CHARACTERIZING STORM WATER RUNOFF FROM NATURAL GAS WELL SITES

IN DENTON COUNTY, TEXAS

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Dissertation Prepared for the Degree of

DOCTOR OF PHILOSOPHY

UNIVERSITY OF NORTH TEXAS

May 2008

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Wachal, David J. Characterizing Storm Water Runoff from Natural Gas Well Sites in Denton County, Texas. Doctor of Philosophy (Environmental Science), May 2008, 123 pp., 22 tables, 17 figures, 193 references.

In order to better understand runoff characteristics from natural gas well sites in north central Texas, the City of Denton, with assistance through an EPA funded 104b3 Water Quality Cooperative Agreement, monitored storm water runoff from local natural gas well sites.

Storm water runoff was found to contain high concentrations of total suspended solids (TSS). Observed TSS concentrations resulted in sediment loading rates that are similar to those observed from typical construction activities. Petroleum hydrocarbons, in contrast, were rarely detected in runoff samples. Heavy metals were detected in concentrations similar to those observed in typical urban runoff. However, the concentrations observed at the gas well sites were higher than those measured at nearby reference sites.

Storm water runoff data collected from these sites were used to evaluate the effectiveness of the water erosion prediction project (WEPP) model for predicting runoff and sediment from these sites. Runoff and sediment predictions were adequate; however, rainfall simulation experiments were used to further characterize the portion of the site where drilling and extraction operations are performed, referred to as the “pad site.” These experiments were used to develop specific pad site erosion parameters for the WEPP model.

Finally, version 2 of the revised universal soil loss equation (RUSLE 2.0) was used to evaluate the efficiency of best management practices (BMPs) for natural gas well sites. BMP efficiency ratings, which ranged from 52 to 93%, were also evaluated in the context of site management goals and implementation cost, demonstrating a practical approach for managing soil loss and understanding the importance of selecting appropriate site-specific BMPs.

ACKNOWLEDGEMENTS

This research was funded by the City of Denton and the United States Environmental Protection Agency (USEPA) through a 104(b)(3) Water Quality Cooperative Agreement. The City of Denton's Watershed Protection Program employees helped considerably with storm water data collection and analyses.

I would like to thank my committee members Drs. Hudak, Banks, Harmel, Acevedo, Dickson, and La Point for their guidance and support. I especially thank Dr. Banks for his time, patience, and involvement throughout the project. I would also like to extend my appreciation to Dr. Daniel Yoder for his assistance with RUSLE 2.0 and Dr. William Elliot for his assistance with the rainfall simulation experiments and WEPP erosion parameterization.

Lastly, but most importantly, I warmly thank my wife, Amy, for her constant source of encouragement, love, and support and my daughter, Rhyan, for providing the inspiration needed to complete this work.

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CHAPTER 1

INTRODUCTION

Natural gas development results in a substantial amount of land disturbance nationwide, with almost 30,000 new well sites constructed annually. Denton County, Texas lies above the Barnett Shale, one of the largest natural gas formations in North America. Barnett Shale natural gas exploration and production has increased dramatically in recent years due to rising gas prices, a better understanding of local geology, novel fracturing technologies, and the concurrent growth of an extensive infrastructure of gathering and transmission pipelines. Natural gas exploration and production (NGE&P) typically requires the construction of a well pad site, access roads, and pipelines, all of which have the potential for accelerating soil erosion due to modifications of land cover, increased slopes, and flow concentration. Also, there is a potential for storm water runoff to be polluted with petroleum hydrocarbons and metals from the use of refined petroleum products and the corrosion, wearing, and shearing of machinery and equipment for drilling and production operations. While typical construction activities are regulated by federal national pollutant discharge elimination system (NPDES) permit requirements, the Environmental Protection Agency (EPA) recently exempted oil and gas construction activities from NPDES rules, leaving state and local agencies responsible for managing and regulating storm water runoff from these activities.

Specific objectives of this research were to: (1) characterize the types and magnitude of pollutants in storm water runoff from natural gas well sites, (2) evaluate the effectiveness of the water erosion prediction project (WEPP) for modeling runoff and sediment from gas well sites, (3) use simulated rainfall on research plots to characterize runoff specifically from the gas well drilling pad and develop erosion parameters for the WEPP model, and (4) demonstrate how best management practices could reduce sediment pollution from gas well sites using version 2 of the revised universal soil loss equation (RUSLE 2.0). These research objectives were documented in a series of four separate manuscripts.

The first manuscript (Chapter 2) characterizes storm water samples collected from three gas well sites and two reference sites. Storm water samples were analyzed for conventional water quality parameters, sediments, metals, and petroleum hydrocarbons. These data were: evaluated for differences between gas well sites and references sites using summary statistics and nonparametric statistics; discussed in terms of national drinking water standards, ambient aquatic life criteria, and storm water runoff from local watersheds; and compared to previous research related to runoff characterization from typical construction sites. This manuscript was intended to provide guidance in making federal, state, or local storm water management decisions pertaining to natural gas exploration and development. The manuscript has been submitted for publication in *Environmental Monitoring and Assessment*.

The second manuscript (Chapter 3) describes a modeling approach for estimating sediment yield from gas well sites. The objective of this research was to evaluate WEPP runoff and sediment yield predictions relative to measured data from two natural gas well sites. This component of the research was conducted as the monitoring program was ongoing; therefore, data used to evaluate the model was from storm events that occurred in 2006 between the months of February and November. WEPP performance was evaluated with the Nash-Sutcliffe efficiency (NSE), root mean square error (RMSE)-observation standard deviation ratio (RSR), and percent bias (PBIAS), as well as modified versions of NSE and RSR that consider uncertainty in measured validation data. The results demonstrated that WEPP can effectively model runoff and sediment yields from natural gas well sites, thus making it a useful tool for evaluating potential sediment impacts and management alternatives to minimize sediment yields from natural gas well sites. This manuscript has been submitted for publication in *Transaction of the American Society of Agricultural and Biological Engineers* (ASABE).

The third manuscript (Chapter 4) develops soil erodibility parameters for the WEPP model using data derived from rainfall simulation experiments conducted on research plots at two natural gas well sites and one reference sites. The objectives of this study were to compare runoff and erosion data among plots and between sites and develop interrill erodibility parameters for the WEPP model. Parameters derived from the rainfall simulation data used in the model provided very good modeling results and thus can be used as guideline in future gas well modeling studies. The calibrated model was also used to predict average annual sediment yields specifically from natural gas well pad sites, which were compared to results discussed in Chapters 1 and 2. This manuscript has not yet been submitted for publication.

The final manuscript (Chapter 5) provides a practical modeling approach for evaluating six erosion and sediment control best management practices (BMPs) for multiple combinations of different land surface conditions (soil erodibility and slope) commonly found at gas well sites in the area. The objectives of this research were to evaluate the relative effectiveness of six BMP alternatives for natural gas well sites using RUSLE 2.0 and demonstrate a practical approach for quantitatively evaluating BMP alternatives according to site-specific soil erodibility and slope conditions, site management goals, and BMP implementation costs. This research illustrates the importance of evaluating site-specific surface conditions when evaluating erosion and sediment control best management practices. This manuscript has been accepted for publication in *Environmental Geology*; the original publication is available at www.springerlink.com

CHAPTER 2

CHARACTERIZING STORM WATER RUNOFF FROM NATURAL GAS WELL SITES IN NORTH CENTRAL TEXAS

Introduction

In 2006, nearly 30,000 natural gas wells were drilled nationwide (API, 2007). Natural gas well development has the potential to negatively affect water quality in receiving systems due to possible loadings of sediment, petroleum hydrocarbons, and heavy metals in storm water runoff. Natural gas exploration and production requires the construction of a well site, access roads, and collection and transmission pipelines. These land disturbing construction activities have the potential to accelerate erosion and soil loss due to land cover modifications, increased slope, and flow concentration. Primary sources of petroleum hydrocarbons and heavy metals at these sites are from the use of refined petroleum products (Carls et al., 1995) and corrosion, wearing, and shearing of machinery and equipment for drilling and production operations. High levels of petroleum hydrocarbons were found in soil samples at four natural gas well sites in South Padre Island, Texas a decade after the wells were completed (Carls et al., 1995). Other possible sources of petroleum hydrocarbons and metals include additives in drilling fluids and local emissions from on-site machinery and equipment.

While storm water runoff from natural gas well sites has not been previously studied, runoff from land disturbing construction activities is fairly well understood. Erosion rates from construction activities are 2 to 40,000 times greater than pre-construction conditions (Wolman and Schick, 1967) and 10 to 100 times greater than cropland (Goldman, 1986). Estimates of annual sediment delivery into US surface waters from construction activities has ranged from 73 million tonnes (USDOI, 1970) to 4.5 billion tonnes (Willett, 1980). Construction activities are by far the greatest source of sediment yield in developing areas (USEPA, 2002) contributing from 17 to 1121 tonnes per hectare per year (t ha^{-1} $yr^{-1)}$ (USEPA, 2002).

Storm water runoff containing sediment, petroleum hydrocarbons, and metals can negatively affect the aquatic environment. Sediment is the leading source of water quality impairment in US rivers and streams and is the third most ubiquitous source of impairment in US lakes and reservoirs (USEPA, 2000). Suspended sediment can reduce in-stream photosynthesis and transport associated pollutants such as trace elements, toxic organics, and nutrients (Tessier, 1992; Davies-Colley and Smith, 2001). Nutrients in eroded soils can contribute to algal blooms and lake eutrophication (Goldman, 1986). Suspended sediment can also be detrimental to fish populations (Chiassen, 1993; Poff and Allen, 1995; Newcombe and Jensen, 1996; Vondracek, 2003). Highly turbid water can dramatically increase water treatment costs and diminish direct and indirect recreational experiences (AWWA, 1990; Clark et al., 1995). Once deposited, sediment can smother benthic communities, reduce fish egg survival rates, reduce channel capacity, exacerbate downstream bank erosion and flooding, and reduce storage in reservoirs (USEPA, 1993; Schueler, 1997; Henley et al., 2000). Petroleum hydrocarbons have a tendency to bioaccumulate in sediment and soils, persist in the environment for decades, and remain potentially available to sediment dwelling organisms. Some petroleum hydrocarbons are known to adversely impact benthic organisms and can be toxic to aquatic life at low concentrations (Whipple and Hunter, 1979; Stenstrom, 1984). When metals are released into the environment in concentrations higher than natural conditions they can be highly toxic and cause major disruptions of aquatic ecosystems depending on the nature of receiving water.

While typical construction activities are regulated by the federal national pollutant discharge elimination system (NPDES) program – which requires erosion and sediment control best management practices (BMPs), storm water pollution prevention plans, and increased monitoring and site inspections – oil and gas field operations and construction activities are exempt from federal NPDES permitting requirements (USEPA, 2006). This exemption amplifies the need to understand storm water runoff characteristics from such sites. The purpose of this study was to characterize storm water runoff from natural gas well sites in North Central Texas. The primary constituent of concern was sediment, but conventional water quality parameters, metals, and petroleum hydrocarbons were also evaluated due to potential on-site sources. Data collected from gas well sites were compared to: (1) data collected from nearby reference sites to evaluate differences between pre- and post-development site conditions, (2) federal and state water quality standards, and (3) literature values reported in previous research. Results of this study were intended to provide information that can be used to make federal, state, or local storm water management decisions pertaining to natural gas exploration and production.

Methods

Study Area and Site Description

Three natural gas well sites (Site 2, Site 3, and Site 4) and two undisturbed references sites (Site 2R and Site 3R) selected for the study were located in the southwest portion of Denton County, Texas (Figure 2.1). This area of the county lies above Barnett Shale, which is an organically rich geologic formation that may contain the largest onshore natural gas formation in the US (Shirley, 2002). Study sites were located in the Grand Prairie physiographic region, consisting of gently sloping grasslands with scattered shrubs, and trees primarily along creek bottoms (Griffith et al., 2004). Uppermost bedrock beneath the region consists of Lower Cretaceous limestones with interbedded marl and clay (McGowen et al., 1991). Soil underneath Site 2 and Site 3 is classified as Medlin stony clay (fine, montmorillonitic, thermic, vertisols) on slopes of 5 to 12%. Soil underneath Site 4 is classified as Sanger clay (fine, montmorillonitic, thermic, vertisols) on slopes of 3 to 5%. Both soils are moderately alkaline and have very low permeability, moderate/high runoff potentials, and severe erosion potentials (USDA SCS, 1980). Average annual rainfall is approximately 99 cm, the majority of which normally occurs during the spring months of April and May and the fall months of September and October (USDA SCS, 1980). Thunderstorms are common in the spring and can be intense and highly erosive. Runoff from these sites eventually enters Hickory Creek and flows into Lake Lewisville, which is used for water supply and recreation by a large population of North Texas residents.

