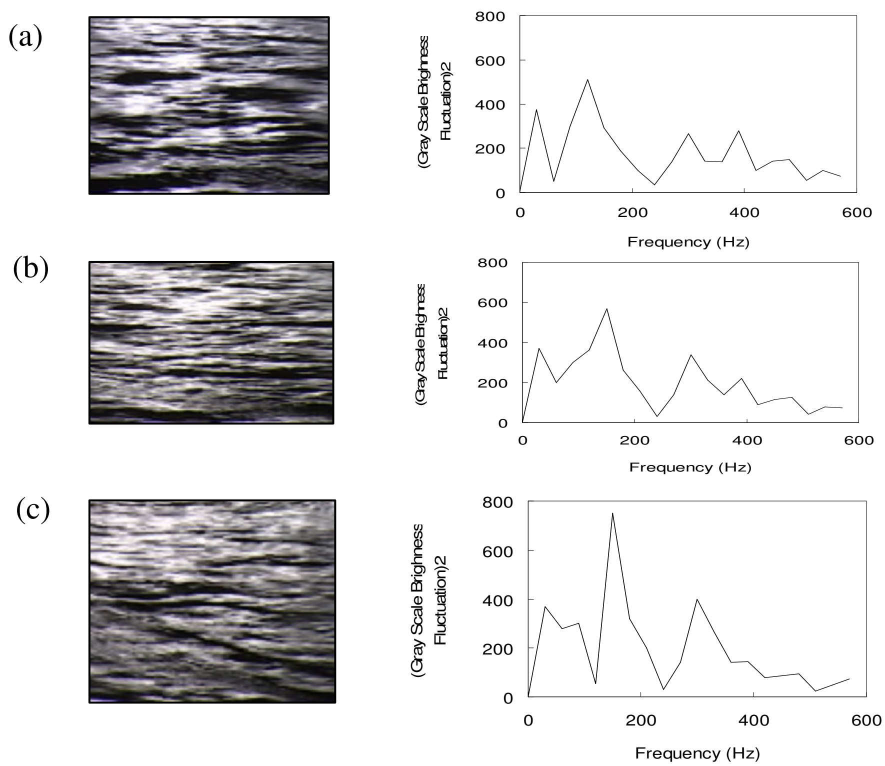
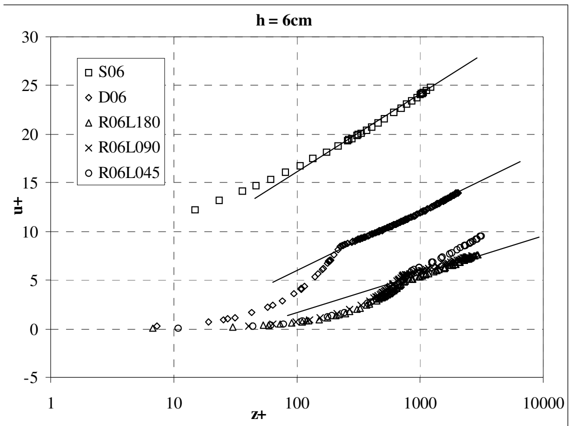


Fig. 6. Power spectra for the free-surface texture recordings for flow over rib roughness with $\lambda = 9$ cm with flow depth of (a) 6 cm, (b) 8 cm, (c) 10 cm

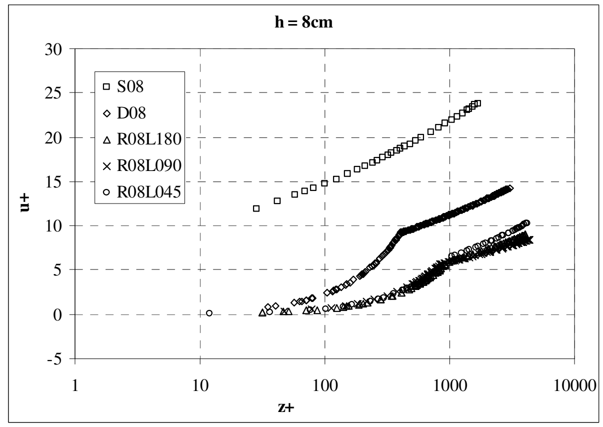
Power Law Fit for Two-layer Velocity Profile

The roughness function is traditionally used to describe the effect of roughness on mean velocity distribution. However, the u^+ vs. z^+ plot of our data shows that the effect of roughness is not limited to a shift from the smooth bed profile, but also the slope of the curve changes, as illustrated in Figure 7.

a)



b)



c)

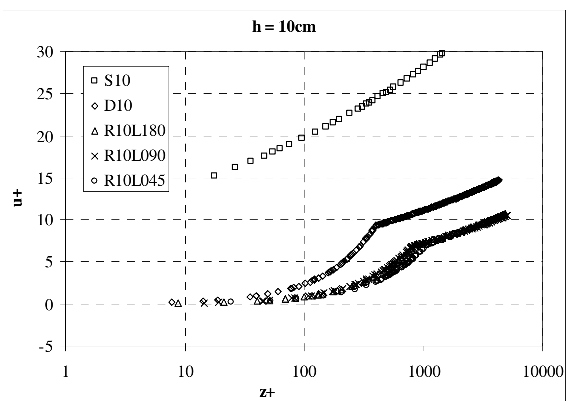


Fig. 7. Power law fit for velocity profiles of the flows over smooth, ribs, and dunes: a) h = 6 cm; b) h = 8 cm; c) h= 10 cm

Velocity Index

Velocity indices for the flow investigated in this study vary in the 0.802–0.938 range. The obtained velocity indices are presented in Table 7 and plotted in Figure 8. The experiments demonstrate that ratio of free surface velocity to depth-averaged velocity depends on the channel bed roughness. The results show, as expected, that the flow over smooth bed has higher velocity indices than those of flow over rough beds. The results also indicate that the velocity index is dependent on the relative submergence of the roughness elements, with higher velocity indices for larger relative submergence.

Overall, the study provides valuable information for implementation of remote discharge measurement methodologies and sheds light on important hydrodynamics characteristics of the open-channel flow over large-scale roughness.

Table 7 Summary of the velocity indices

	Code	α
Smooth Bed	S01	0.659
	S02	0.802
	S03	0.844
	S04	0.875
	S06	0.908
	S08	0.918
	S10	0.938
RL045	R01	0.804
	R02	0.809
	R03	0.828
RL090	R04	0.850
	R05	0.874
	R06	0.898
RL180	R07	0.878
	R08	0.878
	R09	0.887
2D dunes	D01	0.900
	D02	0.895
	D03	0.895
	Sand	0.908
	WM	0.905

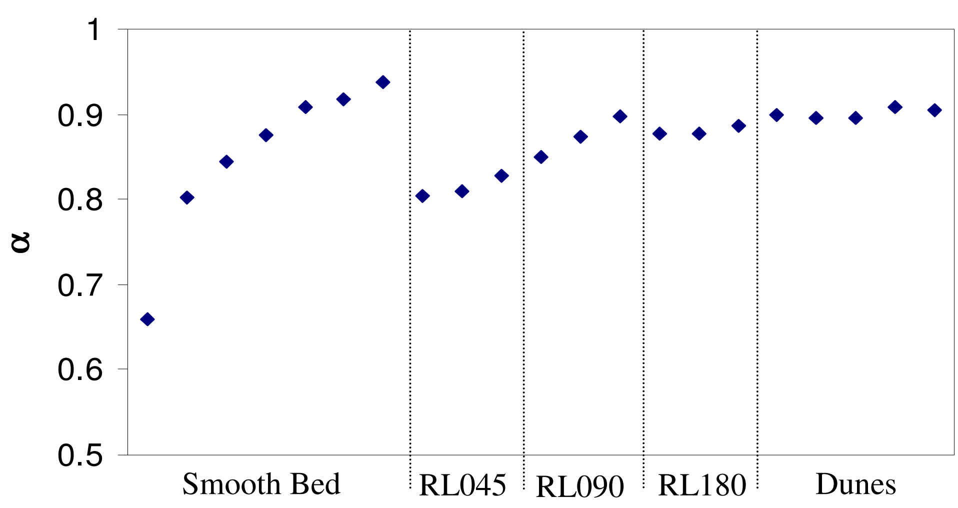


Fig. 8. Graphical representation of the velocity indices obtained from the experimental study

Relationship of Nitroso Compound Formation Potential to Drinking Source Water Quality and Organic Nitrogen Precursor Source Characteristics

Basic Information

Title:	Relationship of Nitroso Compound Formation Potential to Drinking Source Water Quality and Organic Nitrogen Precursor Source Characteristics
Project Number:	2002IA16G
Start Date:	10/10/2002
End Date:	5/1/2006
Funding Source:	104G
Congressional District:	IA 1st
Research Category:	Water Quality
Focus Category:	Water Quality, Non Point Pollution, Toxic Substances
Descriptors:	drinking water, water quality, contaminants, non-point pollution, disinfection
Principal Investigators:	Richard Louis Valentine

Publication

1. Assuoline, Jason, 2004, An Exploratory Study of the Formation of N-Nitrosodimethylamine (NDMA) in Chloraminated Natural Waters, MS Thesis, Dept. of Civil & Environmental Engineering, University of Iowa, Iowa City, Iowa.
2. Assuoline, Jason, Richard L. Valentine, Douglas J. Schnoebelen, Ashley Mordasky. An Exploratory Study of N-Nitrosodimethylamine (NDMA) Formation in Chloraminated Natural Waters, submitted to Environmental Science and Technology.
3. Chen, Zhuo and Richard L. Valentine, 2006. Mechanisms and Kinetics of NDMA Formation from Natural Organic Matter, submitted to Environmental Science and Technology.

Relationship of Nitroso Compound Formation Potential (NCFP) to Drinking Source Water Quality and Organic Nitrogen Precursor Source Characteristics

Richard L. Valentine

Problem and Research Objectives

Recent research indicates that certain disinfection practices may result in the formation of significant amounts of N-nitrosodimethylamine (NDMA), and quite likely other nitroso compounds in drinking water. These compounds are believed to be formed when chlorine is added to water containing ammonia and certain organic nitrogen compounds ("precursors"). Measurements in several drinking water distribution systems suggest that unprotected sources receiving point and non-point waste discharges are particularly susceptible to their formation, especially when chloramination is practiced.

The formation of NDMA and possibly other nitroso compounds in drinking water is an emerging concern because they are generally carcinogenic, mutagenic, and teratogenic (Loeppky et al., 1994; O'Neill et al., 1984). For example, the nitrosamine, N-nitrosodimethylamine, NDMA $(CH_3)_2NNO)$ is a particularly potent carcinogen. Risk assessments from California's Office of Environmental Health Hazard Assessment (OEHHA) and US EPA identify lifetime de minimis (i.e., 10^{-6}) risk levels of cancer from NDMA exposures as 0.002 ppb (2 ng/L) and 0.0007 ppb, respectively. In February of 2002 the California Department of Health Services established an interim action level of 0.01 ppb (10 ng/L) in drinking water.

Many drinking water sources in the Midwest and other parts of the country are unprotected receiving point and non-point waste discharges. Municipal and industrial waste discharges, and those associated with agricultural practices, are potentially important sources of the organic nitrogen precursors required for the formation of nitroso compounds. These waters are correspondingly expected to be susceptible to nitroso compound formation from chloramination. This may limit the use of some water sources for drinking water or restrict treatment options that otherwise have desirable characteristics. Initial observations indicate that some consumers are being exposed to undesirable levels of NDMA. Organic nitrogen is therefore not a simple benign pollutant typically associated with nutrients as generally thought.

A need exists for an improved understanding of the nature and extent of this potential problem. Work is especially needed that relates nitroso compound formation potential to source water quality and origin of organic nitrogen precursors, watershed uses, and to biogeochemical processes that could influence the quantity and types of nitroso compounds potentially produced.

Based upon the ascertained research needs, the following specific objectives of this research study have been formulated with respect to the relationship of source water quality and the formation of NDMA:

1. Characterize the NCFP in a variety of "susceptible" surface and groundwater drinking source waters;
2. Examine the relationship of NCFP to source water quality and land usage; and
3. Conduct mechanistic studies to characterize precursors and the influence of potentially important physical, chemical, and biological processes on the formation of NDMA.

Methodology

Overview

Previously we reported on work that focused on the measurement of the NDMA “Formation Potential” (NDMAFP) in natural water samples obtained from a variety of agriculturally impacted sources in Iowa. The NDMAFP was determined through a series of laboratory assays which focused on the amendment of the surface water samples with the disinfectant monochloramine (NH_2Cl). The NDMAFP in river water samples was examined by season and in relation to other water quality variables such as nitrite, nitrate, and organic nitrogen concentrations. Additional studies were conducted to delineate the potential importance of DMA as an identifiable NDMA precursor versus other unidentified nitrogenous substances. An additional activity was an investigation of the influence of riverbank filtration (RBF) on the formation of NDMA after chloramination. RBF is a practice in which drinking water is withdrawn from shallow wells near a river.

The primary focus of the work performed since the last report has been on mechanistic studies of NDMA formation in drinking source water obtained from rivers previously examined for their NDMA FP (i.e., ag impacted) as well as one pristine source. As such, the findings presented are excerpted wholly from the Ph.D. dissertation of Dr. Zhuo Chen (August 2006, University of Iowa). The work reported here can be divided into three focus areas:

1. Characterization of NDMA formation from selected humic fractions of isolated riverine NOM. NOM was fractionated into 6 humic fractions and the NDMAFP of each was determined. A comparison was made of the contribution of each fraction to the observed NDMAFP of the whole water sample.
2. Characterization of the reaction mechanism and kinetics of NDMA formation from reaction of monochloramine with natural organic matter. The approach was to postulate a reaction mechanism, conduct experiments in which water quality such as pH and chloramine dosages were varied, and then compare measured with predicted NDMA concentrations.
3. Examination of the influence of chemical and photochemical oxidation on NDMA formation. The water was subjected to various chemical oxidants that reacted with NOM as well as simulated sunlight to photo-oxidize some NOM. The influence on NDMA formation was also examined for relationships to changes in the UV spectra.

Source Waters and the Collection, Concentration, and Fractionation of NOM
Natural Organic Matter Sources. This study utilized both concentrated natural organic matter (NOM) and whole natural water samples from two typical agriculturally impacted river sources, a pristine "colored water" wetland, and drinking water obtained from several utilities.