All three gas well sites were constructed on approximately 5% slopes. At each site, the original slope was leveled for the gas well pad surface, resulting in a site profile consisting of a cut slope, pad surface, and fill slope that was approximately 100 m in length (Figure 2.2). While the sites are similarly constructed, the geometry of the cut slopes and pad surfaces vary from site to site. The pad surface is relatively flat and is used for drilling activities, equipment storage, and well maintenance. The term “cut slope” generally refers to the face of an excavated bank required to lower the ground to a desired profile. In contrast, a “fill slope” refers to a surface created by filling an area with soil. All slopes were compacted with a mechanical roller, and an all-weather surface of Grade 1 Flex Base (crushed limestone) was applied to the pad surface. Flex Base is a gravel aggregate commonly used for temporary roads, base material underneath asphalt and concrete paving, and construction pad caps. The Flex Base surface application was approximately 0.3 m in depth and covered an area approximately 0.5 ha. The soil on the cut and fill slopes covered an area of approximately 0.5 ha and was left exposed after compaction. Additional area was disturbed around each site due to general construction activities.

The two reference sites were located in close proximity to the gas well sites on relatively treeless undulating tallgrass prairie dominated by little bluestem (Schizachyrium scoparium). The reference sites were left undisturbed and represent pre-development site conditions. Site characteristics for gas well and reference sites are described in Table 2.1.

Storm Water Monitoring

Flow-interval (1.0 mm of volumetric runoff depth) storm water samples were collected with ISCO 6712 (ISCO, Inc., Lincoln, Nebraska) automated samplers. This method is recommended for small watershed sampling according to Harmel et al. (2006). ISCO samplers were programmed to take up to 18 discrete 1000 ml samples and then, if the runoff event continued, 6 composite samples of 250 ml each (Wachal et al., 2005). This program design extends the sampling period for large storm events. Samples were taken at a single intake point near the bottom of a partially contracted sharp-crested 90^{o} V-notch weir (USDOI, 1997) located at the edge of each gas well pad surface. An impermeable barrier was installed along the down slope portion of the pad surface to direct flow through the weir. The toe of the barrier was set in a 15.2 cm deep trench and backfilled to prevent bypass of runoff under the barrier. Wood posts set approximately 1.2 m apart supported the barrier. The placement of the weir and barrier captures runoff from the cut slope and pad surface but does not capture runoff from the fill slope (Figure 2.1). Flow volume was monitored with ISCO 4250 velocity flow meters (ISCO, Inc., Lincoln, Nebraska) placed 1 m upstream from the outfall of each weir. Rainfall at each site was monitored with a tipping bucket style ISCO 674 Rain Gauge (ISCO, Inc., Lincoln, Nebraska). Both flow and rainfall data were logged at 5-min intervals.

A total of 40 runoff events were sampled at the three gas well sites (Site 2, n=17; Site 3, n=12; Site 4, n=11) and 10 runoff events were sampled at the two reference sites (Site 2R, n=5; Site 3R, n=5) (Appendix A). Additional events were sampled but were not included in the analyses due to incomplete sampling or lack of accurate flow information (55 events), or a small number of samples (≤ 2) that did not entirely represent the storm hydrograph (e.g., 1st sample taken at the beginning of the event, 2nd sample taken at the end of the event, with no samples taken near the peak) (23 events).

Analytical Procedures

Water quality parameters analyzed under the monitoring program, along with the analytical methods and detection limits, are summarized in Table 2.2. Parameters routinely analyzed throughout the study period include conventional parameters (alkalinity, calcium, chlorides, conductivity, hardness, pH, total dissolved solids (TDS)); metals (As, Cd, Cr, Cu, Fe, Pb, Mn, Ni, Zn); petroleum hydrocarbons (total petroleum hydrocarbons (TPH), benzene-toluene-ethylbenzene-xylene (BTEX)); and components of sediment (turbidity, total suspended solids (TSS)). Standard field and laboratory quality assurance/quality control (QA/QC) procedures were followed according to procedures specified in the Quality Assurance Protection Plan for prepared for Water Quality Cooperative Agreement CP-83207101-1 (City of Denton, 2005). Principal QA/QC procedures performed on samples include field, lab, and spike duplicates and field and equipment blanks.

Storm event mean concentrations (EMCs) were calculated for all water quality parameters according to the following equation:

EMC = $\left[\sum_{i=1}^{n} c_i / n\right]$

where *c* = the sample concentration and *n* = the number of discrete samples. Since storm water samples were taken on consistent flow intervals, the arithmetic average of water quality parameter concentrations represents the event mean concentration (EMC). Following acceptable protocols, concentrations below the detection limit were replaced with one half the detection limit value (USEPA, 1996).

Data Analysis

Initial data analyses, which included descriptive statistics and the Shapiro-Wilks test for normality, indicated the data were non-normally distributed. Since log-transformation of the data did not result in a normal distribution for all parameters, non-parametric statistics were used. Parameters containing more than 50% of storm event EMCs below detection limit (As, Cd, Cr, Cu, Pb) were not included in the statistical analyses. A nonparametric GLM ANOVA approach was used to test for differences in parameter EMCs among all sites. The GLM is a type of ANOVA that is more appropriate for unbalanced data (unequal number of observations for each classification factor; see SAS, 2006). Statistically significant GLM analyses ($\alpha = 0.05$)

were followed by Student Newman Kuels (SNK) multiple comparison tests ($\alpha = 0.05$) to test for difference between sites. Results of the multiple comparison tests were used to verify if analyzed constituents observed at each site were statistically similar within each site type (gas well sites or reference sites).

Once verified that sites within each site type (gas well and reference) were not statistically different from each other, data were combined by site type. Grouping the data by site type incorporates all the site variability within each site type and allows for comparison between the group representing all gas well sites and the group representing all reference sites. Differences between site types were assessed using a Wilcoxon rank-sum test ($\alpha = 0.05$). For moderate sample sizes, the Wilcoxon test is considered almost as powerful as its parametric equivalent, the t-test (Cody and Smith, 1997).

In addition to comparisons of discrete concentrations and EMCs, estimates of annual TSS loads in this study were also compared to data collected in previous construction site studies to provide a framework for storm water management decisions. This additional comparison was needed because gas well site surface conditions are similar to typical residential and commercial construction sites, which creates debate on similarities/differences between the types of sites. Annual loadings were estimated assuming that the average EMC determined for each site over the course of the study generally represents annual average runoff conditions. Although there are many factors that influence erosion at disturbed sites, which vary both spatially and temporally, average conditions can provide a useful approximation. Furthermore, because the sampling location provided site characterization of only the cut slope and pad surface, and not the fill slope, which is the most erodible portion of the sites, this approximation representing the entire site is most likely conservative. Average annual sediment yields for each site were calculated according to the product of the average TSS EMC and the volume of runoff estimated from the average annual rainfall (99 cm) using the curve number (CN) method (USDA SCS, 1986). A CN of 93 for gas well sites was previously determined (Wachal and Banks, 2007); this value is similar to the CN value of 94 (Hydrologic Group D) for “newly graded developing areas” (Maidment, 1993).

Results and Discussion

Conventional Parameters

SNK multiple comparison test results are shown in Table 2.3. Results of the Wilcoxon rank-sum test, along with differences between the median EMC concentrations of each site type expressed as the ratio of reference sites to gas well sites, are shown in Table 2.4. Table 2.5 summarizes the range, mean, standard deviation, and median of storm event EMCs for conventional water quality parameters, TSS, and turbidity by site. Individual storm EMCs are shown in Appendix A. Surface runoff of gas well sites appears to be greatly influenced by the limestone aggregated used to construct the gas well pad surface.

Limestone contains large amounts of calcium carbonate ($CaCO_3$), which can influence TDS, conductivity, pH, alkalinity, hardness, and calcium. Total dissolved solid EMCs tended to be higher at gas well sites compared to reference sites, but differences were not significant ($p=0.0561$). Calcium and chlorides, two common constituents of TDS, were statistically significantly greater at gas well sites and were 8 and 1.7 times greater at gas well sites compared to reference sites, respectively. The presence of these inorganic dissolved solids at gas well sites also tends to increase the specific conductivity of surface runoff from gas well sites as conductivity EMCs were significantly greater than reference site EMCs ($p=0.0483$). EMCs for alkalinity ($p<0.0001$), hardness ($p<0.0001$), and pH ($p<0.0001$) were also significantly greater at gas well sites. Calcium carbonate can be a major cause of hard water (hardness), and alkalinity can be influenced by the dissolution of carbonate rocks. Since alkalinity is a measure of the capacity of water to neutralize acids, alkalinity is related to the pH of a solution. The median pH EMC at gas well sites was 8.64 compared to a median EMC of 7.47 at reference sites.

Metals and Petroleum Hydrocarbons

EMC summaries for each analyzed metal constituent are provided in Table 2.6. Storm event EMCs for metals are shown in Appendix B. Generally mean EMCs were greater than median EMCs. EMCs on the low end of the range were below detection limits for As, Cd, Cr, Cu, Pb, and Zn. Arsenic did not have any EMCs above the detection limit at any of the sites. Standard deviations indicated variability was high for most metals at most sites. For discussion purposes, Table 2.7 compares metal concentrations at gas well and reference sites to drinking water standards (USEPA, 2007), ambient acute aquatic life criteria thresholds (USEPA, 2007), and concentrations in storm water runoff from local watersheds reported by Hudak and Banks (2006).

For cadmium, 14% of the EMCs were above the detection limit at gas well sites and 25% were above the detection limit at reference sites. None of the gas well cadmium EMCs were above the drinking water maximum contaminant level (MCL), but one of the reference site's

EMC’s was. Gas well sites and reference sites each had one cadmium EMC above the aquatic life criterion. EMCs for copper were above the detection limit 50% of the time at gas well sites, all of which were also above the aquatic life criterion. Only one of these EMCs was above the drinking water standard. At reference sites, one EMC was above the detection limit for copper, but it was not above the drinking water standard or aquatic life criterion. Chromium EMCs at gas well sites were above the detection limit 35% of the time, none of which were above the MCL or aquatic life criterion. None of the chromium EMCs were above the detection limit at reference sites. Overall, there was a greater number of cadmium, chromium, and copper EMCs above the detection limit for gas well sites compared to reference sites and EMCs tended to be higher at gas well sites, indicating that gas well site activities may increase the incidence of these metals. Potential sources of cadmium at gas well sites are similar to those in urban environments, which include fuel combustion, engine wear, automobile tires, brake pads, and galvanized building materials (Makepeace et al. 1995; Davis et al., 2001; Van Metre and Mahler, 2003). In addition to these sources, paint is also a potential source of chromium at gas well sites. Industrial and mechanical processes associated with drilling may also contribute to copper in gas well site runoff.

EMCs for zinc were above the detection limit 67% of the time at gas well sites compared to only 38% of the time at reference sites. None of the EMCs were above the drinking water standard, but both gas well sites and reference sites each had two EMCs above the aquatic life criterion. Concentrations of zinc at gas well sites could potentially be due to on-site sources such as tires, galvanized steel, and wearing of metal alloys used in engine parts. Zinc concentrations at reference sits could be influenced by deposition of zinc from nearby gas well sites and/or from zinc occurring naturally in the environment. All nickel EMCs were above the detection limit at gas well sites and reference sites; however, none of the nickel concentrations were above the aquatic life criterion. Nickel ($p=0.0279$) EMCs were significantly greater at gas well sites compared to reference sites. The median EMC for nickel at gas well sites was over 3 times greater than the median EMC at reference sites. Potential nickel sources are both natural and anthropogenic. Natural sources include windblown soil and dust, forest fires, volcanoes, vegetation, and meteoric dust (USEPA, 1984). Anthropogenic sources are both direct and indirect. Over 90% of direct sources are from end uses of nickel in the form of metal alloys and indirect sources are primarily the result of coal and oil combustion (USEPA, 1984). Nickel concentrations at reference sites may be influenced by atmospheric depositions of natural and anthropogenic sources, whereas nickel concentrations at gas well sites may be influenced by a combination of atmospheric deposition, wearing of operational equipment high in metal alloys, and the large amount of fuel combusted during drilling and fracturing operations.

All iron and manganese EMCs were above the detection limit at both gas well sites and reference sites. Both iron ($p<0.0001$) and manganese ($p<0.0001$) EMCs were significantly greater at gas well sites compared to reference sites. Iron EMCs at gas well sites were above the drinking water standard and aquatic life criterion 95 and 62% of the time, respectively. At reference sites, half of the iron EMCs exceeded the secondary drinking water standard, but none exceeded the aquatic life criterion. Manganese EMCs exceeded secondary drinking water standards 84% of the time. The median iron EMC at gas well sites was 13.5 times greater than the median EMC at the reference sites; the median manganese EMC was almost 29 times greater. Sources of iron and manganese at gas well sites are probably from both natural and anthropogenic sources. Iron is a major constituent of clay soils and is common in many rocks, including limestone, which is used as the base material for gas well pads. Iron is also used in the production of metal alloys and is the main component of steel. Manganese is naturally occurring in many salts and minerals and is frequently associated with iron uses such as metal alloys and chemical reagents (USEPA, 1986). Lead EMCs were above detection, and the drinking water standard of zero, 58% of the time at gas well sites and 13% of the time at reference sites. Neither gas well sites nor reference sites had EMCs above the aquatic life criterion for lead. Machinery, equipment, construction materials and atmospheric deposition are potential sources of lead at gas well sites.