Whole water and NOM concentrates were obtained from the Iowa River (IRW) and from Valentine Pond in the Keweenaw Peninsula of Upper Michigan (UPW). While the first source is from a river highly impacted by agricultural practices that result in significant additions of inorganic and organic nitrogen, the latter is located in a largely uninhabited and pristine area. Therefore, comparisons are expected to yield valuable information on possible human/animal sources of NDMA precursors. Additional whole water samples were collected from the Cedar River (CRW) and from a surface water source for a drinking water utility in the State of Ohio provided by Malcolm Pirnie Company (MPW).

Concentration of NOM by Reverse Osmosis (RO). NOM was concentrated by reverse osmosis. The reverse osmosis system used in this study was a RealSoft PROS/2S reverse osmosis unit (Stone Mountain, GA). To prevent fouling of the membranes by organic and inorganic constituents, the raw water was filtered through a 5 μm prefilter, 0.45 μm filter, and then passed through Dowex Marathon C cation exchange resin (Dow Chemical, Midland, MI). The RO unit's permeate to retenate flow ratio were maintained at 1:7 ratio at 200 psi. Other researchers who used similar RO units for isolation of Suwannee River DOC, as well as other surface and groundwater DOC, recovered 90% of the NOM at a process rate of 150-180 L/h (Serkiz and Perdue, 1990; Sun et al., 1995). Kitis et al. (2001) showed that reverse osmosis isolation maintains the integrity and reactivity of the NOM with respect to chlorine demand and selected DBP formation. Figure 1 shows a simplified schematic of the pre-filtration and RO unit.

Iowa River water (IRW) sample was drawn directly from the river surface, [DOC] = 3.4 mg-C/L, and seven hundred liters of water was collected and concentrated in February 2004. The Upper Peninsula of Michigan water (UPW) had a DOC concentration of 9.55 mg-C/L. 80 liters of UPW sample was processed in July 2004.

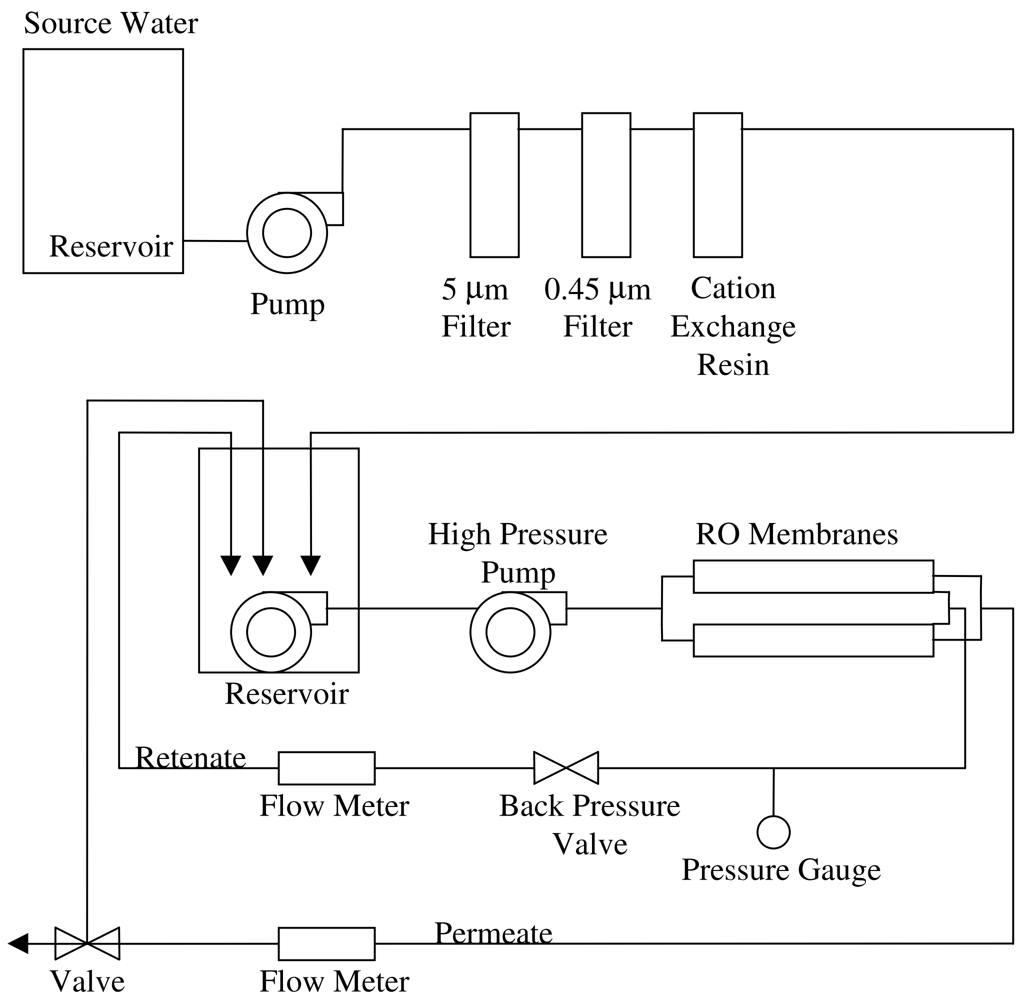


Fig. 1. Schematic of pre-filtration and reverse osmosis unit used to filter and concentrate NOM from source waters

Fractionation of NOM. In this study, the original NOM fractionation method proposed by Leenheer (1981) was adopted with slight modification to separate Iowa River water (IRW) NOM into two categories: hydrophobic and hydrophilic portions. Each portion was also further fractionated into acidic, basic, and neutral fractions by selective elution procedures (Figure 2).

Specifically, the collected IRW NOM concentrate was passed through 0.45 µm membrane before being pumped into a XAD-8 column at 30 bed volumes/h. At saturation, hydrophobic base was eluted with back flush of 0.25 bed volumes of 0.1 N HCl followed by 1.5 bed volumes of 0.01 N HCl. Effluent from the 1st XAD-8 resin was acidified to pH 2.0 with HCl and pumped onto 2nd XAD-8 column at 30 bed volumes/h. After rinsing the column with 1 bed volume of 0.01 N HCl, hydrophobic acid was eluted with back flush of 0.25 bed volume of 0.1 N NaOH, followed by 1.5 bed volume of DI water. Pump both the XAD-8 resin columns dry and hydrophobic neutral fraction was obtained by Soxhlet extracting the resins with methanol. After freeze drying, the hydrophobic neutral was dissolved back into water. Effluent from the 2nd XAD-8 was pumped through H-saturated AG-MP-50 cation exchange resin and hydrophilic-base was forward eluted with 1.0 N NH_4OH. Then the effluent was pumped through Duolite A-7 anion exchange resin and hydrophilic acid was obtained by back flush with 3 N NH_4OH. Only hydrophilic neutral was left in the effluent. All fractions were concentrated into small volumes by rotovapping below 40 °C.

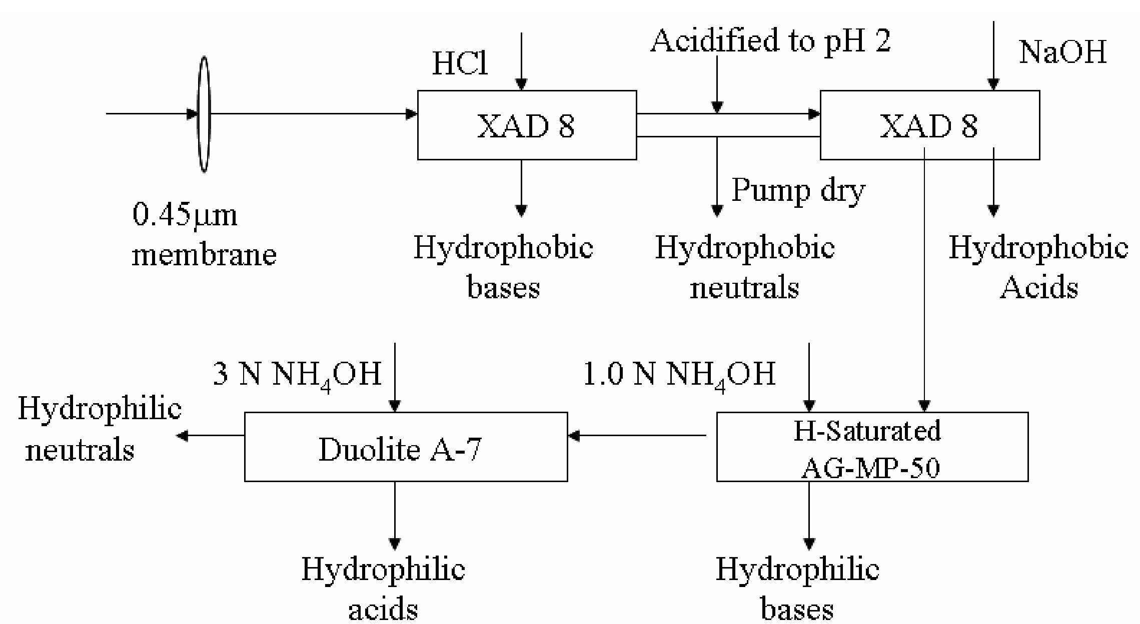


Fig. 2. Schematic for fractionation of NOM (Leenheer, 1981)

Characterization of NOM. DOC measurements, UV spectroscopy, and fluorescence spectroscopy were used to characterize NOM. Dissolved organic carbon (DOC) was measured using a Shimadzu TOC 5000 (Shimadzu Scientific, Columbia, MD) and standardized according to Standard Method 505A (APHA et al., 1992). The detection limits using the platinum catalyst combustion was determined to be 0.25 mg-C/L. UV absorbance and spectral characteristics of the natural organic mater were obtained with a Shimadzu UV1601 dual beam spectrophotometer. Fluorescence spectra were obtained using Perkin-Elmer LS55 Luminescence Spectrometer. Excitation-emission matrix (EEM) spectra were collected with subsequent scanning emission spectra from 290 nm to 600 nm at 10 nm increments by varying the excitation wavelength from 200 nm to 400 nm at 10 nm increments.

Reactors and Experimental Approach

Glassware used in the experiments was washed thoroughly with warm tap water, soaked in a nitric acid bath, and then rinsed again with copious quantities of deionized (DI) water. After washing, the glassware was baked in a muffle furnace at 500^{o}C for 1 hour. All solutions were prepared using DI water obtained from a Barnstead ROPure Infinity™/NANOPure Diamond™ system (Barnstead/Thermolyne Corp., Dubuque, IA). This treatment system produced water with a target resistivity of 18.2 mΩ-cm and [TOC] ≤ 3 ppb. All chemicals purchased and used in this study were ACS reagent grade and properly stored.

The basis of the NDMA FP test has been described by Mitch and Sedlak, 2004. It is based upon addition of preformed monochloramine to water samples and a 7-day reaction time. The use of preformed monochloramine instead of the usual practice of in-situ formation by addition of free chlorine and followed by ammonia addition maximizes NDMA formation. Additional work (results not shown) indicated that NDMA formation was reduced significantly by pre-chlorination. While this suggests a strategy to reduce NDMA formation, it nonetheless creates an artifact difficult to control.

The first step in preparing the reactors was filtering the water samples through Millipore AP25 glass fiber filter designed to remove particles greater than 0.8-1.6 μm in size. Next, the filtered water was measured into the reactor jars so that the final volume of the solution containing sample water, buffer, and NH_2Cl would be 500 mL. Concentrated buffer solutions were used to create pH stability in the reactors throughout the incubation period. A 10 mM phosphate buffer was used for the lower pH 7 while 10 mM bicarbonate was added to maintain a pH near 8. These two pH levels were used in the agriculturally impacted surface water samples, however only the bicarbonate buffer was used for the riverbank filtration reactors. Blank samples for every site were amended with the appropriate buffer to form the same concentration as the reactors spiked with 1mM NH_2Cl. Duplicates were run when there was sufficient sample volume collected.

Concentrated monochloramine stock solution with a 0.1 Cl/N molar ratio was prepared fresh prior to each experiment by addition of reagent grade ammonia and hypochlorous acid to a pH 10 solution containing 10 mM bicarbonate. Dosages added to the reactors ranged from 0.05 mM to 1 mM. The NH_2Cl was measured in control samples on a daily basis in order to ensure that high levels were present throughout the incubation period. NH_2Cl concentrations were determined by DPD-FAS titrimetric method (APHA, AWWA, and WEF; 1998).

Experiments examining the kinetics of NDMA formation and the role of various humic fractions were initiated by addition of the preformed monochloramine at a high dose of 1 mM, or 71 mg/L as Cl_2 was added to all agricultural watershed samples (approximately 20 times that of the typical residual allowed in drinking water distribution systems as dictated by the Stage 1 Disinfection/Disinfectant By-Product Rule). A much lower dosage of 0.05 mM was added to some of the riverbank filtration samples to mimic typical dosages used to maintain a substantial residual throughout a distribution system (approximately 4 mg/L as Cl_2).

All samples were incubated in the dark for 7 days (168 hours) at 20^oC. Reducing light exposure to the samples as much as possible ensured that the quantity of NDMA formed would not be significantly affected by photodegradation. After the 7-day incubation period, the final pH was measured to ensure buffer stability and minimal pH variation throughout the assays. A solid/liquid extraction process was used to concentrate the NDMA formed into a small volume of methylene chloride for analysis (Luo and Clevenger, 2003 and Taguchi et al., 1994). An internal standard of d6-NDMA was added to each reactor to form a baseline concentration of 100 ng/L. This internal standard, in conjunction with calibration curves developed for each assay, facilitated the subsequent quantification of these experiments.