Kayhanian et al. (2001) measured metal concentrations in storm water runoff from 15 highway construction sites in California. Generally, mean concentrations of cadmium, chromium, copper, and nickel concentrations at gas well sites were similar to mean concentrations reported by Kayhanian et al. (2001). However, lead and zinc mean concentrations were higher at the highway constructions sites. Similarities between gas well sites and highway construction sites could be due to similar sources that include engine wear, brakes, tires, and automobile emissions.

Hudak and Banks (2006) reported metal concentrations for first flush and composite (equal to EMCs since samples were collected on flow intervals) storm water samples collected from three local mixed use (agricultural/urban) watersheds, including the Hickory Creek watershed. In their study, only lead and zinc had median composite concentrations above the detection limit. For lead, the composite median concentration was 0.0043 mg l^{-1}; comparatively median lead EMCs at gas well Site 2 and Site 3 were 0.006 and 0.009 mg l^{-1}, respectively.

Median EMCs of lead and zinc were below the detection limit at reference sites. The median zinc composite concentration reported by Hudak and Banks (2006) was 0.059 mg l^{-1}. The highest median EMC observed at gas well Site 2 was 0.047 mg l^{-1}. Maximum EMCs were higher than maximum composites reported by Hudak and Banks (2006) at one or more gas well sites for chromium, copper, lead, and nickel. Maximum EMCs were less than maximum composites for chromium, copper, lead, nickel, and zinc at reference sites. Site 3R had a higher maximum cadmium EMC, compared to the maximum composite reported by Hudak and Banks (2006). Comparison of gas well site data with local watershed data generally indicates that gas wells have higher maximum metal concentrations. In contrast, the median and maximum EMCs for all metals observed at the reference sites were lower than the median and maximum composites reported by Hudak and Banks (2006), with the exception of cadmium.

Primary sources of petroleum hydrocarbons at natural gas well sites in North Central Texas are refined petroleum products used by equipment and machinery on site such as gasoline, diesel, hydraulic oil, lubricating oils and grease. These constituents could find their way onto the site and then into storm water as a result of accidental spills, illegal dumping, and incidental runoff. Other potential sources include fluids used in the drilling process and crude oil produced along with natural gas; however these sources are thought to be low for gas wells drilled in the Barnett Shale since these wells typically use water-based drilling fluids and generally do not produce appreciable amounts of crude oil along with natural gas. TPH concentrations at gas well sites and reference sites were below the detection limit for all samples analyzed. BTEX was detected in a few of the discrete gas well sites samples (Appendix B), but all EMCs were below the detection limit. At reference Site 2R, BTEX EMCs were above the detection limit (Table 2.7) for all events sampled. Total BTEX standard for discharge of water contaminated by petroleum fuel or petroleum substances in Texas waters is 0.10 mg l^{-1} (TCEQ, 2007). Reference site BTEX EMCs were less than the Texas standard, ranging from 0.003 to 0.008 mg l^{-1} (Appendix B). The source of BTEX at reference Site 2R is unknown, but potential sources include farming equipment or illegal dumping.

Turbidity and Total Suspended Solids

Turbidity and TSS EMC summaries are provided in Table 2.5. Turbidity at gas well sites was high, ranging from 690.7 to 2040.8 NTU. In contrast, turbidity at reference sites was low, ranging from 3.3 to 40.2 NTU. Turbidity EMCs were significantly higher ($p<0.0001$) at gas well sites and the median EMC of gas well sites was 42 times greater than the median EMC of reference sites. The median gas well site EMC for turbidity was 10 times greater than the median EMC of 15 storm events monitored by the City of Denton (2007) near the outlet of the Hickory Creek watershed (Figure 2.1). This location was monitored quarterly from 2001 through 2006 as part of the City of Denton's Watershed Protection Program.

TSS EMCs were significantly greater ($p<0.0001$) at gas well sites compared to reference sites. EMCs at gas well sites ranged from 394 to 9898 mg l^{-1} and ranged from 3 to 43 at reference sites. Across all gas well site storm water samples (n=663), concentrations ranged from a few to 26,560 mg l^{-1}. The median TSS EMC at gas well sites was 157 times greater than the median EMC at reference sites and 36 times greater that than the median EMC of storm events monitored near the outlet Hickory Creek watershed (City of Denton, 2007). Based on concentration data collected in this study, annual estimated sediment yield for gas well Site 2, Site 3, and Site 4 was 41, 29, and 21 t ha^{-1} yr^{-1}, respectively. Based on these data illustrating increased turbidity and TSS, erosion and sediment control practices are recommended on gas well sites to reduce adverse of effects of increased sediment yields.

Wolman and Schick (1967) conducted one of the first studies that attempted to measure sediment concentrations and annual yields from construction sites. At two construction site locations near Baltimore, Maryland, sediments were sampled from nearby streams and were found as high as 60,000 mg l^{-1}. Sediment yields from the same sites were estimated at 253 and 491 t ha^{-1} yr^{-1} using measured sediment concentrations and rainfall-flow relationships. The authors point out that these yields were extrapolated from exceedingly small sites that were assumed to be under construction for an entire year.

In southeastern Wisconsin, three construction sites were monitored over a two-year period using automated storm water samplers at their watershed outlet (Daniel et al., 1979). In this study, sediment concentrations ranged from a few mg l^{-1} for small storms up to 60,000 mg l^{-1} for extreme events. Concentrations for moderate storms were around 15,000 to 20,000 mg l^{-1} and the average annual sediment yield of the three sites was 19.2 t ha^{-1} yr^{-1}. Madison et al. (1979) also collected storm water samples using automated samplers from residential construction sites in Wisconsin over a two-year period. During the first year of this study construction was intense, although by the second year the sites were stabilizing. In the first year, when in-storm variability was high, sediment concentrations ranged from a few hundred to as high as 75,000 mg l^{-1} with EMCs ranging from 2,500 to 7,000 mg l^{-1}. In the second year, in-storm variability was lower and concentrations ranged from 100 to 13,000 mg l^{-1} with EMCs ranging between 1000 and 3,500 mg l^{-1}. Annual sediment yields at the three sites under developing conditions ranged from 15.9 to 36.3 t ha^{-1} yr^{-1}, although yields decreased during the second year of the study at two sites. Yields increased slightly at the third site because construction continued.

Schueler and Lugbill (1990) took grab samples at 6 construction sites during the middle portion of 10 storm events. In their study, TSS concentrations ranged from 24 to 51,800 mg l^{-1} with a median of 680 mg l^{-1}. Kayhanian et al. (2001) monitored 15 highway construction sites in California using automated sampling equipment. Sediment concentrations at these sites ranged from 12 to 3,850 mg l^{-1} with mean of 499 mg l^{-1}. Lower median/mean concentrations found in these studies is probably due to the use erosion control measures that had been implemented at all sites in these studies, whereas runoff sampled at the gas well sites assessed in the current study were not influenced by any erosion control measures.

Nelson (1996) also used grab samples to characterize sediment concentrations at 5 construction sites in Alabama. Sediment concentrations at these sites ranged from 100 to 27,000 mg l^{-1} with a median concentration of 4,300 mg l^{-1}. Annual sediment yields were estimated at 265 t ha^{-1} yr^{-1}. More recently, USGS (2000) sampled runoff from the edge of two small construction sites; one residential (0.14 ha) and one commercial (0.70 ha). At the commercial site storm EMCs ranged from 76 to 22,285 mg l^{-1} with an average EMC of 15,000 mg l^{-1}. Storm EMCs at the residential site ranged from 19 to 14,074 mg l^{-1} and averaged 2,400 mg l^{-1}. Annual sediment yield for the commercial and residential sites, estimated from regression techniques, were 7.6 and 1.8 t ha^{-1} yr^{-1}, respectively. While sediment concentrations measured by USGS (2000) were similar to those measured in the current study, annual yields were less. The difference in annual yields may be the result of less runoff occurring at the USGS sites compared to gas well sites, which generally have very low infiltration rates resulting in a high proportion of runoff.

While there is much variability in sediment concentration and annual yields from study to study, several similarities exist. Overall, sediment concentrations reported from previous construction site studies range from a few mg l^{-1} to 75,000 mg l^{-1} with the storm EMC or medians generally falling between 1,000 and 20,000 mg l^{-1}. At gas well sites concentrations range from a few to 26,560 mg l^{-1} with storm EMCs ranging from 394 to 9898 mg l^{-1}. A few of the reported annual sediment yields (Nelson, 1996; Wolman and Schick, 1967) were much higher than those estimated for gas well sites; however, many others reported similar annual yields ranging from 1.7 to 36.3 t ha^{-1} yr^{-1} (Daniel et al., 1979; Madison et al., 1979; USGS, 2000).

Conclusion

In North Central Texas, storm water runoff was collected from the edge of three natural gas well sites and two small drainage basins representing natural pre-development conditions (referred to as reference sites). Storm water samples were analyzed for conventional water quality parameters, metals, petroleum hydrocarbons, and sediments. Event mean concentrations (EMC) of alkalinity, calcium, chlorides, conductivity, hardness, and pH were significantly higher at gas well sites compared to reference sites. Total dissolved solids EMCs tended to be higher at gas well sites, but differences were not statistically significant.

Generally, the concentration of metals was higher at gas well sites compared to reference sites, and EMCs were significantly greater for Fe, Mn, and Ni. A number of storm EMCs at gas well sites were above national drinking water standards and aquatic life criteria for some constituents. At reference sites fewer EMC exceedances of drinking water standards occurred and only one EMC exceeded the aquatic life criterion for cadmium. Median EMCs from gas well sites were similar to metals EMCs reported by Hudak and Banks (2006) for local mixed-use watersheds, but maximum EMCs at gas well sites were generally higher. At the reference sites, median and maximum metal EMCs were generally lower than those observed by Hudak and Banks (2006). Overall, the concentrations of metals tend to be higher at gas well sites compared to both nearby reference sites and storm water runoff from local mixed-use watersheds. Total petroleum hydrocarbons (TPH) were not detected in any of the samples collected at gas well sites or reference sites, although a few individual gas well site samples contained low concentrations of Benzene-Toluene-Ethylbenzene-Xylene (BTEX). However, BTEX EMCs were below detection limits.

TSS and turbidity EMCs at gas wells sites were significantly greater than those observed at reference sites. The median TSS EMC at gas well sites was 136 times greater than the median EMC at reference sites. Compared to the median EMCs of storm sampled near the outlet of the Hickory Creek Watershed by the City of Denton's Watershed Protection Program, the gas well site median EMC was 36 times greater. TSS EMCs and annual sediment loadings at gas well sites, which ranged from 394 to 9898 mg l^{-1} and 21 to 40 t ha^{-1} yr^{-1}, respectively, were comparable to those reported by previous studies aimed at characterizing sediments in construction site runoff. It is important to note that annual loadings are not total site loadings as erosion occurring from the fill slope was not measured by the sampling design. These results indicate that gas well site construction activities greatly increase the rate of sedimentation compared to pre-development conditions, and that these increases are similar in magnitude to typical construction sites that are currently regulated under the federal NPDES program.

The findings in this research suggest that gas well sites have the potential to negatively impact the aquatic environment due to site activities that result in increased sedimentation rates and an increase in the presence of metals in stormwater runoff. While these activities do not appear to result in high concentrations of petroleum hydrocarbons in storm water runoff, accidental spills and leaks are still a potential source of impact. In lieu of federal storm water requirements for natural gas exploration and development sites, state and local governments should consider some form of regulation, perhaps similar to current Phase I and Phase II NDPES requirements for construction sites, to reduce the potential impact of storm water runoff from these sites. Regulatory requirements should include storm water pollution prevention plans, erosion and sediment control best management practices, provisions for containing spills and leaks, procedures for site inspections and enforcement of control measures, and sanctions to ensure compliance.

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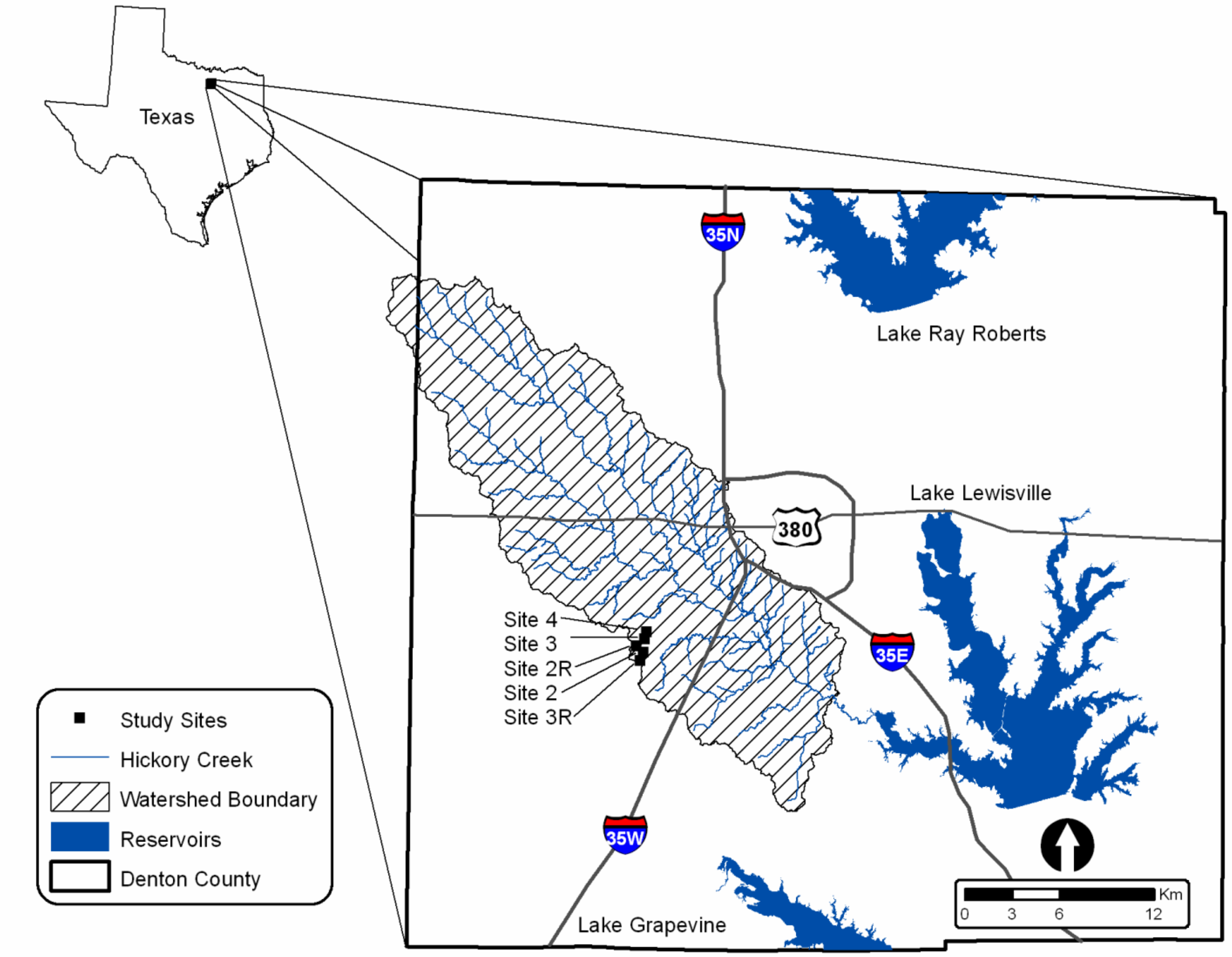

Figure 2.1. Study area - Denton County, Texas.

Figure 2.2. Gas well pad (Site 2) on modified hillslope.

Table 2.1. Site Characteristics and Storm Event Sampling

	Site 2		Site 3		Site 4		Site 2R	Site 3R
	Cut Slope	Pad Surface	Cut Slope	Pad Surface	Cut Slope	Pad Surface	Catchment Area	Catchment Area
Slope Length (m)	34.6	77.4	10.0	79.2	12.0	120.0	244.0	236.0
Average Slope (%)	9.0	1.5	31.0	0.6	12.0	0.4	7.3	5.7
Sampled Area (ha)	0.61		0.40		0.20		4.69	4.53
Soil Series	Medlin		Medlin		Sanger		Medlin	Medlin
Storm Events Sampled	17		12		11		5	5
Sampling Period	31 Oct. 2005 to 29 Jun. 2007		18 Mar. 2006 to 18 Jun. 2007		5 May 2007 to 18 Jun. 2007		24 Apr. 2007 to 18 Jun. 2007	30 Mar. 2007 to 29 Jun. 2007

Table 2.2. Methods and Detection Limits for Analyses

Parameter	Method	Detection Limit
Alkalinity	SM 2320 B	1.0 mg l^{-1}
Calcium	EPA 200.8	0.5 mg l^{-1}
Chlorides	SM 4500 Cl (D)	0.15-10.0[a] mg l^{-1}
Conductivity	SWQMP[b]	10 S m^{-1}
Hardness	SM 2340 C	1.0 mg l^{-1}
pH	SWQMP[b]	NA
TDS	SWQMP[b]	10.0 mg l^{-1}
TSS	SM 2540 D	4.0 mg l^{-1}
Turbidity	SWQMP[b]	NA
Arsenic (As)	EPA 200.8	0.01 mg l^{-1}
Cadmium (Cd)	EPA 200.8	0.001 mg l^{-1}
Chromium (Cr)	EPA 200.8	0.01 mg l^{-1}
Copper (Cu)	EPA 200.8	0.01 mg l^{-1}
Iron (Fe)	EPA 200.8	0.05 mg l^{-1}
Lead (Pb)	EPA 200.8	0.001 mg l^{-1}
Manganese (Mn)	EPA 200.8	0.01 mg l^{-1}
Nickel (Ni)	EPA 200.8	0.01 mg l^{-1}
Zinc (Zn)	EPA 200.8	0.05 mg l^{-1}
TPH	TCEQ 1005.3	5.0 mg l^{-1}
BTEX	EPA 2081 B	1.0 μg/L[c]; 2.0 μg/L[d]

[a] based on turbidity of sample; [b] Surface Water Quality Monitoring Procedures Manual (TCEQ, 1997) using Hydrolab, Y.S.I., or other similar meter; [c] for each Benzene, Toluene, Ethylbenzene, o-Xylene; [d] for each m-Xylene and p-Xylene

Table 2.3. SNK Multiple Comparison Test Results

	Gas Well Sites			Reference Sites	
Parameter	Site 2	Site 3	Site 4	Site 2R	Site 3R
Alkalinity	A[a]	A	A	B	B
Chlorides	A	AB	A	AB	B
Conductivity	A	A	A	A	A
Hardness	A	A	A	B	B
pH	A	A	A	B	B
TDS	A	A	A	A	A
TSS	A	A	A	B	B
Turbidity	A	A	A	B	B
Calcium	A	A	A	B	B
Iron	A	A	A	B	B
Magnesium	A	A	A	B	B
Manganese	A	A	A	B	B
Nickel	A	A	A	A	A

[a] Concentrations from sites with different letters are statistically different

Table 2.4. Wilcoxon Rank-Sum Results and Ratio of Median EMCs

Parameter[a]	n^b/n^c	p-value	Ratio[d]
Alkalinity	40/10	<0.0001	6.9
Chlorides	40/10	0.0058	1.7
Conductivity	40/10	<0.0483	1.2
Hardness	40/10	<0.0001	3.2
pH	40/10	<0.0001	1.2
TDS	40/10	0.0561	1.2
TSS	39/8	<0.0001	157.1
Turbidity	37/9	<0.0001	42.5
Calcium	36/8	<0.0001	8.0
Iron	36/8	<0.0001	13.5
Manganese	36/8	<0.0001	28.9
Nickel	36/8	0.0027	3.2

[a] Parameters containing more than 50% of storm event EMCs below detection limit (As, Cd, Cr, Cu, Pb) were not analyzed; [b] no. of gas well site samples; [c] no. of reference site samples: [d] ratio of median reference sites EMC and median gas well sites EMC.

Table 2.5. Physical and Chemical EMC Summary

Parameter	Variable	Gas Well Sites			Reference Sites	
		Site 2	Site 3	Site 4	Site 2R	Site 3R
Alkalinity	Range	146.7 – 9333.3	341.7 – 2650.0	293.3 – 2075.0	43.3 – 145.0	45.7 – 133.3
(mg l^{-1})	Mean / SD	1641.7 ± 2231.5	807.7 ± 602.4	764.9 ± 511.5	95.2 ± 44.4	94.4 ± 33.6
	Median / (n)[a]	810.0 (17)	629.2 (12)	630.0 (11)	94.7 (5)	95.3 (5)
Calcium	Range	48 – 2303	63 – 891	142 – 762	17 – 42	16 – 49
(mg l^{-1})	Mean / SD	448.9 ± 590.3	356.8 ± 257.1	308.5 ± 174.7	30.0 ± 11.2	33.8 ± 13.7
	Median / (n)	209.0 (15)	303.0 (11)	258.0 (10)	30.5 (4)	35.0 (4)
Chlorides	Range	20.3 – 94.7	17.3 – 280.0	22.7 – 240.0	21.3 – 56.0	14.7 – 29.3
(mg l^{-1})	Mean / SD	47.9 ± 22.8	57.2 ± 72.3	82.7 ± 62.0	35.7±12.9	21.3 ± 6.8
	Median / (n)	47.3 (17)	32.7 (12)	70.7 (11)	35.7 (5)	18.7 (5)
Conductivity	Range	123.3 – 571.8	59.5 – 343.8	115.6 – 1013.1	63.8 – 301.3	90.8 – 254.2
(S m^{-1})	Mean / SD	223.2 ± 89.4	179.3 ± 92.8	372.6 ± 259.1	163.6 ± 87.8	168.2 ± 58.3
	Median / (n)	197.3 (17)	150.7 (12)	332.3 (11)	149.3 (5)	168.3 (5)
Hardness	Range	99.3 – 493.3	128.0 – 466.7	136.7 – 580.0	46.7 – 152.0	43.3 – 125.3
(mg l^{-1})	Mean / SD	253.7 ± 118.3	251.9 ± 103.9	320.2 ± 133.6	86.1 ±41.2	84.4 ± 29.7
	Median / (n)	208.0 (17)	262.5 (12)	313.3 (11)	75.3 (5)	86.7 (5)
pH	Range	7.62 – 9.11	8.19 – 9.13	7.96 – 9.32	6.77 – 7.90	7.27 – 7.90
(std. units)	Mean / SD	8.59 ± 0.35	8.65 ± 0.29	8.5 ± 0.37	7.38 ± 0.41	7.54 ± 0.26
	Median / (n)	8.63 (17)	8.67 (12)	8.47 (11)	7.47 (5)	7.48 (5)
TDS	Range	79.0 - 318.0	38.6 – 223.5	75.5 – 657.3	41.6 – 195.3	59.0 – 165.6
(mg l^{-1})	Mean / SD	146.7 ± 61.3	116.5 ± 60.5	244.9 ±167.6	106.2 ± 56.8	111.1 ± 38.2
	Median / (n)	128.3 (17)	96.9 (12)	237.7 (11)	97.0 (5)	112.3 (5)
TSS	Range	781.0 – 9898.0	906.9 – 5968.0	394.0 – 4608.5	2.7 – 42.8	5.0 – 22.0
(mg l^{-1})	Mean / SD	4233.8 ± 2875.4	2988.1 ± 1599.8	2208.0 ± 1219.8	20.0 ± 17.8	16.2 ± 7.6
	Median / (n)	3370.8 (17)	2969.0 (11)	1894.0 (11)	17.3 (4)	18.9 (4)
Turbidity	Range	690.7 – 2033.9	931.4 – 2040.8	457.0 – 1427.1	3.3 – 40.2	3.3 – 28.5
(NTU)	Mean / SD	1426.9 ± 346.5	1147.2 ± 329.8	993.4 ± 261.5	20.0 ±15.2	19.7 ± 11.6
	Median / (n)	1000.0 (14)	1000.0 (11)	982.3 (11)	15.2 (5)	23.5 (4)

[a] number of samples

Table 2.6. Heavy Metals EMC Summary

		Gas Well Sites			Reference Sites	
Parameter	Variable	Site 2 (15)[a]	Site 3 (11)	Site 4 (10)	Site 2R (4)	Site 3R (4)
Arsenic	Range	BDL[b]	BDL	BDL	BDL	BDL
(mg l^{-1})	Mean / SD	BDL	BDL	BDL	BDL	BDL
	Median	BDL	BDL	BDL	BDL	BDL
Cadmium	Range	BDL - .002	BDL – 0.005	BDL – 0.004	BDL	BDL – 0.012
(mg l^{-1})	Mean / SD	BDL	0.001 ± 0.001	0.001 ± 0.001	BDL	0.004 ± 0.006
	Median	BDL	BDL	BDL	BDL	BDL
Chromium	Range	BDL – 0.085	BDL – 0.052	BDL – 0.038	BDL	BDL
(mg l^{-1})	Mean / SD	0.022 ± 0.024	0.016 ± 0.018	BDL	BDL	BDL
	Median	0.012	BDL	BDL	BDL	BDL
Copper	Range	BDL – 8.347	BDL – 0.035	BDL – 0.048	BDL	BDL – 0.011
(mg l^{-1})	Mean / SD	0.574 ± 2.150	0.017 ± 0.012	0.012 ± 0.016	BDL	BDL
	Median	0.015	0.019	BDL	BDL	BDL
Iron	Range	0.4 – 36.7	0.2 – 26.4	0.4 – 21.4	0.2 – 0.3	0.2 – 0.5
(mg l^{-1})	Mean / SD	9.23 ± 10.3	7.61 ± 8.73	4.25 ± 6.29	0.28 ± 0.05	0.35 ± 0.13
	Median	6.0	4.5	2.4	0.3	0.35
Lead	Range	BDL – 0.049	BDL – 0.030	BDL – 0.022	BDL	BDL - 0.001
(mg l^{-1})	Mean / SD	0.011 ± 0.015	0.011 ± 0.011	0.005 ± 0.007	BDL	BDL
	Median	0.006	0.009	0.002	BDL	BDL
Manganese	Range	BDL – 1.311	0.02 – 0.926	0.045 – 0.853	0.006 – 0.011	0.004 – 0.021
(mg l^{-1})	Mean / SD	0.358 ± 0.406	0.371 ± 0.297	0.267 ± 0.235	0.009 ± 0.002	0.012 ± 0.008
	Median	0.241	0.298	0.179	0.009	0.012
Nickel	Range	0.003 – 0.133	0.003 – 0.088	0.006 – 0.071	0.004 – 0.010	0.002 – 0.013
(mg l^{-1})	Mean / SD	0.036 ± 0.038	0.031 ± 0.029	0.021 ± 0.019	0.007 – 0.003	0.008 ± 0.006
	Median	0.021	0.024	0.017	0.007	0.009
Zinc	Range	BDL – 0.188	BDL – 0.098	BDL – 0.119	BDL – 0.036	BDL – 0.03
(mg l^{-1})	Mean / SD	0.048 ± 0.054	0.042 ± 0.034	0.034 ± 0.040	0.015 ± 0.015	0.011 ± 0.013
	Median	0.023	0.047	0.01	BDL	BDL