All experiments examining the role of prechlorination/preoxidation were conducted in batch reactors (one-liter capacity clear Pyrex bottles with PTFE screw caps). The pre-chlorination studies involved adding 0.03-0.08 mM free chlorine for a prescribed contact time (15 sec~1 h) followed by addition of ammonia at a Cl/N molar ratio of 1:1. Chemical pre-oxidants used included O_3, H_2O_2, and $KMnO_4$. The dosages of H_2O_2 were 3 mg/L and 30 mg/L, while the dosage of $KMnO_4$ was fixed at 10 mg/L. Ozone was formed freshly in distilled water at 12 mg/L using an ozonator (Model ss-150, Pillar Technologies, Hartland, WI) and added to the tested NOM solutions at O_3/DOC weight ratio of 2:1. A Suntest CPS+ sun light simulator (Atlas Electric Devices Co., Chicago, IL) was used for simulated solar irradiation at a constant irradiance of 250 W/m^2 to investigate the influence of photochemical oxidation. Quartz tubes were used for the irradiation and the irradiation time was fixed at 1 h. Following a pre-oxidant contact time of 1 h, preformed monochloramine was added (0.05 mM). The post-chloramination reaction time following the pre-chlorination and pre-oxidation step was allowed to extend for 24 to 120 hours. NDMA concentration was measured periodically and compared to that obtained in control experiments without pre-chlorination or pre-oxidation.

Analytical
Monochloramine was measured by DPD-FAS titration. NDMA was measured with a Varian GC CP3800 coupled with a Saturn 2200 MS/MS detector. The column used is Varian gas chromatograph which had a length of 30 m, film thickness of 0.25 μm and insider diameter of 0.25 mm. The general temperature ramping protocol for the GC started with a temperature of 35°C for 4 minutes. Next the temperature was increased to 140°C at a rate of 20°C/min. A secondary ramp elevated the temperature to 200°C at a rate of 50°C/min. This temperature was held for 9.55 minutes. The total time of each sample was 20 minutes. The Saturn 2200 MS/MS detector was used under the following settings: m/z 81 for quantification of d6-NDMA and m/z 75 for NDMA.

Principal Findings and Significance

Precursor Studies of the Formation of NDMA from Selected Humic Fractions
NDMA Formation Potential. NDMA formation potential tests (Mitch et al., 2003) were conducted using both whole Iowa River water, and reconstituted river water prepared using RO concentrate to replicate the whole water DOC concentration of 3.4 mg C/L. RO concentrated Iowa River NOM (IRW NOM) was further fractionated into six operationally-defined fractions using the standard procedure proposed by Leenheer (1981). Figure 3 is a pie chart showing DOC recovery of each fraction. Figure 3 clearly shows that hydrophobic acids (HPOA), which usually contain both humic acids and fulvic acids, is the major contributor of total DOC in the tested river water (72%). Hydrophobic bases (HPOB) and hydrophobic neutrals (HPON) contributed 1.98% and 1.1%, respectively. Hydrophilic fractions generally make up a smaller fraction of DOC than that due to hydrophobic fractions, which is shown here. Hydrophilic acids (HPIA) contributed 7.9%, while hydrophilic bases (HPIB) and hydrophilic neutrals (HPIN) contributed 4.6% and 0.8%, respectively, of the total DOC. A relatively small, 11% loss in total DOC was realized, probably due to the washing and elution of columns and evaporation during the Roto-vapping procedure.

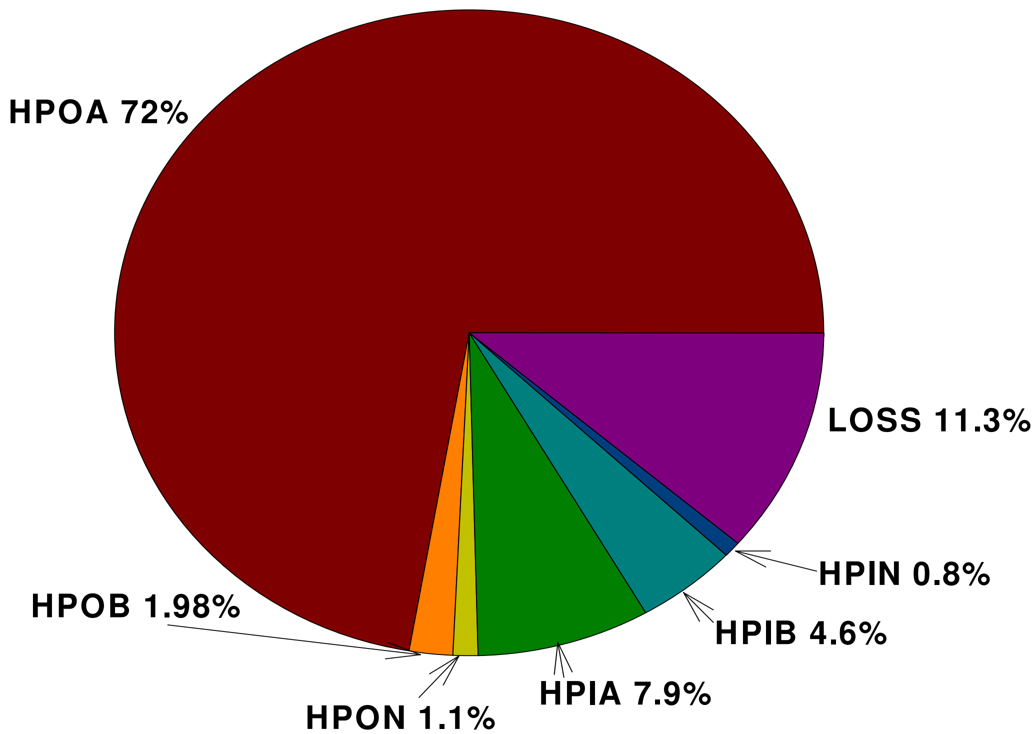


Fig. 3. DOC recovery of NOM fractionation (IRW NOM, fractionation using procedure proposed by Leenheer, 1981)

Figure 4 shows the NDMA formation potential in the original unprocessed whole water sample as a function of time. NDMA formation steadily increased with time attaining a value of 112 ng/L at the end of the 7-day test period. In comparison, approximately 100 ng/L of NDMA was formed in laboratory prepared water containing RO concentrate at the same DOC concentration as the unprocessed water. In addition, the concentration-time profiles were similar. This nearly 90% recovery of NDMA formation potential indicated that the RO concentration method that was used produced concentrated NOM from natural water with excellent preservation in reactivity. In other words, most NDMA precursors were captured and concentrated during this RO concentration procedure.

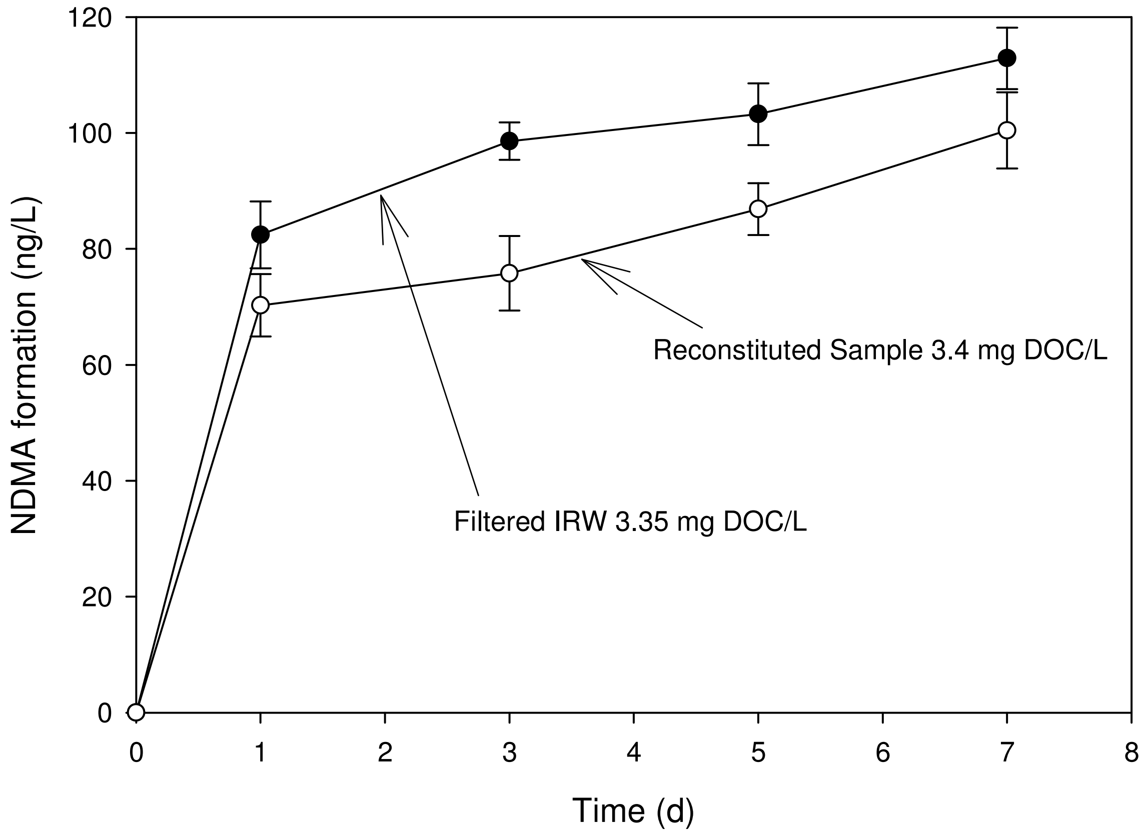


Fig. 4. Preservation of NDMA formation potential in RO concentrate water (1mM NH_2Cl, pH = 7.0, T = 20ºC)

The NDMA formation potential of each humic fraction was also determined. Table 1 tabulates the NDMA formation potential normalized to carbon content of each fraction (nanograms of NDMA formed per milligram of DOC). For example, 77.5 nanograms of NDMA was formed from every milligram of carbon in the hydrophilic bases (HPIB) fraction. Obviously HPIB showed the highest NDMA formation potential compared to other NOM fractions. Two trends are evident from the data. First, hydrophilic fractions tend to form more NDMA than hydrophobic fractions. Second, basic fractions tend to have a larger NDMA formation potential than acid fractions. This may be due to the higher nitrogen content in the polar hydrophilic and basic fractions (Croue et al., 1999).

Table 2 summarizes the expected contribution of all fractions to the NDMA formation potential measured in the whole water. In this table, the NDMA formation potential was calculated as normalized to the DOC concentration found in original river water (3.4 mg/L). After fractionation, based on the DOC percentage contribution of each fraction and the NDMA formation potential test results of each fraction (Table 1), the predicted NDMA FP from the sum of the six fractions is determined to be approximately 95 ng/L. This can be compared to the 112 ng/L in the original Iowa River water, which indicates excellent recovery of NDMA formation potential in both the RO water and in the humic fractions.

Table 1 List of NDMA formation potential of fractionated NOM fractions

Fractions	DOC contribution	NDMA FP* (ng/(mg DOC))	Total NDMA FP contribution
HPOA	72.00 %	27.47	71 %
HPOB	1.98 %	31.43	2.2 %
HPON	1.10 %	22.44	0.9 %
HPIA	7.90 %	43.50	12.3 %
HPIB	4.60 %	77.50	12.8 %
HPIN	0.80 %	25.76	0.7 %

FP*: Formation potential

Table 2 3.4 mg/L DOC equivalent NDMA formation potential

Samples	NDMA formation potential (ng/L)
Original river water	112
RO water	100
Sum of fractions	95

Relationship of Changes in SUVA to NDMA Formation. Aromatic moieties in NOM have been shown to affect UV absorbance spectrum of NOM (Chen et al., 2000; Vogt et al., 2004). The SUVA value at 254 nm and 272 nm are especially considered as good indicators of the aromaticity and reactivity of NOM. However, the relationship between the SUVA value of each NOM fraction and their individual NDMA formation potential indicated that the initial SUVA value of NOM is not a comprehensive index for NDMA formation potential. It certainly represents the aromaticity of NOM, but apparently not the functional groups that are responsible for NDMA formation.

Additional experiments were conducted to investigate the relationship between changes in the $SUVA_{272}$ value and the amount of NDMA formed in chloraminated water. This was done using reconstituted IRW NOM at the DOC level of 3.4 mg/L and dosed with 0.05 mM preformed NH_2Cl. Periodically, the monochloramine residual was quenched by addition of excess ascorbic acid and both NDMA concentration and $SUVA_{272}$ value were determined.

The formation of NDMA was found to correlate quantitatively with the change of $SUVA_{272}$ occurring during the course of the reaction. Figure 5 shows a linear relationship between the reduction of $SUVA_{272}$ and the formation of NDMA, suggesting the possibility of tracking NDMA formation by measuring changes in SUVA.