[a] number in parentheses indicates the number of storm events analyzed for metals; [b] below detection limit

Table 2.7. Drinking Water Standards, Aquatic Life Criteria, and Local Conditions (Values in mg l^{-1})

			Runoff from Local Watersheds[a]	
Parameter	Standard	Aquatic Life Criteria	Median EMC	Maximum EMC
Arsenic	0.01[b]	0.34	BDL	BDL
Cadmium	0.005[b]	0.002	BDL	0.012
Chromium	0.1[b]	0.57	BDL	0.063
Copper	1.3[c]	0.013	BDL	0.112
Iron	0.3[d]	1.0	-	-
Lead	0[b]	0.065	0.0043	0.0208
Manganese	0.05[d]	NA	-	-
Nickel	NA	0.47	BDL	0.11
Zinc	5.0[d]	0.12	0.059	1.343
TPH	15.0[e]	NA	-	-
BTEX	0.1[e]	NA	-	-

[a] Hudak and Banks, 2006;[b] MCL;[c] MCL goal;[d] secondary standard; [e] TPDES general permit no. TXG830000

CHAPTER 3

EVALUATION OF WEPP FOR RUNOFF AND SEDIMENT YIELD PREDICTION ON NATURAL GAS WELL SITES

Introduction

Sediment is the leading source of water quality impairment in US rivers and streams and is the third most ubiquitous source of impairment in US lakes and reservoirs (USEPA, 2000). Although the movement of sediment into water bodies is a natural process, its severity can be amplified by land disturbing construction activities. Toy and Hadley (1987) estimated construction activities had disturbed nearly 1.7% of all US land by 1980. Estimates of annual sediment delivery into US surface waters resulting from construction activities has ranged from 80 million tons [73 million tonnes] (USDOI, 1970) to 5 billion tons [4.5 billion tonnes] (Willett, 1980). Erosion rates from construction have been estimated to be 10 to 100 times the rate of agricultural land use (Goldman, 1986), and construction sites are by far the leading source of sediment in developing areas, with sediment yields ranging from a few tonnes to over 1100 tonnes ha^{-1} yr^{-1} (USEPA, 2002).

Negative impacts from erosion and sedimentation result when excess sediment is suspended in the water column or deposited in stream channels and lake bottoms. Suspended sediment can reduce in-stream photosynthesis, while nutrients in eroded soils can contribute to algal blooms and lake eutrophication (Goldman, 1986). Highly turbid water may result in the loss of sediment intolerant fish species (Poff and Allen, 1995), dramatically increase water treatment costs (AWWA, 1990), and diminish direct and indirect recreational experiences (Clark et al., 1995). Once deposited, sediment can substantially alter stream ecosystems by smothering benthic communities, reducing fish egg survival rates, reducing channel capacity, exacerbating downstream bank erosion and flooding, and reducing storage in reservoirs (Schueler, 1997). It has been estimated that the cost of physical, chemical, and biological damage from erosion and sedimentation in North America may exceed $16 billion annually (Osterkamp et al., 1998).

Natural gas exploration and production is a type of land disturbing activity that requires construction of a well site, access roads, and pipelines. These construction activities have the potential to accelerate soil loss due to land cover modifications, increased slopes, and flow concentration. In 2006, almost 30,000 natural gas wells were drilled nationwide (API, 2007), which is a substantial number considering that each well site disturbs approximately 1 to 2 ha of land surface. While it is fairly well documented that typical residential and commercial construction activities greatly increase erosion and sedimentation, little is known about erosion and sedimentation from natural gas exploration and production activities. Currently, oil and gas field operations and construction activities are exempt from federal national pollutant discharge elimination system (NPDES) permitting requirements (USEPA, 2006). Since the NPDES requires erosion and sediment control best management practices (BMP) to minimize off site movement of sediment from construction sites, potential impacts from unregulated oil and gas sites may be a concern for state and local governments responsible for ensuring water quality.

Erosion models have been used for decades to predict soil loss and land management effects from cropland, rangeland, and, to a lesser extent, disturbed site conditions. For construction sites, the most appropriate erosion prediction models are process-based and maintain both empirical and physical relationships within a physically based structure (Moore et al., 2007). The water erosion prediction project (WEPP) and version 2 of the revised universal soil loss equation (RUSLE2) both meet these criteria and have both been used for modeling soil loss and sediment yield from disturbed land cover conditions. However, WEPP provides a few advantages over RUSLE2, including: (1) the ability to estimate spatial distributions of both soil loss and deposition along a hillslope, (2) an interface to predict runoff and sediment yield from single storm events in addition to annual averages, and (3) the capability of estimating erosion and deposition on hillslopes and small watersheds.

Several researchers have evaluated WEPP parameters with measured data from agricultural fields (Liebenow, 1990; Risse et al., 1994, 1995a,b; Zhang et al., 1995a,b; Nearing et al., 1996; Zhang, 1996; Tiwari, 2000; Bhuyan, 2002), rangelands (Nearing et al., 1989; Simanton et al., 1991;Wilcox et al., 1992; Savabi et al., 1995), small watersheds (Nearing and Nicks, 1997; Liu et al., 1997), and forests (Morfin et al., 1996; Tysdal et al., 1997; Elliot 2004; Covert 2005; Dun, 2006). However, there has been much less work focusing on evaluating WEPP parameters for construction site conditions (Lindley et al., 1998; Laflen et al., 2001; Pudasaini, 2004; Moore et al., 2007).

The objective of this study was to evaluate WEPP predictions of runoff and sediment yields relative to measured data from two natural gas well sites in North Central Texas. Model results were evaluated with Nash-Sutcliffe efficiency (NSE), percent bias (PBIAS), and the ratio of the root mean square error to the standard deviation of measured data (RSR). Comparison of measured and predicted runoff and sediment yield also included consideration of uncertainty in the measured calibration and validation data.

Materials and Methods

Site Description

Input data for model calibration and validation were collected from two natural gas well sites located in the Grand Prairie physiographic region of North Central Texas approximately, at 97.23° N and 33.16° W. Grand Prairie physiography consists of gently sloping grasslands with scattered shrubs, and trees primarily along creek bottoms. Site soil was classified as Medlin stony clay (fine, montmorillonitic, thermic, vertisols) on slopes of 5 to 12% (USDA SCS, 1980). This soil type is moderately alkaline and has very low permeability, high runoff potential, and severe erosion potential (USDA SCS, 1980).

Both gas well sites were constructed on 5% slopes, which required leveling the surface for the gas well pad surface, resulting in site profiles consisting of a cut slope, pad surface, and fill slope that was approximately 100 m in length (Figure 3.1). The pad surface is relatively flat and is used for drilling activities and equipment storage. The term “cut slope” generally refers to the face of an excavated bank required to lower the ground to a desired profile. In contrast, a “fill slope” refers to a surface created by filling an area with soil. All slopes were compacted with a mechanical roller and an all-weather surface of Grade 1 Flex Base was applied to the pad surface. Flex Base is a gravely aggregate commonly used for temporary roads, base material underneath asphalt and concrete paving, and construction pad caps. The Flex Base surface application was approximately 0.3 m in depth and covered an area approximately 0.5 ha. The soil on the cut and fill slopes covered an area of approximately 0.5 ha and was left exposed after compaction. It is important to note that infiltration rates can be reduced by up to 99% on construction sites compared to predevelopment conditions (Gregory, 2006). Site characteristics are described in Table 3.1.

Site Monitoring

Flow-interval (1.0 mm of volumetric runoff depth) storm water samples were collected with ISCO 6712 (ISCO, Inc., Lincoln, Nebraska) automated samplers. This method is recommended for small watershed sampling according to Harmel et al. (2006). Samples were taken at a single intake point near the bottom of a partially contracted sharp-crested 90^{o} V-notch weir located at the edge of each pad surface. A barrier was installed along the down slope portion of the pad surface to direct flow through the weir. This sampling design captures runoff from the cut slope and pad surface but does not capture runoff from the fill slope (Figure 3.1). Flow volume was monitored with ISCO 4250 velocity flow meters (ISCO, Inc., Lincoln, Nebraska) placed 1 m upstream from the outfall of each weir. Rainfall at each site was monitored with a tipping bucket style ISCO 674 Rain Gauge (ISCO, Inc., Lincoln, Nebraska). Both flow and rainfall data were logged at 5-min intervals. Fifteen storm events generated a total of 20 sampling events at the two sites (Table 3.2).

Total suspended solids (TSS) concentrations were analyzed in collected samples using Standard Method 2540D (APHA, 1992). Because water samples were taken on consistent flow intervals, the arithmetic average of TSS concentrations represents the event mean concentration (EMC). Total storm loads were calculated by multiplying the TSS EMC by the total storm flow.

Model Description

WEPP is a process-based, distributed parameter, continuous simulation model based on fundamentals of stochastic weather generation, infiltration theory, hydrology, soil physics, plant science, hydraulics, and erosion mechanics (Flanagan et al., 1995). Infiltration is calculated using the Green Ampt Mein Larson (GAML) model (Mein and Larson, 1973; Chu, 1978) for unsteady rainfall. Runoff, the difference between the rainfall and infiltration, is routed overland using a semi-analytical solution of the kinematic wave model (Stone et al., 1992). WEPP's erosion component uses a steady-state sediment continuity equation that considers both interrill and rill erosion processes. Interrill erosion involves soil detachment and transport by raindrops and shallow sheet flow, while rill erosion processes describe soil detachment, transport, and deposition in rill channels (Flanagan and Nearing, 1995).

Input Parameters

Major inputs for WEPP include climate data, topography, management conditions, and soil attributes. WEPP's stochastic climate generator, CLIGEN, uses 10 daily climate parameters. Four precipitation parameters- precipitation, storm duration, peak intensity, and time to peak- were used to generate a single storm climate file for each event at each site. The other six climate parameters- maximum and minimum temperature, solar radiation, wind velocity and direction, and dew point temperature- were generated by CLIGEN during model simulation. Slope profiles for each site were derived from high resolution digital terrain models created from gas well site surveys. Slope profiles were simplified and entered into the WEPP using the slope editor (Table 3.1).

A management input file for a cut slope surface is available in the WEPP software and was used for the cut slope portion of the site. The WEPP default cut slope management parameters represent limited vegetation growth on a smooth soil surface. For pad surfaces, the initial plant parameters in the cut slope management file were modified to represent a rock surface. The principal characteristics of a rock surface are that it is extremely dense and has an extremely low decomposition rate (Laflen et al., 2001). Prior to model calibration, management file parameters as described above were further modified to represent gas well site conditions. Additional parameters modified in the management file are listed in Table 3.4.

Soil parameters for the cut slopes were obtained from WEPP's Medlin soil series input file. Soil information for any soil in the US can be obtained from the USDA-NRCS Soil Survey Geographic database (USDA-NRCS, 2007). For the pad surface soil parameters, a custom soil file was created using parameters suggested by Laflen et al. (2001) for soils underlying crushed rock in construction applications. This type of soil surface yields high runoff values with low soil loss.

Soil Parameter Calibration

Ideal model calibration involves: (1) using data that includes a range of conditions (Gan et al., 1997), (2) using multiple evaluation techniques (Legates and McCabe, 1999), and (3) calibrating all constituents to be evaluated (Moriasi et al., 2007). Using a similar approach to Bhuyan et al. (2002), model calibration was conducted using the smallest, middle, and largest sediment yield events over the study period to account for variation in the measured data. Soil parameters sensitive to model response were manually adjusted to bring the predicted runoff and sediment yield values within the range of observed values. Typically, calibration involves sensitivity analyses; however, several researchers (Nearing et al., 1990; Alberts et al., 1995; Bhuyan et al., 2002) have already found that baseline rill and interrill erodibility, effective hydraulic conductivity, and critical shear stress are sensitive model parameters in WEPP. These parameters were adjusted in order of their relative sensitivities to model response, with the most sensitive parameter adjusted first. Both predicted runoff and sediment yield were calibrated with these four parameters. The range of values used for calibration of soil erodibility for cut slopes were kept within suggested limits for cropland (Alberts et al., 1995). For gas well pad surfaces, the range of values was based on literature values for impervious site conditions (Laflen et al., 2001) and values provided in the WEPP management file for a "graveled road surface on clay loam." Ranges of soil parameter values used for calibration are shown in Table 3.4. Default and calibrated WEPP soil parameters are listed in Table 3.6.