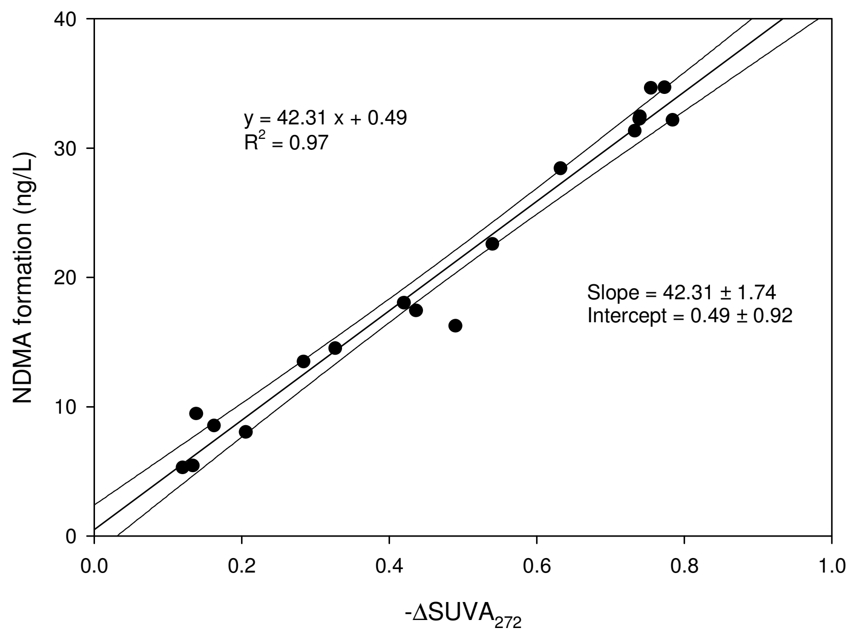


Fig. 5. Relationship between specific UV absorbance change of NOM and NDMA formation in the reaction of NOM with monochloramine (pH 7.0, T = 20ºC, IRWC NOM TOC = 3.5 mg/L, [NH2Cl]0 = 0.05 mM, slope and intercept are shown with their 95% confidence intervals)

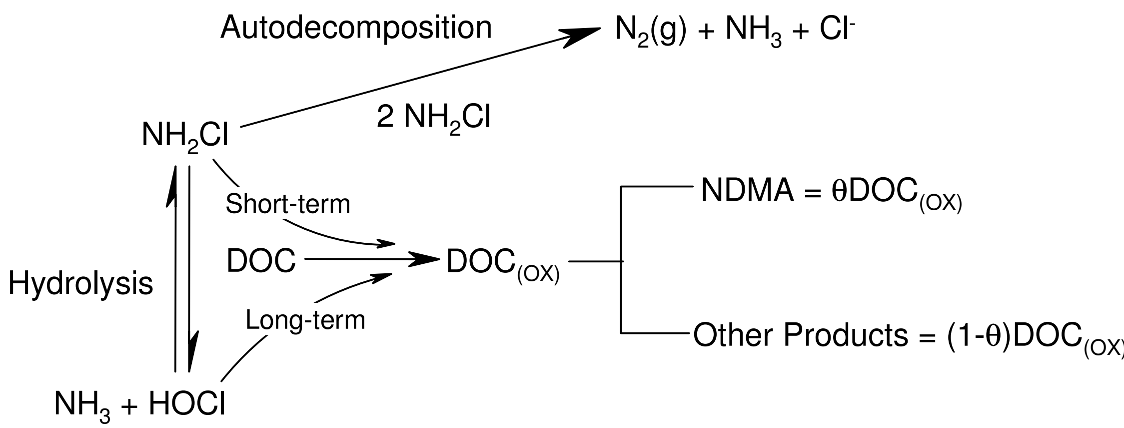
Interestingly, the SUVA value of each NOM fraction did not show any correlation with NDMA formation potential, while the $\Delta SUVA_{272}$ demonstrated a strong linear relationship with NDMA formation. As discussed earlier, SUVA has been extensively used as indicator of the reactivity of NOM with disinfectants (Westerhoff et al., 2004). Some studies directly used SUVA values as a predictor for the formation of halogenated DBP precursors (Korshin et al., 1997; Kitis et al., 2001). Duirk (2003) also demonstrated that $SUVA_{254}$ value could be used as a surrogating index for the reactivity of NOM with monochloramine. Nevertheless, the work reported here seems contradictory in that NDMA formation did not correlate with the SUVA value of the humic fractions, while the change in SUVA did correlate. This is probably attributable to structural differences in various humic fractions that lead to differences in precursor content. SUVA might be a good predictor of NDMA formation potential if all water sources had the same distribution of each humic fraction, and presumably the same distribution of potential NDMA precursors. Processes such as oxidation or removal of NOM that causes a reduction in SUVA may govern the rate of NDMA formation. But the SUVA changes from treatment that removes NOM would only be a good predictor of NDMA reduction if all fractions were removed to the same percent (Croue et al., 1999; Allpike, et al., 2005).

Mechanisms and Kinetics of NDMA Formation by Reaction of NOM with Monochloramine

Model Development. The formation of NDMA is a relatively slow process generally occurring on a time scale of days (Mitch et al., 2003; Choi and Valentine, 2001), which is comparable to the time scale of monochloramine loss from autodecomposition (Vikesland et al., 2001) and from oxidation of NOM (Vikesland et al., 1998; Duirk et al., 2005). The monochloramine-NOM reaction model proposed by Duirk (2003) was used to describe these parallel processes to predict both the concentration of monochloramine and the amount of NOM oxidized with time. A schematic of the proposed model is shown in Figure 6.

Duirk et al. (2005) modified a description of monochloramine autodecomposition (Figure 6) by incorporating a simple second-order kinetic description of NOM oxidation involving two types of reactive sites (reactions 11–12). Site type S_1 represents a comparatively small fraction of reactive dissolved organic carbon (DOC) that reacts very rapidly producing an "initial chloramine demand." Site type S_2, comprising more than 90% of the reactive DOC, reacts quite slowly (over days), accounting for most of the monochloramine loss typically observed over a five-day period. While the rate constants k_1 and k_2 are associated with these two types of reactive sites respectively, constant k_1 is simply set conveniently high so that the demand due to site type S_1 is accounted for rapidly, within an hour. The value of k_2 on the other hand is critical to predicting the slow loss of monochloramine with time. NOM oxidation is primarily attributed to reaction involving a direct reaction of DOC with HOCl that exists in minute amount in chloramine solutions, produced from such reaction as hydrolysis of monochloramine (reaction 2).

Duirk et al. (2005) validated the model using several whole water sources, NOM concentrates obtained by reverse osmosis, and fractionated humic isolates. The model could account for dependencies on pH, ammonia, DOC, and monochloramine concentrations. Results were consistent with the assumption that humic-type substances were the dominant reactive type of NOM.



Note: θ: a simple stoichiometric coefficient

DOC: dissolved organic carbon

$DOC_{(OX)}$: oxidized dissolved organic carbon

Fig. 6. Model schematics of NDMA formation from reactions between NOM and monochloramine

This model was modified to account for NDMA formation by assuming that the rate of NDMA formation was proportional to the rate of NOM oxidation. This is similar to the approach recently taken to describe the formation of dichloroacetic acid from the oxidation of NOM (Duirk et al., 2002). A simple stoichiometric coefficient θ_{NDMA} was incorporated to linearly correlate the formation of NDMA with the oxidation of NOM. Other researchers have correlated the formation of trihalomethanes (THMs) and haloacetic acids (HAAs) with free chlorine demand, presumably from the oxidation of NOM (Clark et al., 2001; Gang et al., 2002). This is equivalent to the assumption that ratio of NDMA formed to NOM oxidized is a constant θ_{NDMA} over the entire course of the reaction (Equation 1):

$$\frac{d[NDMA]}{dt} = \theta_{NDMA} \frac{d[DOC]_{OX}}{dt} \qquad \text{Equation 1}$$

The rate of NDMA formation can therefore be expressed as equation 2 in terms of model parameters and variables:

$$\frac{d[NDMA]}{dt} = \theta_{NDMA}\{k_1[NH_2Cl][DOC]\times S_1 + k_2[HOCl][DOC]\times S_2\} \qquad \text{Equation 2}$$

S_1 = DOC short-term reactive site fraction

S_2 = DOC long-term reactive site fraction

k_1, k_2 = reaction rate constants

θ_{NDMA} = stoichiometric coefficient

All constants characterizing monochloramine autodecomposition were obtained from literature (sources shown in Table 3). Only k_1, S_1, k_2, S_2 and θ_{NDMA} were determined for this model as NOM source specific parameters. The model was expressed as a system of ordinary differential equations (ODEs), the whole set of which was solved using Scientist^{TM} (Scientist, 1995). Scientist^{TM} uses a modified Powell algorithm to minimize the unweighted sum of the squares of the residual error between the predicted and experimentally determined values to estimate the model parameters.

The comprehensive reaction model differentiates monochloramine loss into two pathways: auto-decomposition and oxidation of NOM. The initial modeling activity required determining the parameters that characterize monochloramine loss independently of the value of θ_{NDMA}, which is only used to predict NDMA formation. The final estimated model parameters for each particular NOM source, obtained by averaging parameter estimates for all data sets for the same NOM source, are listed in Table 3. The model estimation of NOM reactive sites are consistent with the total reactive sites determined by free chlorine titration methodology. This shows that titrating the NOM with free chlorine serves as a good independent method to determine reactive site fractions.

Table 3 Model estimated parameters with their 95% confidence intervals for tested NOM sources including IRW, UPW and CRW NOM

Sources	IRW[1]	UPW[2]	CRW[3]
k_1	$1.21\times10^{4}\pm4.30\times10^{3}$	$1.12\times10^{4}\pm6.60\times10^{3}$	$1.43\times10^{4}\pm3.07\times10^{3}$
S_1	0.011 ± 0.003	0.009 ± 0.003	0.015 ± 0.006
k_2	$5.92\times10^{5}\pm1.87\times10^{4}$	$4.88\times10^{5}\pm1.72\times10^{4}$	$6.01\times10^{5}\pm6.30\times10^{4}$
S_2	0.53 ± 0.04	0.48 ± 0.04	0.62 ± 0.05
S_1+S_2	0.54 ± 0.04	0.57 ± 0.04	0.63 ± 0.06
S_T[4]	0.59	0.45	0.65
θ_{NDMA}	$2.85\times10^{-5}\pm5.17\times10^{-6}$	$1.67\times10^{-5}\pm4.10\times10^{-6}$	$3.23\times10^{-5}\pm2.19\times10^{-6}$

1: IRW: Iowa River water, n = 10
2: UPW: Valentine Pond water in Upper Peninsular of Michigan, n = 7
3: CRW: Cedar River water, n = 4
4: Total reactive sites determined by free chlorine titration

NDMA Formation. With the establishment of the model's capability to describe monochloramine loss in the presence of NOM (data not shown), the model was used to calculate the amount of NOM oxidized (in terms of equivalent amounts of monochloramine) as a function of reaction conditions and time. Figure 7 shows a linear relationship between NDMA formed and NOM oxidized for IRW water. A similar relationship was found for the other two source waters. This linear relationship is consistent with the assumption that the rate of NDMA formation is given by Equation 1 and that the value of θ_{NDMA} is a source specific constant that is independent of pH and hence the rate of NOM oxidation.

Figure 8 compares measured and predicted NDMA formation from IRW at DOC concentrations of 1.7 to 6.8 mg/L, which are in the typical DOC range representative of many water supplies (Roth and Ozment, 1998). A higher concentration of DOC resulted in higher NDMA formation. During the early stage of the reaction course, the model appears to slightly underestimate NDMA production at low DOC levels and slightly overestimate it at high DOC levels. But overall, the model predicted NDMA formation quite well. Figure 9 shows the influence of pH from 6.0 to 9.0 on NDMA formation. A good match between experimental data and model prediction is again indicated. The effect of free ammonia on NDMA formation was also investigated using a fixed initial monochloramine concentration and chlorine to ammonia molar ratios ranging from 0.1 to 0.7 (Figure 10). Results show that increasing ammonia reduces the rate of NDMA formation, consistent with the expected influence of ammonia on the rate of NOM oxidation.

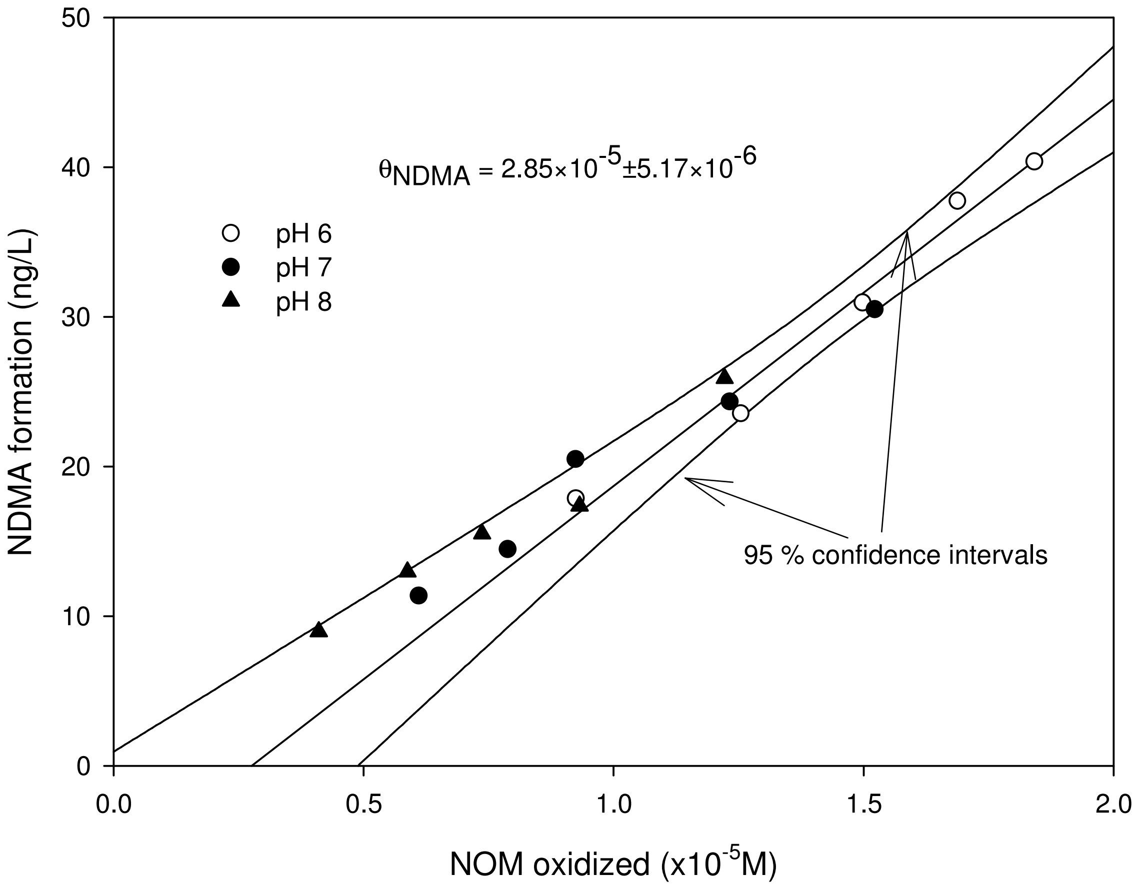


Fig. 7. Linear relationship between NDMA formation and NOM oxidation and determination of NDMA with its 95% confidence intervals (IRW concentrate DOC = 3.4 mg/L, [NH2Cl]0 = 0.05 mM, I = 8 mM)