Model Evaluation

Model evaluation techniques for calibration and validation should include at least one dimensionless statistic, one absolute error index statistic, one graphical technique, and other information such as the standard deviation of measured data (Legates and McCabe, 1999). Dimensionless techniques provide model evaluations in relative terms, whereas error indices quantify the differences in units of the data of interest (Legates and McCabe, 1999). Specific model evaluation statistics used in this research were selected based on recommendations according to Moriasi et al. (2007) and included Nash-Sutcliffe efficiency (NSE), root mean square error (RMSE)-observation standard deviation ratio (RSR), and percent bias (PBIAS). The

Nash-Sutcliffe model efficiency coefficient (Nash and Sutcliffe, 1970) is expressed in equation 1 as:

$$\text{NSE} = 1 - \left[\frac{\sum_{i=1}^{n} (O_i - P_i)^2}{\sum_{i=1}^{n} (O_i - O)^2} \right] \tag{1}$$

where O_i and P_i are observed and predicted values for the ith pair, and O is the mean of the observed values. NSE ranges from $-\infty$ to 1; a value of 1 indicates a perfect fit between the observed and predicted data. NSE values ≤ 0.5 are considered unsatisfactory (Moriasi et al., 2007), and NSE values ≤ 0 indicate the mean observed value is a better predictor than the simulated value.

Moriasi et al. (2007) developed a model evaluation statistic (RSR) that standardizes RMSE using the standard deviation of the observations. Since the RSR combines the error index and standard deviation, this statistic meets the model evaluation recommends of McCabe and Legates (1999). RSR is the ratio of the RMSE and standard deviation of the measured data, as calculated with equation 2:

$$\text{RSR} = \frac{\text{RMSE}}{\text{STDEV}_{\text{obs}}} = \frac{\left[\sqrt{\sum_{i=1}^{n} (O_i - P_i)^2} \right]}{\left[\sqrt{\sum_{i=1}^{n} (O_i - O)^2} \right]} \tag{2}$$

RSR ranges from 0 to a large positive value. Lower values indicate better model performance, with a value of 0 being optimal. RSR values > 0.70 are generally considered unsatisfactory (Moriasi et al., 2007).

PBIAS measures the average tendency of the simulated data derived from the model to be larger or smaller than measured data (Gupta et al., 1999). PBIAS is calculated as shown in equation 3:

$$\text{PBIAS} = \left[\frac{\sum_{i=1}^{n} (O_i - P_i) * (100)}{\sum_{i=1}^{n} (O_i))} \right] \tag{3}$$

Positive values indicate model overestimation bias, and negative values indicate model underestimation bias; a value of zero is optimal and indicates no bias. PBIAS has the ability to clearly indicate model performance (Gupta, 1999). PBIAS is generally considered unsatisfactory for runoff if the value is ≥ ±25 and unsatisfactory for sediment if the values is ≥ ±55 (Moriasi et al., 2007).

Measurement Uncertainty

Measurement uncertainty is rarely included in the evaluation of model performance, even though all measured data are inherently uncertain. Harmel and Smith (2007) developed modifications to the deviation term in four goodness-of-fit indicators (NSE, Index of Agreement, RMSE, and MAE) to improve the evaluation of hydrologic and water quality models based on uncertainty of measured calibration and validation data. Modification 1, which is applicable when the probable error range (PER) is known or assumed for each measured data point, was used in this research. Following procedures developed by Harmel et al. (2006), the PER for runoff and sediment loads was estimated based on the experimental site and data collection methods. For GW1, the PER for runoff was ±16% and for sediment loads was ±25%. For GW2, the PER for runoff and sediment loads was ±27% and ±33%, respectively. It is not uncommon for storm water data to consist of partially sampled events, incomplete flow data, or rainfall information obtained from a location other than the sample site, all of which increase measurement uncertainty. These issues, however, did not affect data used in this study. These PER estimates are comparable to expected uncertainty from typical sampling scenarios for runoff (±6% to ±19%) and for sediment loads (±7% to ±53%) from Harmel et al. (2006).

Once estimated, the PER is used to calculate the upper and lower uncertainty boundary for each measured data point. If the predicted value is within the uncertainty range, the deviation is set to zero (Harmel and Smith, 2007). For predicted values that lie outside the uncertainty boundaries, the deviation is the difference between the predicted value and the nearest uncertainty boundary. Modification 1 minimizes the error estimate for each measured and predicted data pair and was used in conjunction with NSE and RSR to calibrate and validate the model. RSR was adapted in the research to accommodate Modification 1.

Results and Discussion

Measured and predicted runoff and sediment yields are shown in Table 3.5. Measured event runoff at GW1 and GW2 ranged from 3.7 to 34.1 mm and 6.7 to 18.8 mm, respectively. Sediment yield was also greater for GW1, ranging from 51 to 668 kg compared to 53 to 270 kg for GW2. Three storm events were used to calibrate the soil parameters, and the remaining 17 events were used to validate the model. NSE, RSR, and PBIAS, as well as modified versions of NSE and RSR that consider measurement uncertainty were used to evaluate model performance. Model performance ratings were based on guidelines provided by Moriasi et al. (2007). Performance ratings and evaluation statistics are shown in Table 3.7.

Model Calibration

Model parameters were adjusted for the calibration set until model evaluation statistics for both runoff and sediment yield were "satisfactory" or better based on Moriasi et al. (2007) for all evaluation statistics (NSE > 0.50, RSR < 0.70, PBIAS for runoff ≤ ± 25, PBIAS for sediment ≤ ±55). Initially, default soil parameter values predicted runoff values in the range of measured values, but predicted sediment yields were substantially lower than measured values. In order to meet "satisfactory" model performance, interrill and rill erodibility values were increased and critical shear stress was decreased from default Medlin soil parameters. Similarly, interrill erodibility was increased and critical shear stress was decreased from the Flex Base soil parameters (Table 3.3). These changes resulted in higher predicted sediment yields compared to default Medlin and Flex Base soil parameters. Calibrated hydraulic conductivity values for both the Medlin soil and Flex Base were similar to default values. NSE for the calibration set for runoff and sediment were 0.52 and 0.49, respectively, and RSR for runoff and sediment yield were 0.70 and 0.72, respectively. While NSE of 0.49 and RSR of 0.72 fell just below the range of "satisfactory" model performance, when the model was evaluated according to the uncertainty limits of the measured data, modified NSE and RSR for runoff and sediment yield performance ratings increased to "very good." PBIAS values indicated that the calibrated model parameters under-predicted both runoff (-23%) and sediment yield (-24%) but model performance was "satisfactory." Model calibration results are illustrated graphically in Figure 3.2a, b.

Model Validation

Calibrated model parameters were applied to validation data for GW1 and GW2 separately. Runoff model performance was better for GW2, and sediment yield model performance was better for GW1. Model performance for GW1 was considered “good” with NSE and RSR values of 0.68 and 0.56 for runoff and 0.63 and 0.61 for sediment yield, respectively. Considering measurement uncertainty, Modification 1 resulted in “very good” performance ratings for NSE and RSR. Graphical results were in agreement with the statistical results (Figure 3.2c, d). A general visual agreement between measured and predicted data indicates adequate model performance over the range of constituents being simulated (Singh et al., 2004). PBIAS performance ratings were “good” for runoff and “very good” for sediment yield with values of 15% and -11%, respectively, that indicate slight under-prediction for runoff and slight over-prediction for sediment yield.

For GW2, model predictions were “very good” for runoff (NSE=0.76 and RSR= 0.49) but” unsatisfactory” for sediment yield (NSE=0.32 and RSR=0.83). However, Modification 1 improved NSE and RSR performance ratings from “unsatisfactory” to “very good.” Graphical results are shown in Figure 3.2 (e, f) and were in agreement with the statistical results. Runoff PBIAS estimates were “very good” for runoff (-2%) and “good” for sediment yield (16%). In contrast to GW1, the model under-predicted sediment yield.

Consideration of uncertainty in the measured data provides a realistic evaluation of model performance. If the model is judged solely on its ability to produce values similar to the measured data, instead of values within the uncertainty limits of the measured data, then the model may be assumed to be precise but may not be accurately reproducing actual hydrological and water quality conditions (Harmel et al., 2006). However, when measurement uncertainty is considered in model evaluation, it is important to estimate uncertainty appropriately without consideration of perceived deficiency for relatively high uncertainty estimates and without attempts to improve assessed model performance with inflated measurement uncertainty.

Model evaluation in this research demonstrates the improvement in assessed model performance that results from the consideration of measurement uncertainty. For runoff, all of the model evaluation statistics and graphical methods indicated "good" to "very good" performance of the calibrated model. For sediment load, the model evaluation statistics and graphical method produced mixed results from "unsatisfactory" to "very good." This mixed result confirms the importance of utilizing multiple evaluation methods to assess overall model performance as noted by Legates and McCabe (1999) and Moriasi et al. (2007). It is also important to note that (1) the assessment of "very good" model performance when measurement uncertainty was included indicates that simulated results were generally within the uncertainty boundaries of measured data and that (2) the statistics modified to consider measurement uncertainty provide valuable, supplemental information to be used in conjunction with traditionally-applied statistical and graphical methods for model evaluation.

Minor differences in GW1 and GW2 evaluation statistics and model performance could be due to numerous factors, including constantly changing micro-topography, slight differences in site construction practices, and the relatively small data set used to calibrate and validate the model. From event to event, runoff and erosion are constantly changing the micro-topography of the site by filling and creating sinks. While this phenomenon occurs to some extent at all scales, the relative effect on sediment yield at a small scale is potentially much greater than effects at larger scales. However, on relatively flat, highly modified surfaces, changing micro-topography is difficult to characterize from event to event. While construction practices are similar from site to site, minor differences in grading, filling, and compaction of the surface all have the potential to affect infiltration and soil erodibility properties. Finally, evaluation statistics used to calibration and validation are sensitive to small samples, although it should be noted that, small samples are not uncommon in model evaluations since storm water monitoring is resource intensive.

While there were some minor differences in runoff and sediment yields between sites, the predicted detachment and deposition patterns were similar. The majority of soil losses occur on the cut slopes at both sites. Maximum soil detachment for GW1 was 51 kg m^{-2} at 27.7 m down slope and for GW2 was 104 kg m^{-2} at 8.95 m down slope. Maximum deposition occurred at the base of both cut slopes and was 20.5 kg m^{-2} at 45.1 m down slope for GW1 and 188 kg m^{-2} at 12.3 m down slope for GW2. Pad surface soil detachment exceeded deposition at both sites but contributed only a small portion to overall sediment yields.

Application of WEPP to Disturbed Sites

In contrast to other land use practices such as agriculture, rangeland, and forest applications, few studies have tested WEPP on land disturbed by construction activities. Lindley et al. (1998) developed algorithms and computer code for the hydraulic portions of the WEPP Surface Impoundment Element (WEPPSIE) to evaluate practices to reduce erosion such as ponds, terraces, and check dams. The WEPPSIE sediment algorithms were verified against data collected on two experimental impoundments consisting of a total of 11 model runs. Laflen et al. (2001) provide recommendations for soil and management parameters for construction site conditions, such as paved surfaces, crushed rock, and erosion mats, but parameters were not verified with measured data. WEPP model predictions were found to be reasonable for three single storm event intensities on research plots for three land use treatments representing construction site conditions (rotary hoed, rolled smooth, and topsoil restored) (Pudasaini, 2004). Recently, Moore et al. (2007) were successful in developing and applying WEPP input parameters for construction and post-construction phases of a commercial construction site on a small 4 ha watershed. Soil and management parameters were tested and adapted based on 37 runoff samples and three sediment samples. Best model efficiencies for runoff and sediment yields resulted from replacing the surface soil horizon characteristics with subsurface horizon characteristics and supplementing the cut slope management parameters with experimental bare soil inputs.

WEPP's ability to model both temporal and spatial distribution of soil loss and deposition provides important model functionality for disturbed site conditions. WEPP can simulate runoff and sediment yields daily, monthly, annually, or by event. The temporal flexibility of the model is important for evaluating management alternatives. Laflen et al. (2001) used WEPP to estimate potential soil loss from a highway construction site for a variety of construction timeline scenarios to determine the critical time of year for severe erosion. The authors found that WEPP was applicable to construction sites in their application, although WEPP could be easier to use with some additional modifications including the ability to change materials and topography during the WEPP run. In terms of reducing source loads from disturbed areas, management alternatives may include planning construction to coincide with those seasonal weather cycles that are least likely to generate erosive storm events. Moore et al. (2007) illustrated how modeling periods could also be broken down according to changing site conditions, considering different soil and management characteristics and topography, which may be useful for evaluating sediment yields during various site development phases. Event based simulations allow for calibration and validation of WEPP using a relatively small amount of data, as illustrated in this research, compared to the data required to calibrate erosion models that estimate soil losses on an annual basis. Calibration and validation provides credibility to the model results that may not otherwise exist, which is particularly important when source assessments, load allocations, and management decisions are determined for specific site conditions. However, once the model has been calibrated and validated, WEPP should be run in continuous simulation to obtain an annual average. Annual averages determined from continuous simulation are more accurate because, unlike single storm predictions, continuous simulation can account for the complex overlap of temporal and spatial variability of both the driving force of erosion (i.e. rainfall) and the resisting force of the environment (i.e. erodibility) (Nearing, 2004).