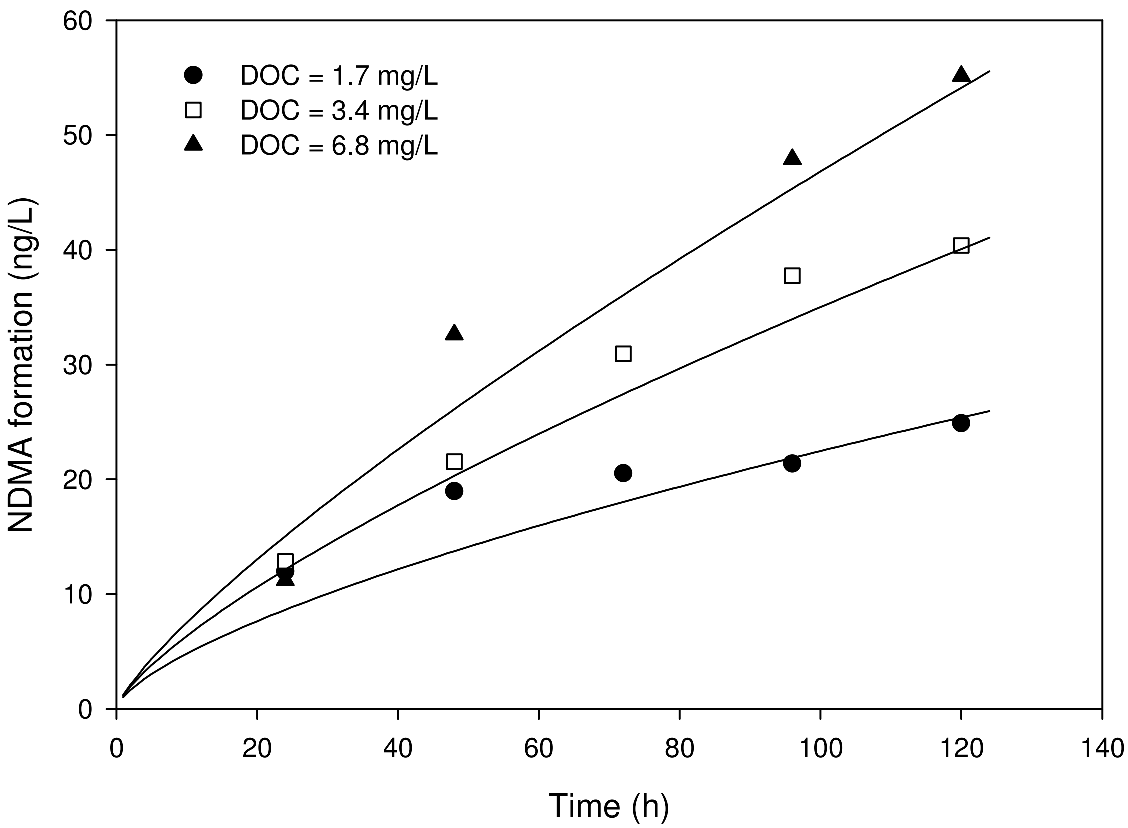


Fig. 8. Model prediction of NDMA formation at various DOC levels from the reactions between monochloramine and NOM (IRW concentrate, pH = 7.0, [NH2Cl]0 = 0.05 mM, I = 8 mM,line represents model prediction)

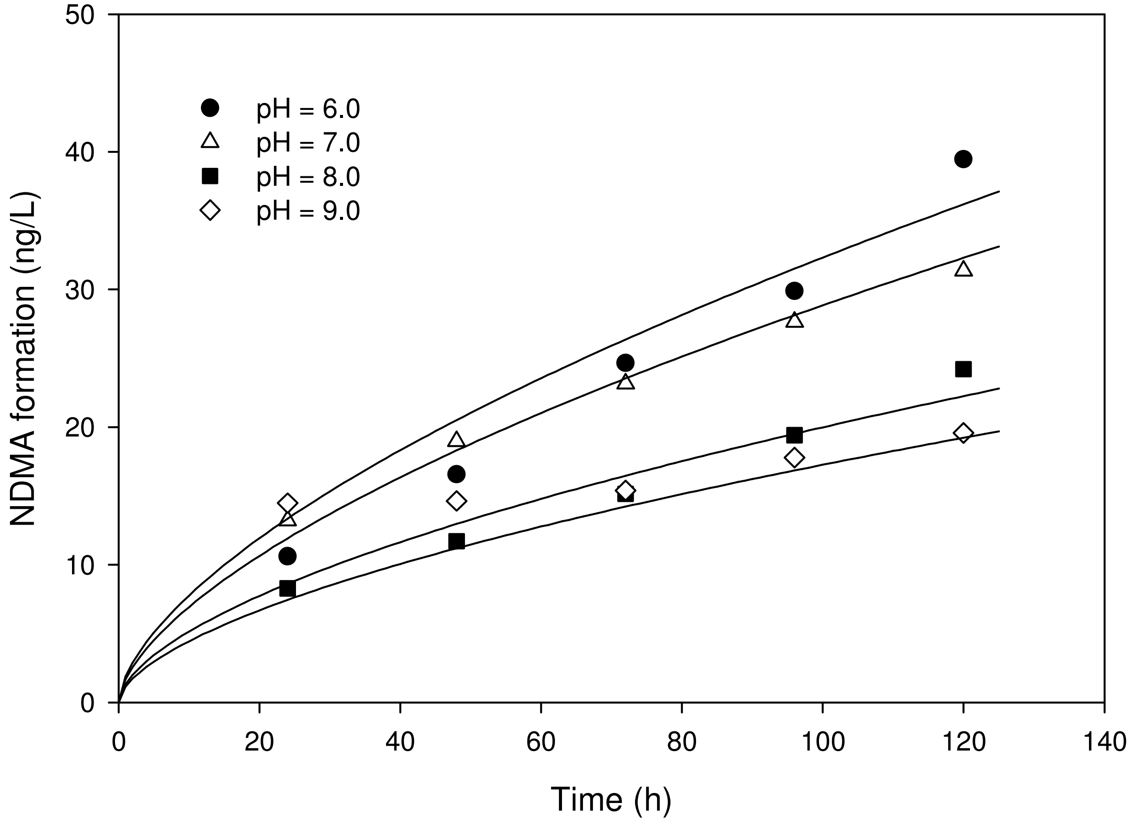


Fig. 9. Model prediction of NDMA formation at various pH values from the reactions between monochloramine and NOM (IRW concentrate DOC = 3.4 mg/L, $[NH2Cl]0$ = 0.05 mM, I = 8 mM, line represents model prediction)

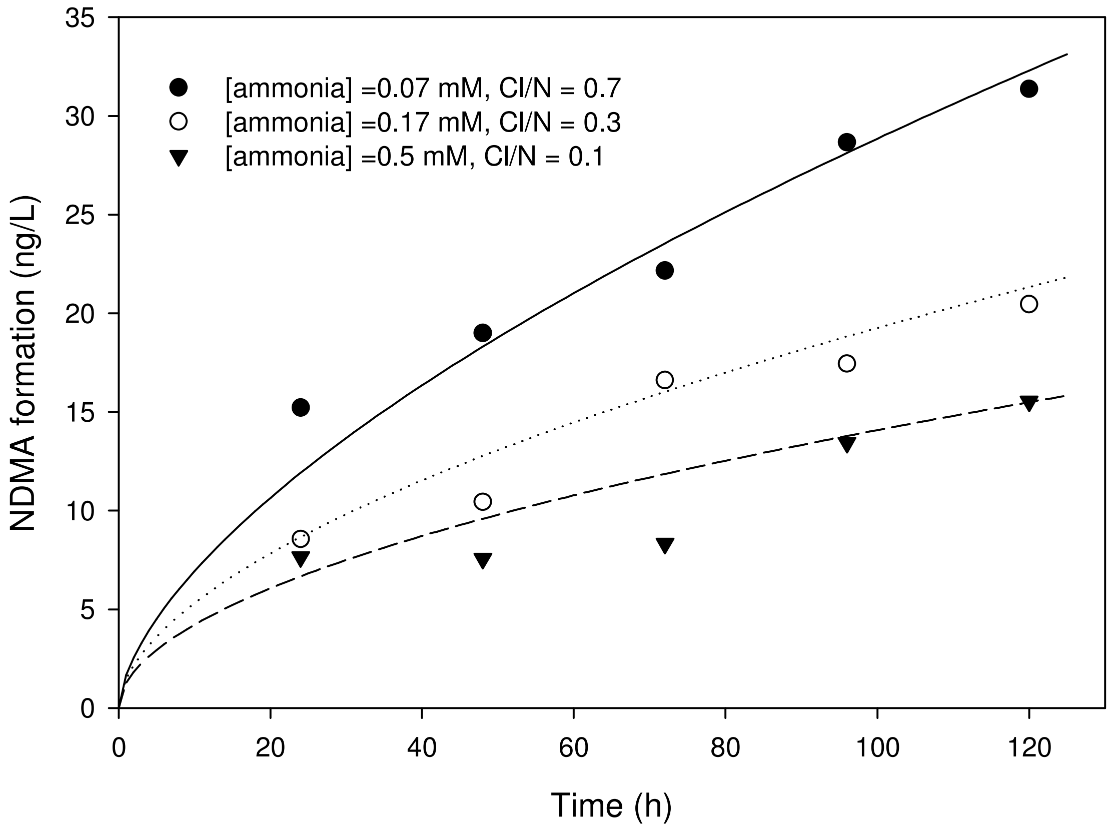


Fig. 10. Effect of ammonia on NDMA formation (IRW NOM DOC = 3.4 mg/L, pH = 7.0, $[NH2Cl]_0 = 0.05$ mM, I = 8 mM, line represents model results)

Comparison of Model Parameters. Comparing the estimated model parameters for tested three NOM sources, all the rate constants as well as the site fraction values were remarkably similar, within a factor of 25% of each other (Table 3), indicating very similar NOM oxidation characteristics among tested NOM sources. The value of θ_{NDMA} however, varied by a factor of two, ranging from a high of 3.23×10^{-5} for source CRW NOM to a low value of 1.67×10^{-5} for water containing UPW NOM. While the values for CRW and IRW derived NOM were statistically similar, the value of θ_{NDMA} for UPW NOM was significantly smaller. This difference presumably reflects differences in the nature of the NDMA precursors. Perhaps this difference is indicative of differences in NOM characteristics, or, conversely, the presence of other types of precursors, possibly associated with agricultural discharges into the two rivers.

Influence of Chemical and Photochemical Oxidation on NDMA Formation

Influence of Pre-chlorination on NDMA Formation. NDMA formation decreased with the application of free chlorine prior to the addition of ammonia to produce chloramines. For example, the amount of NDMA formed after 120 hours in chloramine containing water was reduced by 50% by a free chlorine dosage of 0.08 mM and 10 minutes of contact time prior to chloramination. Increasing the pre-chlorination contact time also resulted in a further reduction in NDMA formation (Figure 11). It should be pointed out that the free chlorine demand was comparable in these studies so that the total initial monochloramine concentration was comparable in all experiments. Therefore the reduction in NDMA formation caused by pre-chlorination cannot be attributed to a lower initial chloramine concentration. NDMA formation also decreased with increasing chlorine dosage at a fixed free chlorine contact time (Figure 12). This finding is the opposite of that observed for the influence of pre-chlorination on the formation of many halogenated DBPs (Eldib and Ali, 1995).

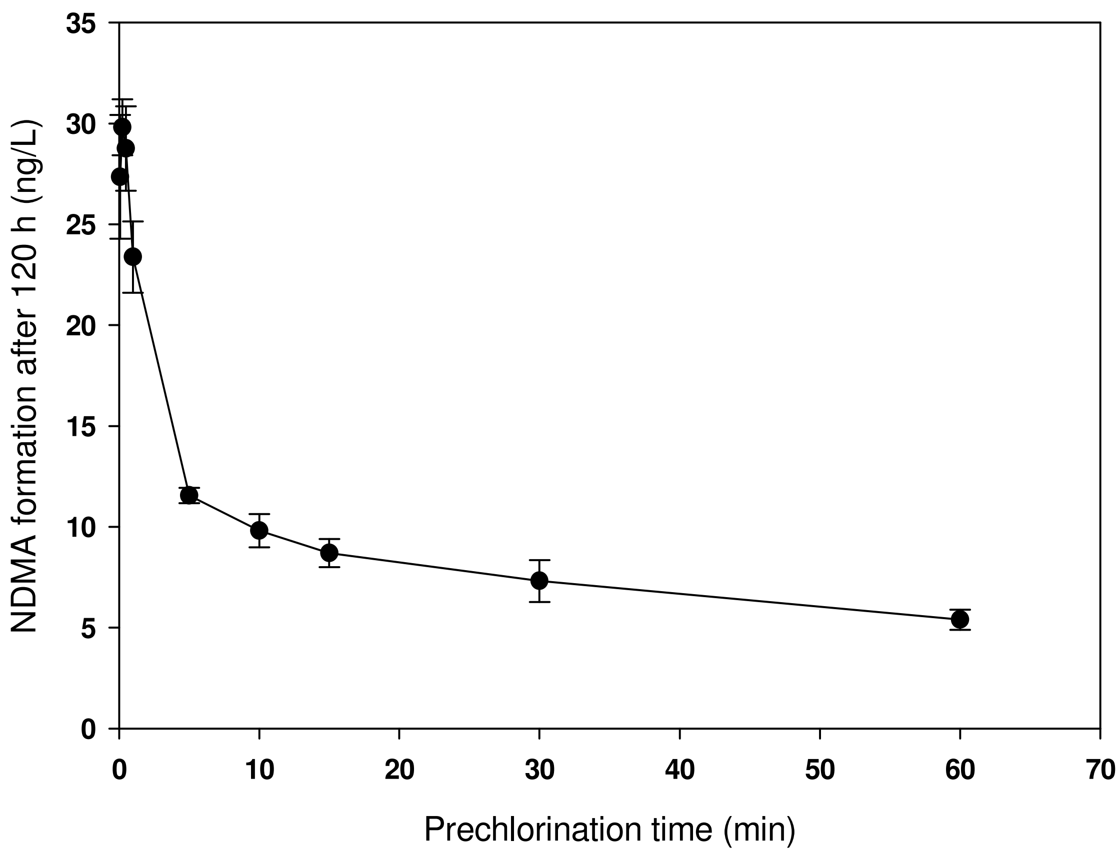


Fig. 11. Effect of pre-chlorination time on NDMA formation (IRW NOM DOC = 3.4 mg/L, pH = 7.0±0.2, Cl2 = 0.08 mM, Cl/N = 0.7)

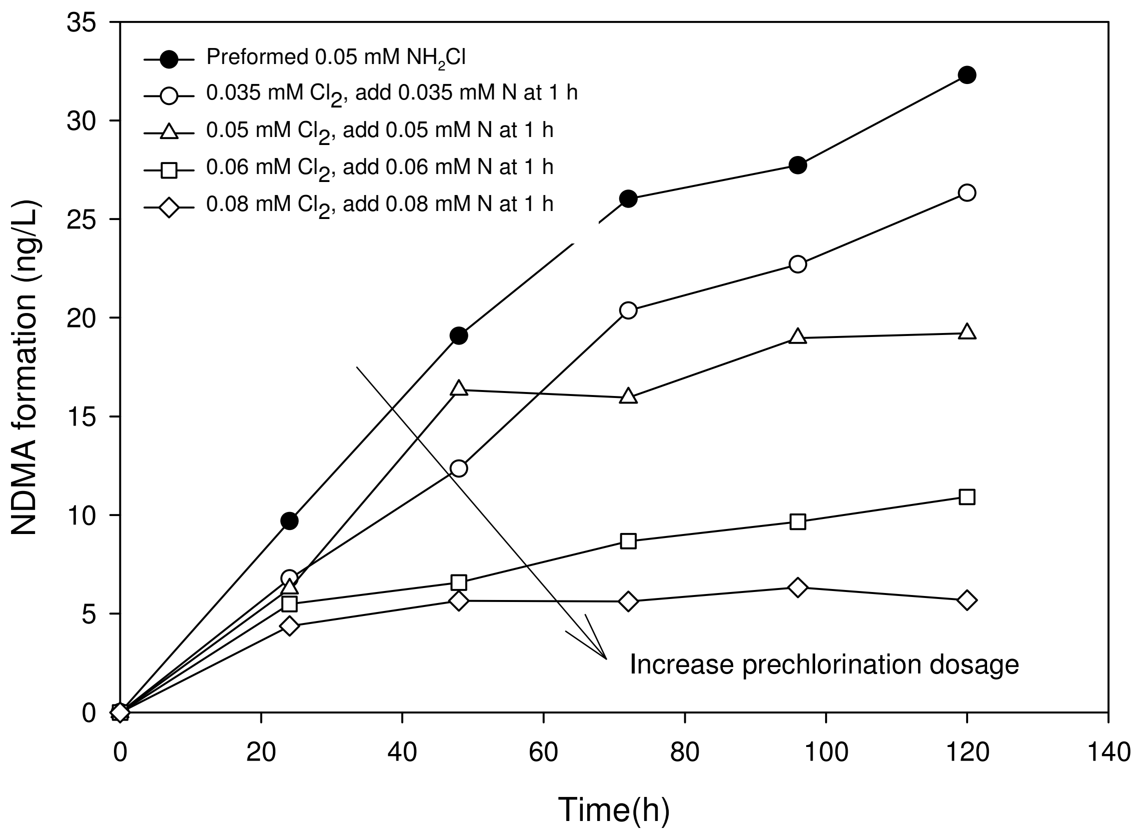


Fig. 12. Effect of pre-chlorination dose on NDMA formation (IRW NOM DOC = 3.4 mg/L, 4 mM NaHCO3, pH = 7.0, pre-chlorination time = 1 h, Cl/N = 0.7)

A similar trend of decreasing NDMA concentration with pre-chlorination was obtained using the whole surface water for a drinking water utility in the State of Ohio provided by Malcolm Pirnie Company (data not shown). After 108 minutes of pre-chlorination, NDMA formation after 5 days contact with chloramine was reduced approximately 50 %. Increasing pre-chlorination time to 345 minutes resulted in only a moderate further decrease in NDMA formation indicating a rapidly decreasing effect of time.

Influence of Other Pre-oxidants on NDMA Formation. Figure 13 shows the effect of pre-oxidation on NDMA formation in water containing RO concentrated IRW NOM. Similar to the effect of pre-chlorination, water subjected to pre-oxidation produced significantly less NDMA during the subsequent contact period with preformed monochloramine. For example, the application of 10 mg/L $KMnO_4$ or 3 mg/L H_2O_2 reduced NDMA concentration by about 50% during the subsequent contact with monochloramine compared to that obtained without pre-oxidation. Increasing the concentration of pre-oxidant also resulted in a further decrease in the NDMA formation. The influence of both pre-chlorination and pre-oxidation on reducing NDMA formation suggests a common mode of action. It is hypothesized that this is due to the destruction of NDMA precursors.

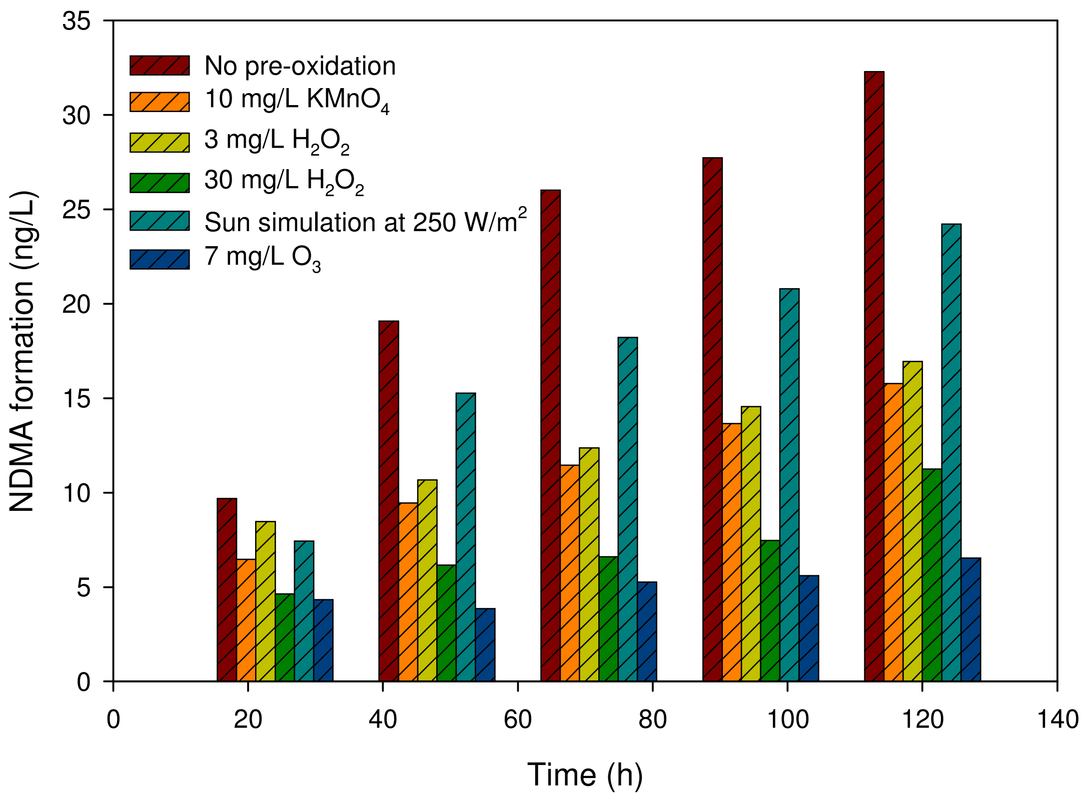


Fig. 13. Effect of pre-oxidation on NDMA formation during post-chloramination (IRW NOM, DOC = 3.4 mg/L, pH = 7.0±0.2, pre-oxidation time = 1 h, [NH2Cl]0 = 0.05 mM)

Correlation of NDMA Formation to Changes in SUVA. NDMA formation in water not subjected to pre-chlorination or pre-oxidation was found to correlate linearly with the simultaneous decrease in $SUVA_{272}$ caused by the slow oxidation of NOM by monochloramine (Figure 14). This finding is similar to that obtained by Korshin et al. (1997) and Kitis et al. (2001) for the formation of total organic halogen (TOX) and haloacetic acids (HAAs). This strongly suggests that SUVA is also a good indicator of NDMA precursor content.

Interestingly, when water was first subjected to pre-chlorination, a decrease in NDMA formation is observed as expected, and the relationship between NDMA formation and the post-chloramination induced SUVA change was still linear but with a smaller slope. This may be attributable to the preferential destruction of a fraction of NOM that contributes more to the NDMA precursor content. This is consistent with the studies on NDMA precursors that indicates different NDMA formation potentials associated with different humic fractions.

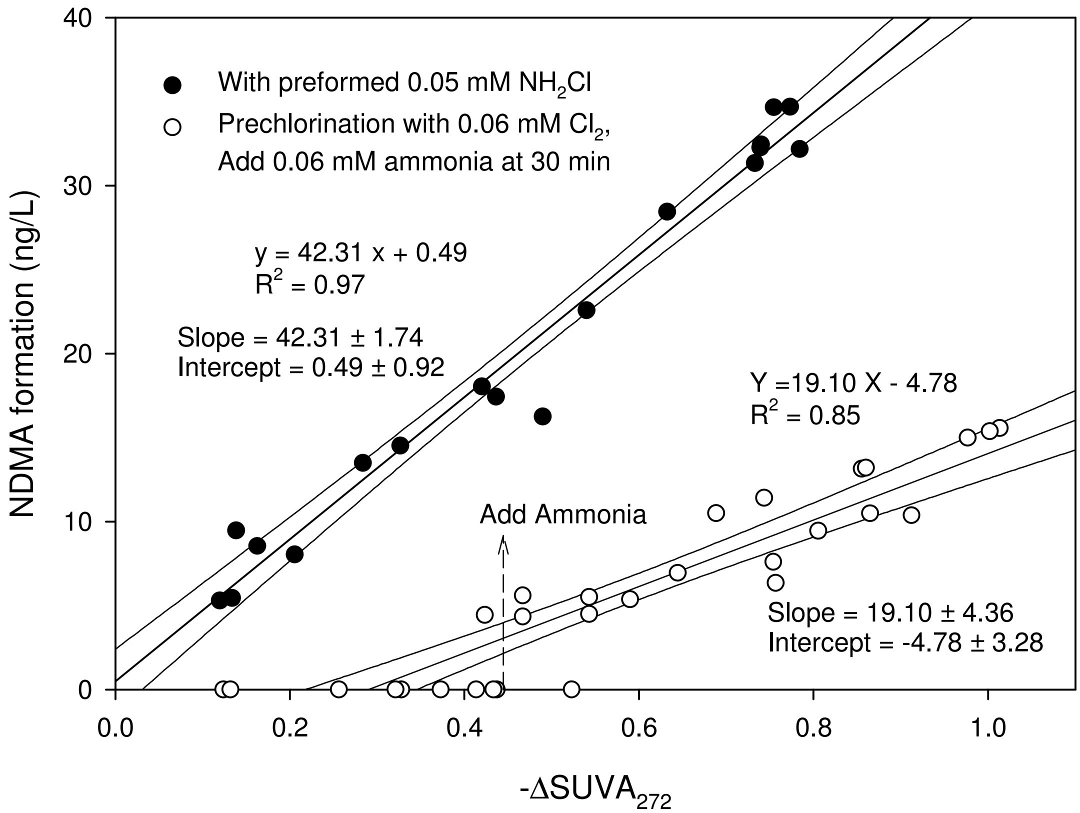


Fig. 14. Relationship between SUVA272 and NDMA formation (IRW NOM DOC = 3.4 mg/L, Cl2 dosage = 0.06 mM, pre-chlorination time = 0.5 h, pre-formed [NH2Cl]0 = 0.05 mM, slopes and intercepts are shown with their 95% confidence intervals)

The amount of NDMA formed after 120 hours contact with chloramine in water was reduced by an amount proportional to the decrease in $SUVA_{272}$ caused by pre-chlorination or pre-oxidation (Figure 15). Quite unexpected is the observation that all pre-treatments have the same influence on reducing NDMA formation when normalized to the change in SUVA caused by pre-oxidation/pre-chlorination. This same relationship held even for a sunlight induced SUVA loss. Clearly a common mode of action is indicated which appears to be the destruction of precursors as indicated by a loss in SUVA. How this SUVA reduction is achieved does not influence the resulting loss in NDMA formation.

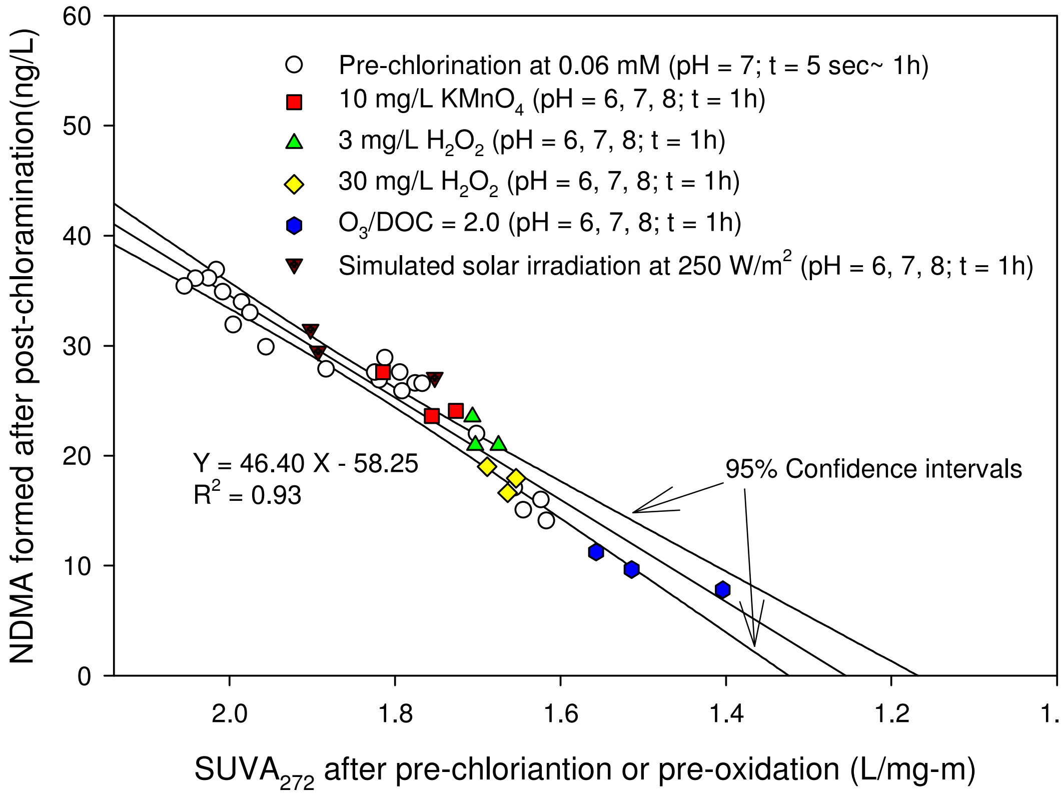


Fig. 15. Correlation between the change in SUVA272 and NDMA formation (IRW NOM DOC = 3.4 mg/L, pH = 6~8, pre-oxidation time = 5 sec~1h, chloramine contact time = 120 hours, [NH2Cl]0 = 0.05 mM)

Summary and Conclusions

A primary purpose of this study was to investigate the mechanisms and kinetics of NDMA formation in natural water typical of that used as a drinking water source. Studies on the nature of the precursors showed that they are conserved when the NOM is concentrated by reverse osmosis (RO) and isolated into different humic fractions. Different humic fractions exhibited different NDMA formation potentials when normalized to carbon content. This indicates a fraction-dependent precursor content. Hydrophilic fractions were found to form more NDMA than hydrophobic fractions and basic fractions tend to form more NDMA than acid fractions. The dominant source of NDMA precursors in river water was determined to be the hydrophobic acid (HPOA) fraction of NOM because it is also the dominant fraction comprising NOM.