Because sediment yields are commonly reported in annual terms, running the model in continuous simulation to obtain an annual average provides sediment yield predictions that can be compared to other studies. When calibrated gas well parameters were run in continuous simulation, annual predicted sediment yields from GW1 and GW2 were 38.0 and 20.9 t ha^{-1} yr^{-1}. Wolman and Schick (1967) conducted one of the first studies that attempted to measure annual yields from construction sites. Using measured sediment concentrations and rainfall-flow relationships, sediment yields from two sites were estimated at 253 and 491 t ha^{-1} yr^{-1}. Based on two years of monitoring, Daniel et al. (1979) reported that average sediment yield from three construction sites was 17.5 t ha^{-1} yr^{-1}. In another two-year study, sediment yields at three residential construction sites ranged from 39 to 90 t ha^{-1} yr^{-1} (Madison et al., 1979). More recently, USGS (2000) sampled runoff from the edge of two small construction sites, one residential (0.14 ha) and one commercial (0.70 ha). Sediment yield for the commercial and residential sites based on one year of data were 7.6 t ha^{-1} yr^{-1} and 1.8 t ha^{-1} yr^{-1}, respectively. A comparison of predicted annual sediment yields from gas well sites provided in this study to sediment yields reported in previous construction site studies suggests that, in terms of sediment yields, natural gas well sites are similar to construction sites.

Finally, the spatial component of erosion is important for designing the most effective erosion control practices and for targeting the most erodible areas of a hillslope. WEPP Hillslope contains erosion control management practices that are applicable to disturbed areas, including seeding and filter strips, and WEPPSIE has a suite of sediment control practices including terraces, check dams, filter fences, and straw bales. Other erosion control practices not specifically parameterized by default values in the model can be simulated according to specific runoff characteristics. For example, Laflen et al. (2001) explain how altering model defaults for plant growth and the critical shear value of soil can mimic the effects of an erosion mat.

Conclusion

In this study, WEPP runoff and sediment yield predictions were compared to measured data for two natural gas well sites located in North Central Texas. Model predictions were evaluated with graphical methods and NSE, RSR, and PBIAS statistics. Model predictions were also evaluated using modified versions of NSE and RSR that account for uncertainty in measured calibration and validation data. WEPP soil parameters were calibrated according to suggested parameters from the WEPP manual, model observations, and previous research. During the calibration process, rill and interrill erodibility, critical shear stress, and hydraulic conductivity were adjusted until predicted runoff and sediment yield values were "satisfactory." The calibration process resulted in rill and interrill erodibility parameters that were higher than default soil parameters and critical shear values that were lower that default values.

The calibrated model produced "good" to "very good" results for runoff and "unsatisfactory" to "very good" results for sediment yield. These results confirm the importance of utilizing multiple evaluation methods, both statistical and graphical, to assess overall model performance. The measurement uncertainty for the model validation data was estimated to be ±16% and ±27% for runoff and ±25% and ±33% for sediment yields, which is comparable to expected uncertainty from typical sampling scenarios. When measurement uncertainty was included in model evaluation, predictions were "very good" for both runoff and sediment yield. This alternative method, which compares predictions with uncertainty boundaries rather than single, inherently uncertain measured values, provides valuable supplementary information for model evaluation.

Additional monitoring of runoff and sediment yields for the same sites, additional sites located in different regions, and on different soil types and topographies would improve the evaluation of WEPP for natural gas well sites. However, since monitoring is expensive and site conditions may change substantially over time, we recommend that future erosion and runoff research related to gas well sites be conducted on research plots with rainfall simulation using methodologies similar to those that were used in previous WEPP calibration and validation studies.

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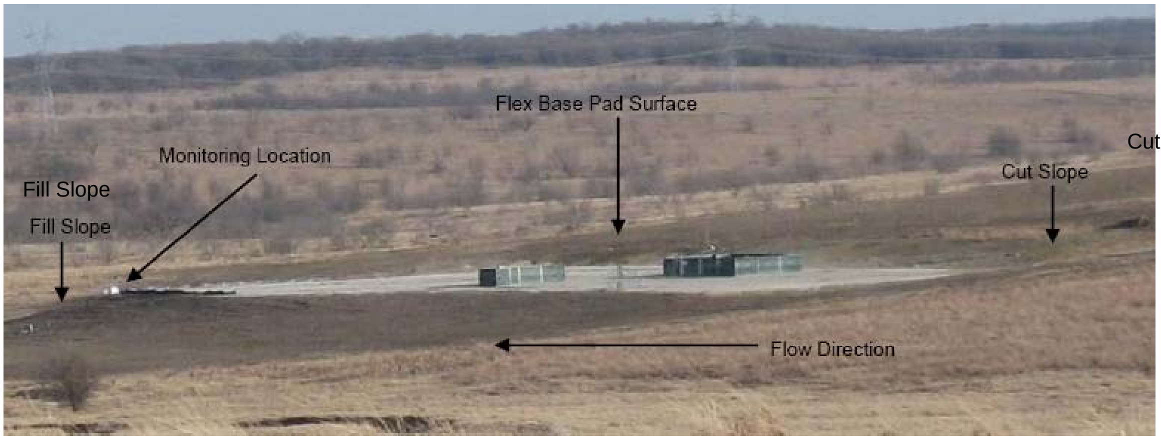

Figure 3.1. Gas well pad surface (GW1) on modified hillslope.

Table 3.1. Gas Well Site Characteristics

	Gas Well #1 (GW1)		Gas Well #2 (GW2)	
	Cut Slope	Pad Surface	Cut Slope	Pad Surface
Slope Length (m)	34.6	77.4	10.0	79.2
Average Slope (%)	9.0	1.5	31.0	0.6
Disturbed Area (ha)	2.1		1.9	
Sampled Area (ha)	0.45		0.36	
Soil Series	Medlin	Custom	Medlin	Custom
Management	Cut Slope		Cut Slope	
Storm Events Sampled	12		8	
Sampling Period	2 Feb. 2006 to 5 Nov. 2006		20 Mar. 2006 to 29 Nov. 2006	

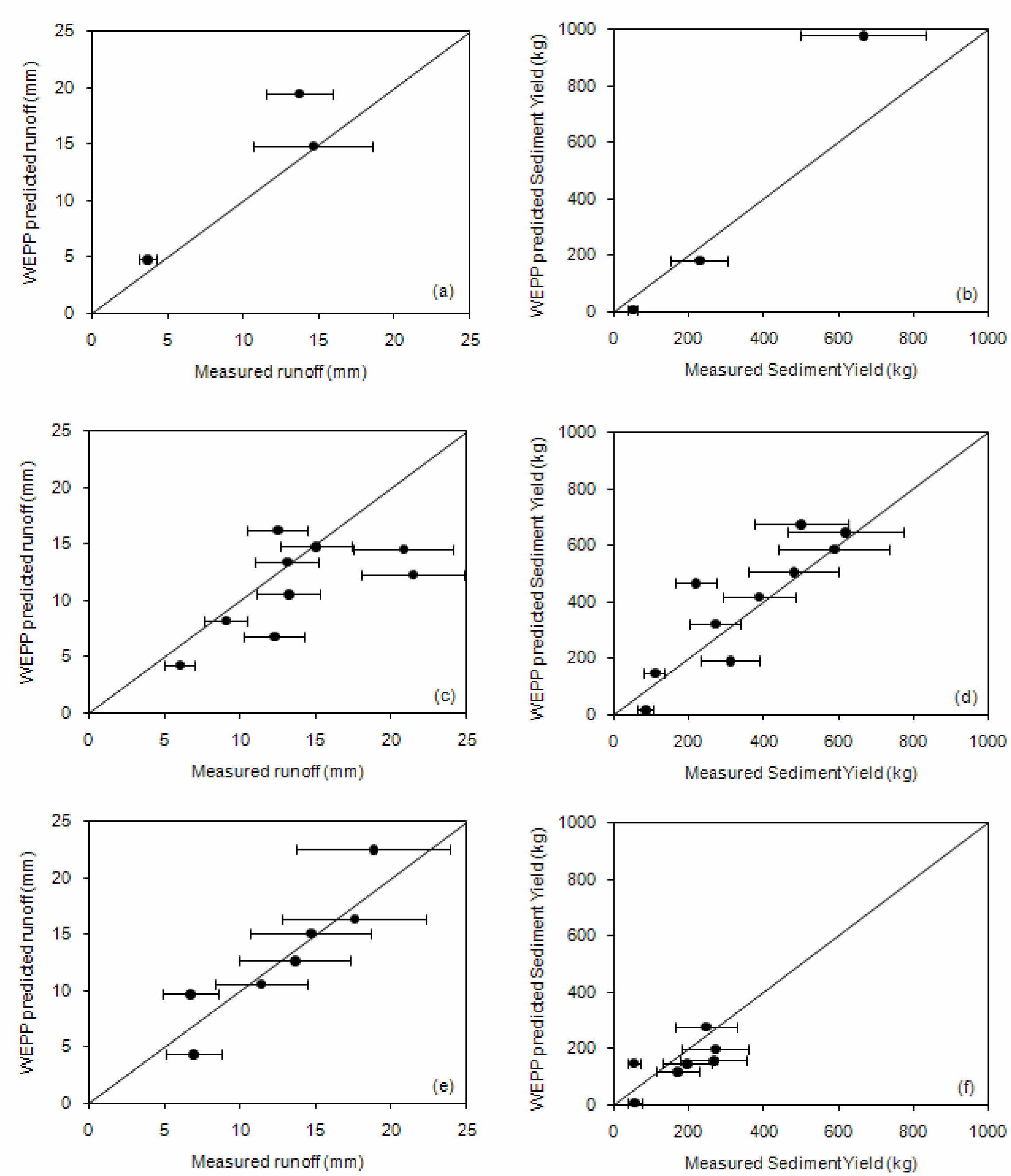

Figure 3.2. Scatterplots of measured and predicted runoff (mm) and sediment yield (kg) modified with Modification 1 to include the uncertainty range (PER) for each measured value: (a) calibrated runoff (PER = ±16%, ±27%); (b) calibrated sediment yield (PER = ±25%, ±33%); (c) GW1 runoff (PER = ±16%); (d) GW1 sediment yield (PER = ±25%); (e) GW2 runoff (PER = ±27%); (f) GW2 sediment yield (PER = ±33%).

Table 3.2. Precipitation Parameters for Sampling Events

Site	Sampling Date	Precip. (mm)	Peak Int. (mm h^{-1})	Storm Dur. (hr)	Time to Peak (%)
GW1	24 Feb. 2006	48.5	7.0	23.0	40
GW1	20 Mar. 2006	23.1	18.0	3.0	55
GW1	21 Apr. 2006	30.7	9.4	30.0	5
GW1	5 May 2006	21.6	2.9	17.0	18
GW1	6 May 2006	10.4	3.8	4.3	90
GW1[a]	17 June 2006	25.4	24.9	1.1	40
GW1	27 Aug. 2006	14.7	49.0	0.3	60
GW1[a]	29 Aug. 2006	14.2	2.3	12.5	25
GW1	18 Sept. 2006	21.1	8.3	11.0	60
GW1	10 Oct. 2006	21.8	17.5	1.5	5
GW1	15 Oct. 2006	25.4	4.1	10.0	50
GW1	5 Nov. 2006	14.0	13.0	1.1	70
GW2[a]	20 Mar. 2006	23.1	18.0	3.0	55
GW2	21 Apr. 2006	30.7	6.9	30.1	5
GW2	29 Arp. 2006	28.4	14.7	15.0	57
GW2	5 May 2006	19.0	15.0	3.1	23
GW2	6 May 2006	11.4	4.1	5.0	60
GW2	17 June 2006	20.0	15.0	2.0	45
GW2	5 July 2005	17.0	28.3	0.6	40
GW2	29 Nov. 2006	35.8	17.1	9	40

[a] Storm event used for calibration

Table 3.3. WEPP Input Management Parameters

	Cut Slope		Pad Surface	
	Default Input File	Modified Input File	Default Input File	Modified Input File
Darcy Weisbach friction factor	5	1	5	1
Days since last tillage	0	0	0	200
Days since last harvest	0	0	0	2000
Cumulative rainfall since last tillage (mm)	0	1000	0	1000
Initial interrill cover (%)	5	0	5	5
Initial ridge height after last tillage (mm)	1	1	1	2
Initial rill cover (%)	5	0	5	5
Initial roughness after last tillage (mm)	1	1	1	2
Rill spacing (cm)	0	60	0	0

Table 3.4. Calibration Range for Soil Parameters

	Cut Slope (Medlin)		Pad Surface (Flex Base)	
	Min.	Max.	Min.	Max.
Interrill Erodibility K_i (kg sec m^{-4})	5.0×10^5	12.0×10^6	1.0×10^7	1.0×10^2
Rill Erodibility Kr (sec m^{-1})	0.002	0.05	1.0×10^{-5}	1.0×10^{-3}
Crit. Shear Stress τ (Pa)	0.03	7.0	10	100
Hydraulic Cond. K_{ef} (mm h^{-1})	0.1	2.0	0.1	0.5