A reaction model incorporating monochloramine auto-decomposition, NOM oxidation, and NDMA formation using five water-specific parameters, provided validation for hypothesized reactions. Most important is the hypothesis that the rate of NDMA formation is linearly related to the rate of NOM oxidation and therefore NDMA formation can be described by a simple stoichiometric relationship between NOM oxidized and NDMA formed. This is consistent with the linear relationship between the reduction in $SUVA_{272}$ that occurs with the oxidation of NOM and the formation of NDMA. Surprisingly, the five parameters that determine the rate of NOM oxidation and NDMA formation were very similar for samples obtained from a variety of sources, including a pristine source not impacted by agricultural practices or waste discharges. This implies that NDMA formation in the agriculturally impacted surface water sample is not likely to be governed by special agriculture-associated precursors. However, this finding may be a consequence of the particular season in which the river samples were taken.

Pre-chlorination and pre-oxidation were shown to be effective in reducing the potential of NOM to form NDMA from subsequent reactions in the presence of chloramines. Although the direct mechanism is not clear, NOM fluorescence spectra and UV absorbance changes indicate a loss of aromaticity upon oxidation, which appears indicative of a depletion or destruction of NDMA precursors. This also shows that free chlorine itself can play a dual role in influencing NDMA formation. The experimental and modeling work clearly shows that the oxidation of NOM by trace levels of free chlorine existing in the presence of monochloramine is required to produce NDMA presumably through the formation of precursors. But in high concentrations, typically associated with the practice of pre-chlorination, it destroys precursors.

The reduction in NDMA formed after 5 days in the presence of chloramines was found to be linearly correlated to the pre-oxidation and pre-chlorination induced reduction in $SUVA_{272}$. Quite unexpectedly, this relationship is independent of the nature of the oxidant. The reduction in NDMA formation with reduction in $SUVA_{272}$ caused by various oxidants including free chlorine, hydrogen peroxide, potassium permanganate, ozone, and simulated sunlight all fit on a single regression line. This suggests that the change in $SUVA_{272}$ could possibly be used as a parameter to predict changes in NDMA formation when pre-oxidation including pre-chlorination is practiced.

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An Integrated Immunological-GIS Approach for Bio-monitoring of Ecological Impacts of Swine Manure Pollutants in Streams

Basic Information

Title:	An Integrated Immunological-GIS Approach for Bio-monitoring of Ecological Impacts of Swine Manure Pollutants in Streams
Project Number:	2002IA25G
Start Date:	9/15/2002
End Date:	9/14/2006
Funding Source:	104G
Congressional District:	Iowa 3rd
Research Category:	Water Quality
Focus Category:	Methods, Non Point Pollution, Agriculture
Descriptors:	water quality, fish, immunology, GIS, swine manure, non-point source pollution
Principal Investigators:	James A. Roth, Bruce Willard Menzel, Dusan Palic, Clay Lynn Pierce

Publication

1. Palic, D., C.B. Andreasen, D.E. Frank, B.W. Menzel, and J.A. Roth. (2005). A rapid, direct assay to measure degranulation of primary granules in neutrophils from kidney of fathead minnow (Pimephales promelas Rafinesque, 1820). Fish and Shellfish Immunology 19(3), 217-227.
2. Palic, D., C.B. Andreasen, D.E. Frank, B.W. Menzel, and J.A. Roth. (2005). Gradient separation and cytochemical characterization of neutrophils from kidney of fathead minnow (Pimephales promelas Rafinesque, 1820). Fish and Shellfish Immunology 18(3), 263-267.
3. Palic, D., C.B. Andreasen, D.M. Herolt, B.W. Menzel, and J.A. Roth, 2006. Immunomodulatory Effects of B-glucan on Neutrophil Function in Fathead Minnows (Pimephales promelas Rafinesque, 1820). Developmental and Comparative Immunology (In press).
4. Palic, D., D.M. Herolt, C.B. Andreasen, B.W. Menzel, and J.A. Roth, 2006. Anesthetic Efficacy of Tricaine Methanesulphonate, Metomidate and Eugenol: Effects on Plasma Cortisol Levels and Neutrophol Function in Fathead Minnow (Pimephales promelas Rafinesque, 1820). Aquaculture (In press).

An Integrated Immunological-GIS Approach for Bio-monitoring of Ecological Impacts of Swine Manure Pollutants in Streams

James A. Roth, Dusan Palić, Bruce W. Menzel, Clay L. Pierce

Problem and Research Objectives

Thirty years after enactment of the Clean Water Act, 40% of our nation's rivers, lakes, and coastal waters are still considered unfit for fishing, swimming, drinking or aquatic life. The U.S. EPA identified agricultural operations as the primary cause of non-point source pollution in the nation's impaired rivers and lakes. At least 10% of the nation's impaired river miles are affected by pollution from livestock operations. In portions of the Midwest, confinement livestock operations are a particular problem in this regard. Cases of massive deaths of aquatic organisms, often referred to as fish kills, are an extreme manifestation of the ecological impact of fecal contamination. Typically, they result from high concentrations of toxic ammonia contained in the manure or from depletion of dissolved oxygen in the water caused by decomposition of the pollutant. Chronic effects of manure pollution are poorly known, because of the difficulty of measuring them and placing them in ecological context. Moreover, low-level delivery of fecal pollutants can portend larger catastrophic inputs, for example, when a gradually leaking storage lagoon eventually bursts or an erosive, manure-fertilized crop field receives heavy rainfall.

State and federal agencies engaged in reducing non-point source water pollution are interested in obtaining new technologies for identifying, measuring and anticipating pollution occurrence. Clearly, development of tools that could integrate biological and environmental information to produce site-specific predictive models for guiding pollution-prevention management practices is highly desirable. The proposed research would develop a novel tool that integrates molecular biological and ecological approaches to quantitatively evaluate environmental impacts of swine manure pollutants. Although the technique will be developed with specific reference to Midwestern waters, it will be more broadly applicable, both geographically and with reference to other forms of pollution that engender immune responses in animals. Thus, we believe that the technique has potential to be widely adopted by state and federal environmental management agencies.

The research conducted under this grant reflects the need for integrated, multidisciplinary approaches to deal with complex environmental issues. It combines physiological laboratory techniques, computer modeling of agricultural landscapes and non-point source pollution pathways, and field-based ecological analyses to create a new and integrated approach for evaluating impacts of livestock fecal contamination on Midwestern streams.

The research relates to two major priorities of the NIWR National Competitive Grants Program.

A) It will complement work by the USGS related to non-point source pollution, contributing to development of integrated watershed decision support tools for assessing organics and microorganisms transport and fate, along with their effects on aquatic systems.
B) It promises development of a new water quality sensor technology that will be based on integrated methodologies and will provide results that are readily accessible through the Internet.

This research is predicated on the hypothesis that low levels of swine liquid manure slurry and anaerobic lagoon liquid released to open water cause changes in immunological response in fish and increase fish susceptibility to infection.

The initial objectives, therefore, are: 1) to evaluate this hypothesis through a series of laboratory immunological assays applied to the test organism, the fathead minnow (*Pimephales promelas*); and 2) to identify one or more assays for use as a bio-monitoring technique to detect ecological impact of manure pollution in nature. A subsequent task involves use of digital environmental databases that are maintained and managed by the USGS BRD Iowa Cooperative Fish and Wildlife Research Unit at Iowa State University. The objective is 3) to characterize a number of Iowa watersheds and stream systems according to their potential susceptibility to hog manure pollution and to use this information to design a water quality and fish sampling regime. Finally, water and fish collected at selected stream sites will be analyzed through a battery of chemical and immunological procedures with the objectives 4) to quantitatively measure ecological impact of manure pollution on the streams, and 5) to evaluate the utility of this approach as a biomonitoring tool for environmental protection agencies.

Methodology

The fathead minnow is a native Iowa species, abundant and ubiquitous in small streams. Thus, it is a good choice as a representative of fish communities exposed to low level concentrations of swine manure pollutants released into Iowa waters. Moreover, it is commonly used as a standard bioassay organism in toxicological analyses, so there is substantial knowledge on its tolerance to a wide array of environmental physical conditions and pollutants. Additionally, colonies can be easily established and maintained in the laboratory. Fathead minnows used for the experiment will be raised in a controlled environment, without previous exposure to swine manure.

The immune response will be determined by activity of phagocytic cells, through several forms of measurement. Evidence from limited research involving fish suggests that assays measuring respiratory burst and degranulation are useful for determining phagocytic function in fish, and the procedure also seems to hold promise as a bio-indicator for fish health. Additionally, histological examinations of melano–macrophage centers (MMC) in liver, spleen, kidneys, intestines, skin and gills will be used to measure effects of long term exposure to manure.

Geographic Information Systems (GIS) technologies provide a tool to enter, store, manipulate and integrate geo-referenced data on, for example, landscape features, water quality, and aquatic organisms. The project will apply GIS technology and landscape modeling to calculate possible swine pollutant flow path patterns in Iowa watersheds having large hog confinements and in those where liquid manure fertilizer is applied on crop fields. Using this approach, we will estimate temporal and spatial distribution of manure loads and concentrations that reach receiving waters. This will provide the basis for a field sampling regime to determine actual conditions of water quality and fish communities at stream sites selected to represent a range of calculated manure pollutant loadings. Water quality data on fecal coliform bacteria, phosphorus compounds and ammonia, among others, will be evidence concerning actual loading of manure material. Ecological impact of the pollution will be evaluated by the developed immunological assays performed on wild-caught fathead minnows. Statistical comparisons will be made between the calculated and measured evidence for the pollutant to determine the accuracy and reliability of the immunological approach in actual practice. As a further check on the procedure, Index of Biotic Integrity (IBI) values will be calculated based on wild fish community parameters. The IBI is a commonly used bioindicator of stream environmental quality. It serves as a summary measure of biotic community response to pollution and other forms of habitat degradation. It is being used routinely for long-term environmental monitoring programs in Iowa and other Midwestern states. This design, therefore, will allow comparisons between this established coarse-scale environmental indicator and the experimental fine-scale immunological indicator.

Principal Findings and Significance for Period from 03/2005 to 02/2006

The effects of acute and chronic stress on the fathead minnow neutrophil function *in vivo* were determined. Fathead minnows exposed to acute and chronic stress conditions had significantly reduced degranulation, demonstrating that the degranulation assay can be used to measure both acute and chronic stress effects on neutrophil function in this species. This step demonstrated the capability of the assay to measure reduction in neutrophil function. Scientific background and baseline data for use of fathead minnow neutrophil function in future research and aquatic ecosystem evaluation was provided. Fathead minnows are shown to be a useful model to investigate neutrophil degranulation in fish exposed to different environmental conditions. Experiments characterizing effects of sublethal manure exposure on innate immune funciton in fathead minnows are underway.

The spatial distribution of manure on fields has not been investigated in the context of aquatic ecosystem health. The inter-relatedness of disposal facilities, management plans, field location, rate of manure application, and stream network were investigated using spatial analysis over different data sets. The GIS model was developed to identify stream sections that have high likelihood to encounter potential hazards for stream biota.

Fig. 1. Spatial distribution of A: Animal waste control facilities with operating permits in Iowa (red), Manure Management Plans for Animal Feeding (green); and B: Fish kills (red), in The State of Iowa.

A B

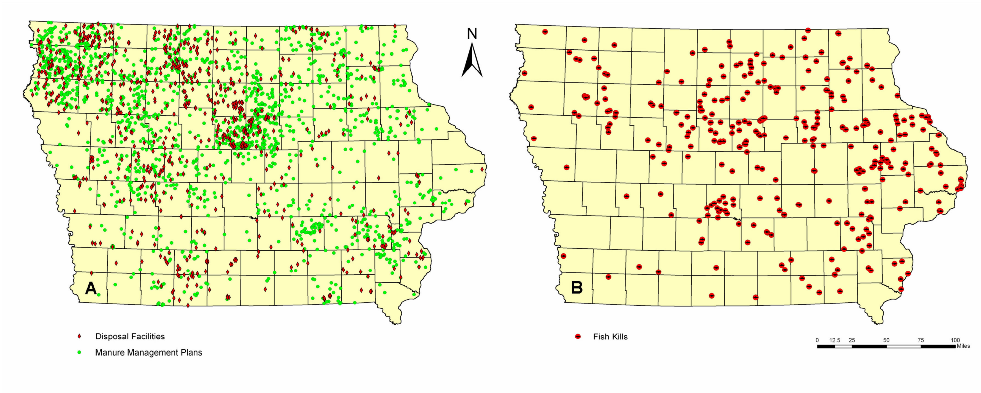


Fig. 2. Digitized fields used for manure disposal. Green: Agriculture Disposal facilities; Red: Manure Management Plans; Yellow: field boundaries. (Hamilton County)

Fig. 3. Predicted flow of manure after field application. Quantity of manure is expressed as intensity of color (dark red = more). Gray areas indicate potential critical points in streams (blue). (Hamilton County)

Figure 2.

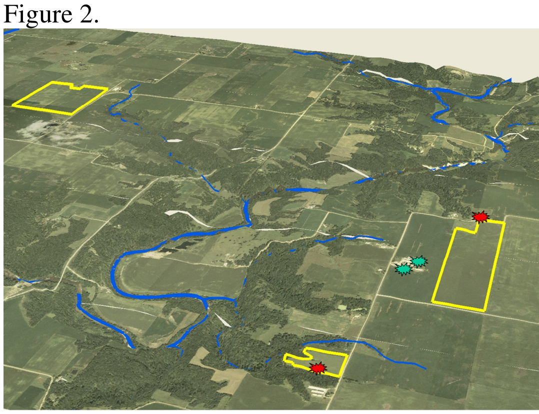


Figure 3.

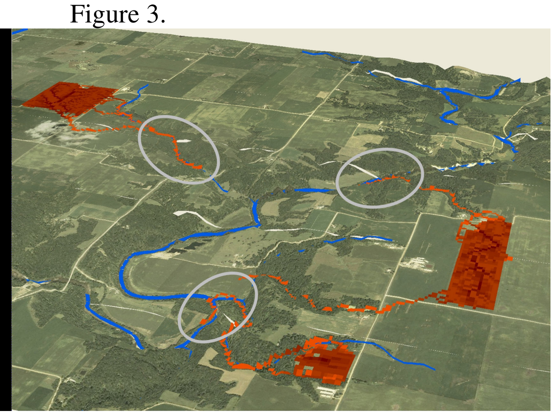
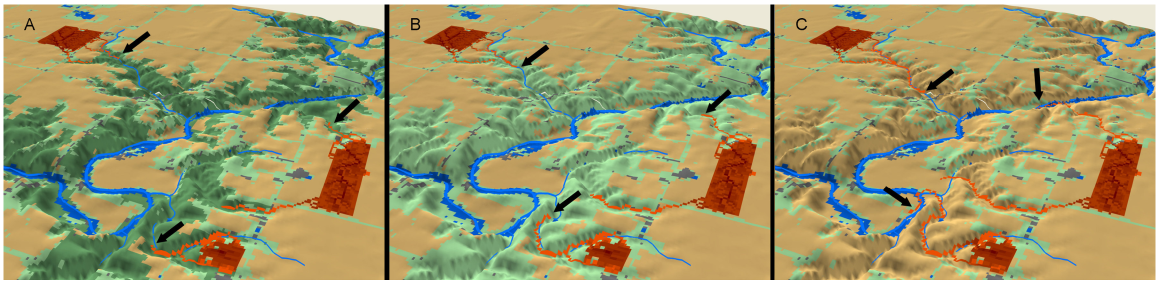


Fig. 4. Three possible scenarios for manure flow based on differences in land cover. Dark green: forest (A); Light green: grassland (B); Tan: row crops (C). Quantity of manure is expressed as the intensity of color (dark red = more). Black arrows indicate the distance of manure flow in streams (blue). (Hamilton County)



Using spatial distribution of AGDs, MMPs, and fish kills, the areas with potential environmental concern were identified in northeast and northwest Iowa (Fig 1.). Based on data accessibility, Hamilton County was selected for detailed study. Conversion of vector to raster datasets, calculations of manure flow path, and setting the threshold level were performed in ArcGIS for different field situation scenarios. The manure flow from three different fields was calculated and critical points in stream were identified (Figs 2-4.). The critical points were determined based on the presence of the manure in stream sections above the threshold level. The difference in land cover has a significant effect on the flow path length and quantity of manure reaching streams ($P<0.01$). The flow of manure through forested areas is the shortest and through row crop is the longest (Fig 4.).

Based on preliminary analysis of available datasets, we suggest that the TauDEM flow path analysis and the developed geospatial model have potential to be used as a tool to determine stream sections with increased probability of critical impairment due to manure contamination. A fully developed tool could assist in the selection of stream sampling points relevant to ecosystem health assessment and reduce the cost of field surveys. Changes in land cover can be modeled to help management decisions, improve spatial distribution of animal facilities, and predict possible outcomes of restoration efforts.

In summary, work performed from 03/2005 to 02/2006 clearly demonstrated that:

1. Neutrophil functional assays show potential for use in studying effects of immunomodulatory compounds, as well as effects of environmental stress on fish physiology, providing us with new tools to be used in the assessment of aquatic ecosystem health.
2. The TauDEM flow path analysis and the developed geospatial model have potential to be used as a tool to determine stream sections with increased probability of critical impairment due to manure contamination. A fully developed tool will assist in the selection of stream sampling points relevant to ecosystem health assessment and reduce the cost of field surveys.
3. Changes in land cover can be modeled to help management decisions, improve spatial distribution of animal facilities, and predict possible outcomes of restoration efforts.

Fate of Veterinary Antibiotics in Manure Lagoons

Basic Information

Title:	Fate of Veterinary Antibiotics in Manure Lagoons
Project Number:	2003IA37B
Start Date:	3/1/2003
End Date:	2/28/2006
Funding Source:	104B
Congressional District:	Iowa 3rd
Research Category:	Water Quality
Focus Category:	Toxic Substances, Water Quality, Treatment
Descriptors:	antibiotics, biodegradation, adsorption, fate and transport
Principal Investigators:	Say Kee Ong, Thomas B. Moorman

Publication

1. Kolz, A.C., S.K. Ong, and T.B. Moorman. 2005. Sorption of Tylosin onto Swine Manure, Chemosphere, 60(2):284-289.
2. Kolz, A.C., T.B. Moorman, S.K. Ong, K.D. Scoggin, and E.A. Douglas. 2005. Degradation and metabolite production of tylosin in anaerobic and aerobic swine manure slurries, Water Environment Research, 77(1):49-56.
3. Ong, S.K., W. Lertpaitoonpan, T. Moorman, 2006. Sorption of sulfamethazine to soils: Effect of organic carbon and pH, International Conference on Waste Management for a Sustainable Future, Jan. 10-12, Bangkok, Thailand.
4. Moorman, T.B., W. Lertpaitoonpan, K.L. Henderson, S.K. Ong, M.D. Tomer, and J.R. Coats, 2006. Fate and behavior of bacteria and veterinary antibiotics resulting from swine manure applications in Iowa, 231st ACS National Meeting, March 26-30, Atlanta, GA.
5. Kolz, A.C., T.B. Moorman, S.K. Ong, K.D. Scoggin, 2004. Degradation of Tylosin and metabolite production in anaerobic and aerobic swine manure slurries, WEFTEC, Oct. 4-7, New Orleans, LA. (Awarded 2nd Place for poster.)

Fate of Veterinary Antibiotics in Manure Lagoons

Say Kee Ong, Tom Moorman

Problem and Research Objectives

Antibiotic residues and increased numbers of antibiotic-resistant bacteria have been reported near confined animal feeding operations (CAFOs) and in agricultural watersheds. A major route for entry of veterinary pharmaceuticals into watersheds is through land application of animal biosolids and spills of animal waste at facilities using these drugs. Swine CAFOs often use antibiotics for therapeutic or growth-promoting purposes. Manure generated at CAFOs, containing excreted residues, is commonly stored in earthen lagoons for several months before land application. Incomplete degradation of pharmaceuticals, in vivo and during manure storage before biosolids are land-applied, could be a contributing form to the presence of these drugs in waterways.

The fate of these chemicals is of environmental importance as it has been shown that highly resistant pathogens may develop within the manure management facilities and that these chemicals may interfere with the endocrine system of various aquatic species. Currently, research on the fate of these compounds in the manure management system and in the environment is very limited. The objectives of this study are to investigate the fate of two common antibiotics, tylosin and sulfamethazine, used in the swine industry. The focus will be on the sorption and degradation of these antibiotics in manure lagoons under anaerobic and aerobic conditions.

Methodology

The proposed research consists of analytical methods development, batch sorption studies and batch degradation studies. The antibiotics to be tested are tylosin and sulfamethazine, two major antibiotics used in the swine industry. Manure will be obtained from various manure lagoons and characterized for pH, total organic carbon, total dissolved solids, and ammonia.

A key aspect of studying antibiotics in the environment is the ability to analyze the antibiotics in various media and in low concentrations. Different solvents for extraction of antibiotics from both liquid and sludge from waste manure using suitable solvents were tested. The antibiotics were analyzed using liquid chromatograph and liquid chromatograph-mass spectroscopy (LC-MS).

Batch sorption experiments were conducted according to the American Society of Testing and Materials E1195-01 (ASTM, 2002). Sodium azide was added to each vial to inhibit microbial degradation. Anaerobic degradation studies were conducted using a series of 120 mL serum bottles containing sludge from manure lagoons. The serum bottles were spiked with a given amount of antibiotic and the vials were purged with nitrogen to ensure dissolved oxygen is removed. At different times, vials were sacrificed and the concentrations of the antibiotics in both liquid and solid phases analyzed. The parent compound remaining and metabolites, if any, were determined using LC-MS. Aerobic degradation experiments will be similarly conducted.

Principal Findings and Significance

Tylosin disappearance followed a biphasic pattern, where rapid initial loss was followed by a slow removal phase. The 90% disappearance times for tylosin, relomycin (tylosin D), and desmycosin (tylosin B) in anaerobically incubated slurries were 30 to 130 hours. Aerating the slurries reduced the 90% disappearance times to between 12 and 26 hours. Biodegradation and abiotic degradation occurred, but strong sorption to slurry solids was probably the primary mechanism of tylosin disappearance. Dihydrodesmycosin and an unknown degradate with molecular mass of m/z 934.5 were detected. Residual tylosin remained in slurry after eight months of incubation, indicating that degradation in lagoons is incomplete and that residues will enter agricultural fields.

Sorption of tylosin was conducted on manure solids (<2 mm) and colloidal materials (<1.2 lm) collected from open (OL) and covered (CL) anaerobic swine manure lagoons. The aqueous concentration of tylosin in the sorption studies bracket the levels expected in lagoons, between 1 mgL^{-1} and 30 mgL^{-1}. Sorption isotherms were found to be slightly non-linear for 2 mm solids, with Freundlich distribution coefficients (K_f) of 39.4 with n = 1.32 for CL slurry and 99.5 with n = 1.02 for OL. These values are comparable to those reported for loam soils, but higher than those reported for sandy or clay soils and lower than those reported for fresh manure. Normalization of K_d to the organic carbon content of the solids gave K_{oc} values of 570 L/kg^{-1} and 818 L/kg^{-1}, for CL and OL solids, respectively. The K_d and K_f values were not significantly different between colloids and 2 mm solids in OL slurry, but were significantly different in CL due to the non-linearity of the colloid isotherm. Based on the K_d values obtained and comparing the K_d values of other antibiotics, tylosin is strongly sorbed to manure, and would be more mobile than tetracyclines, but less mobile than sulfonamides, olaquindox, and chloramphenicol. However, tylosin mobility may be facilitated through transport with colloidal manure materials.

Batch sorption studies of sulfamethazine (SMZ) onto five different soils were conducted using soil: liquid ratio equal to 1:3 (w:v). The organic carbon (OC) content of the soils ranged from 0.1 % to 3.8 %, and the pH of the soil slurry was adjusted from 5.5 to 9. SMZ concentrations used ranged from 1.1 to 22.2 mg/L (3.3 - 66.6 µg SMZ/g soil). Sorption isotherms were best fitted by a linear sorption model. Adsorption of SMZ was impacted by both organic carbon and by soil solution pH. The linear partition coefficient (K_d) was found to be lower for high pH values. For example, K_d values for soil with 0.1% OC at pH 5.5 was 0.59 ± 0.18 L/kg and was 0.23 ± 0.06 L/kg at pH 9. At pH 9, the K_d value for a soil with 3.8 % OC was 1.11 ± 0.05 L/kg but was 0.23 ± 0.06 for a soil with 0.1% OC. It appears that the form of SMZ plays an important role in sorption to soils. At pH greater than 7.4 (pKa value of SMZ), SMZ is in its ionized form and was sorbed less to soils than at neutral pH or lower. This would imply that at pH 9, SMZ may be more mobile in soil than for soils with neutral pH or less.

Information Transfer Program

Student Support

Student Support					
Category	**Section 104 Base Grant**	**Section 104 NCGP Award**	**NIWR-USGS Internship**	**Supplemental Awards**	**Total**
Undergraduate	1	8	0	0	9
Masters	3	2	0	0	5
Ph.D.	4	5	0	0	9
Post-Doc.	0	0	0	0	0
Total	8	15	0	0	23

Notable Awards and Achievements

2004--Anne Cleary International Research Fellowship, The University of Iowa Fellowship awarded to Ceyda Polatel for dissertation enhancement research at the University of Karlsruhe, Germany. The research focused on the implementation of Large Eddy Simulation for the present study (August-November 2004).

2005--The University of Iowa Engineering Research Open House Best Poster Award for the poster "The flow structure over a two-dimensional sand dune" (April 7, 2005).

Ashley Mordasky--Best Poster, Iowa State Groundwater Association Annual Meeting, 2005.

Zhuo Chen--Selection of paper for presentation at the Universities Forum, National AWWA Meeting, San Francisco, CA, June 11-15, 2005.

Dusan Palic was recipient of Graduate Research Excellence Award, Iowa State University, Ames, IA, 2005.

Dawn Herolt and Dusan Palic were recipients of 2005 Second Best Poster Award, Student American Veterinary Medical Association Symposium, March 10-12, 2005, College Station, TX.

Angela Kolz’s thesis was awarded the 2005 Montgomery-Watson-Harza/Association of Environmental Engineering and Science Professors (AEESP) Master’s Thesis Award (First Place). This is an international/national competition among all environmental engineering graduates.

Silvia Secchi et al. received a second place award for their poster presentation at the 2005 AAEA Annual Meeting in Providence, RI, "Linking the Economic Costs and Water Quality Benefits of Conservation in Agricultural Lands: An Iowa Assessment."

Silvia Secchi et al. presented their paper "The deisgnation of co-benefits and its implication for policy: water quality vs. carbon sequestration in agricultural soils" at the International Policy Forum on Greenhouse Gas Management, April 28-29, 2005, Victoria, British Columbia, Canada.

Hongli Feng et al. will present their paper "Allocating Nutrient Load Reduction across a Watershed: Implications of Different Principles" at the 2006 Annual Meeting of American Agricultural Economics Association in Long Beach, CA, July 23-26, 2006.

Publications from Prior Projects