Table 3.5. Measured and Predicted Runoff and Sediment Yield

		Runoff (mm)		Sediment Yield (kg)	
Site	Sampling Date	Meas.	Pred.	Meas.	Pred.
GW1	24 Feb. 2006	34.1	28.5	311	190
GW1	20 Mar. 2006	15.0	14.8	500	677
GW1	21 Apr. 2006	12.4	16.3	219	468
GW1	5 May 2006	13.1	13.4	588	590
GW1	6 May 2006	6.0	4.3	84	16
GW1[a]	17 June 2006	13.7	19.5	668	982
GW1	27 Aug. 2006	9.0	8.2	482	508
GW1[a]	29 Aug. 2006	3.7	4.8	51	8
GW1	18 Sept. 2006	13.2	10.6	389	420
GW1	10 Oct. 2006	20.8	14.6	619	650
GW1	15 Oct. 2006	21.4	13.4	109	148
GW1	5 Nov. 2006	12.2	6.8	272	324
GW2[a]	20 Mar. 2006	14.6	14.9	230	271
GW2	21 Apr. 2006	14.7	15.5	54	38
GW2	29 Arp. 2006	17.5	16.4	270	242
GW2	5 May 2006	11.4	10.6	171	54
GW2	6 May 2006	6.9	4.2	56	9
GW2	17 June 2006	13.6	12.7	267	169
GW2	5 July 2005	6.7	10.2	196	275
GW2	29 Nov. 2006	18.8	26.2	247	459

[a] Storm event used for calibration

Table 3.6. Default and Calibrated WEPP Input Soil Parameters

Soil Parameter	Soil Texture	Hydrologic Class	Interrill Erodibility K_i (kg sec m^{-4})	Rill Erodibility K_r (sec m^{-1})	Crit. Shear Stress τ (Pa)	Hydraulic Cond. K_{ef} (mm h^{-1})	Sand (%)	Clay (%)	CEC (meq 100 g^{-1})	Rock (%)
Medlin[a]	Clay Loam	C	3.58×10^6	0.0069	3.5	0.73	30	45	39	3
Medlin[b]	Clay Loam	C	9.58×10^6	0.03	2.35	0.75	30	45	39	3
Flex Base[a]	n/a	n/a	1.0×10^3	0.0001	100	0.1	10	70	25	90
Flex Base[b]	n/a	n/a	1.0×10^6	0.0001	50	0.1	10	70	25	90

[a] Default soil parameters

[b] Calibrated soil parameters

Table 3.7. Evaluation Statistics and Performance Ratings

	NSE				RSR				PBIAS	
	NSE	Performance Rating[a]	Mod. NSE	Performance Rating[a]	RSR	Performance Rating[a]	Mod. RSR	Performance Rating[a]	PBIAS	Performance Rating[a]
Calibration Runoff	0.52	Satisfactory	0.81	Very Good	0.70	Satisfactory	0.43	Very Good	-23	Satisfactory
Calibration Sed. Yield	0.49	Unsatisfactory	0.89	Very Good	0.72	Unsatisfactory	0.34	Very Good	-24	Satisfactory
GW1 Runoff	0.68	Good	0.90	Very Good	0.56	Good	0.28	Very Good	15	Good
GW1 Sediment Yield	0.63	Satisfactory	0.86	Very Good	0.61	Satisfactory	0.38	Very Good	-11	Very Good
GW2 Runoff	0.76	Very Good	0.99	Very Good	0.49	Very Good	0.12	Very Good	-2	Very Good
GW2 Sediment Yield	0.32	Unsatisfactory	0.86	Very Good	0.83	Unsatisfactory	0.38	Very Good	16.	Good

[a] Value ranges for performance ratings were provided by Moriasi et al. (2007)

CHAPTER 4

RAINFALL SIMULATION EXPERIMENTS ON NATURAL GAS WELL PAD SITES: DEVELOPING EROSION PARAMETERS FOR WEPP

Introduction

In North Central Texas, natural gas well development was shown to substantially increase sediment in storm water runoff compared to pre-development site conditions (Chapter 1). Other areas of the United States are also experiencing natural gas development pressures. Because characteristics related to runoff and erosion, such as rainfall, topography, soil, and vegetation, can vary drastically from region to region, it is important to consider local site characteristics in runoff and sediment evaluations. Models are often used to characterize runoff and sediment in lieu of expensive and resource intensive storm water monitoring programs. The water erosion prediction project (WEPP) has been used extensively for runoff and erosion prediction from agricultural fields, rangelands, small watersheds, forests, and to a lesser extent, construction sites.

Chapter 2 evaluated the use of WEPP for predicting sediment yields specifically from natural gas well sites. The model was shown to be adequate for this purpose; however, it was suggested that rainfall simulations be used to improve the evaluation of WEPP parameters for gas well sites. Model parameterization, calibration, and validation are difficult when multiple overland flow elements containing different soil, management, and slope characteristics potentially influence the validation data. Rainfall simulator experiments are useful because they can quickly produce large amounts of data while controlling initial conditions and inputs to the system (Simanton et al., 1991; Alberts et al., 1995).

Development of parameters for the WEPP model relied heavily on rainfall simulation studies. Elliot et al. (1989) conducted rainfall simulations studies on numerous soil types across the United States to develop soil erodibility parameters. Similarly, Liebenow et al. (1990) used rainfall simulations to evaluate interrill erodibility properties for a broad range of cropland soils and presented a method for accounting for slope in the interrill erodibility process. The Interagency Rangeland Water Erosion Team (IRWET) developed the most extensive database of rainfall simulator data, which was used to improve parameter estimation procedures for the rangeland infiltration and erosion components of the WEPP model (Franks et al., 1998). Rainfall simulation studies were also used by Simanton et al. (1991) to evaluate the direct effect of vegetation canopy cover on runoff, infiltration, and erosion and develop associated WEPP parameters. Rainfall simulation studies have also been used to develop parameters for road erosion and evaluate WEPP's ability to prediction erosion from forest roads (Elliot 1995; Foltz and Elliot, 1996).

The objectives of this research were two-fold. First, rainfall simulations were used to measure runoff and sediment from research plots located on natural gas well pad sites and a nearby rangeland site. For the purposes of this research, the rangeland site was used as a "reference" to represent pre-development site conditions. Data were compared to evaluate similarities and/or differences among gas well site plots and between gas well sites. Differences between gas well sites and the reference site were also evaluated. Second, sediment data from gas well pad sites were used to develop interrill erodibility parameters for the WEPP hillslope model and, using these parameters, determine the suitability of WEPP for estimating erosion from natural gas well pad sites.

Methods

Rainfall Simulations

Research plots were constructed at two gas well pad sites (Site 2 and Site 3) and one reference site (Site 3R) in Denton County, Texas (Figure 4.1). Gas well pad sites were constructed on soil classified as Medlin stony clay (fine, montmorillonitic, thermic, vertisols) on 5 to 12% slopes, which generally has rapid runoff and severe erosion (USDA, 1980). The reference site was located on Blackland prairie, with soil also classified as Medlin stony clay. Construction of pad sites required leveling the hillslope to accommodate a relatively flat surface area approximately 0.5 ha in size. The leveled surface area was compacted and covered with an all-weather surface of Grade 1 Flex Base (crushed limestone) approximately 0.3 m in depth. The region receives an average annual rainfall of 990 mm (38.9 in) with April, May, September, and October as the wettest months. For two weeks prior to the first simulated storm event, the research sites received no natural rainfall. Daily average and maximum temperatures were 29.9° C (85.8° F) and 36.6° C (97.8° F), respectively. During the simulation period (14-Aug-07 through 28-Sep-07), the research sites received 83.7 mm of natural rainfall. Simulations were not conducted within one week of a natural rainfall event. Deep cracks were observed in the soil surrounding the pad sites and at the reference site (Figure 4.2). Cracking is typical in clay soils in this region during dry conditions.

A total of nine research plots were constructed, three plots at each gas well site and three plots at the reference site. Plots were sized to fit the effective rainfall distribution from the simulator, 1.5 m wide by 4.5 m long, for a total plot area of 6.75 m^2 (Figure 4.3). Plot slopes ranged between 0.7 and 2.2% at the gas well pad sites and between 2.3 and 3.0% at the reference site. The top and sides of each plot were bordered with a 0.5 m high flexible impermeable material to delineate runoff from plot surroundings. The flexible border was used to because it created a tighter seal with aggregate compared to a metal border. The toe of the border was set in a 0.2 m deep trench to prevent lateral flow and the inside edge of the border was filled with a bentonite slurry to minimize vertical flow along the edge of the border. A PVC collection trough at the bottom of the plots directed all runoff to a collection point for sampling. Care during plot construction resulted in minimal disturbance to the plot area.

Simulated rainfall was applied with a Norton Rainfall Simulator consisting of four 80100 VeeJet oscillating nozzles spaced 1.37 m apart at a height of 2.5 m. During simulated rain events, nozzle water was maintained at 0.42 kg cm^{-2} (6 psi) producing an intensity of 58.7 mm h^{-1}. This intensity is nearly equivalent of a one-hour storm event for a 5-year return for the Denton County area (Hershfield, 1961). Following a typical WEPP sequence (Holland, 1969), rainfall was applied in a series of three consecutive events: a dry run on existing soil moisture conditions, followed by a wet run 24 hours later, followed by a very wet run 30 minutes later. During the very wet run three intensities were applied in a sequence of 58.7, 104.2, and then back to 58.7 mm h^{-1}.

Data collection procedures were similar to those described in Liebenow et al. (1990). Runoff water samples were collected at the start of runoff and then taken every five minutes until steady-state runoff was achieved (three or more samples at a consistent rate). Measurements of the sample volume and the time required to collect the samples were used to determine the runoff rates. Total Suspended Solids (TSS) analysis was conducted on each runoff sample according to procedures outlined in Standard Methods (APHA, 1992).

The data reduction technique used in the study was also similar to that of Liebenow et al. (1990). The last 15 min. of sampling (or last three samples) were used to obtain steady-state conditions for runoff and erosion rates. Runoff rates were converted into a depth per unit time based on a density of 1000 kg m^{-3} and the sampled plot area of 6.75 m^2.

Statistical analysis was performed using SAS version 9.3 (SAS, 2006). A nested analysis of variance (PROC NESTED) was used to determine significant differences between the two gas well sites and among the plots within a site for the runoff and erosion rates. Reference plots did not generate adequate runoff to analyze statistically. An analysis of variance (PROC ANOVA) was used to analyze differences among run types (dry, wet, very wet) for the gas well site plots. Significant differences were established at the 0.05 probability level.

WEPP Simulations

WEPP is a process-based, distributed parameter, continuous simulation model based on fundamentals of stochastic weather generation, infiltration theory, hydrology, soil physics, plant science, hydraulics, and erosion mechanics (Flanagan et al., 1995). Infiltration is calculated using the Green Ampt Mein Larson (GAML) model (Mein and Larson, 1973; Chu, 1978) for unsteady rainfall. Runoff, the difference between the rainfall and infiltration, is routed overland using a semi-analytical solution of the kinematic wave model (Stone et al., 1992). WEPP's erosion component uses a steady-state sediment continuity equation that considers both interrill and rill erosion processes. Interrill erosion involves soil detachment and transport by raindrops and shallow sheet flow, while rill erosion processes describe soil detachment, transport, and deposition in rill channels (Flanagan and Nearing, 1995). WEPP's processes are summarized by Laflen et al. (1991), and its applications are discussed in Laflen et al. (1997). WEPP's model documentation provides a detailed discussion for all major processes (Flanagan and Nearing, 1995). Major inputs are included in the climate, slope, management, and soil input files.

WEPP version 2006.5 was used for all simulations. A single storm climate file was specified for each rainfall event to have an average intensity and duration equal to the simulated rainfall. WEPP's default soil file for a "graveled road surface on clay loam" was edited to represent gas well pad site conditions. Important soil properties include effective hydraulic conductivity, interrill erodibility, rill erodibility, critical shear, initial saturation level, and soil layer characteristics.

Effective hydraulic conductivity was manually adjusted until the predicted runoff was approximately equal to the observed runoff. Since no rilling on pad sites had been observed under natural rainfall conditions and the plots were relatively small, erosion on pad site plots for this study was assumed to be dominated by interrill erosion. Kinnell and Cummings (1993) developed equation 1:

$$D_i = K_i I q S_f \qquad (1)$$

to describe interrill erosion, which was a modification of the empirical relationship described in Liebenow et al. (1990):

$$D_i = K_i I^2 S_f \qquad (2)$$

where

D_i = steady-state interrill erosion rate (mass of soil eroded/unit area/unit time)

K_i = interrill erodibility (mass-time/length^4)

I = rainfall intensity (depth per unit time)

q = steady-state flow discharge (depth per unit time)

$S_f = 1.05 - 0.85 \exp(-4\sin\theta)$, where θ = slope angle (unitless).

Observed steady-state TSS concentrations were converted into the interrill erosion rate (D_i) and then to interrill erodibility values (K_i) using Equation 1.

Default rill erodibility (K_r = 0.0002 sec m^{-1}) and critical shear (τ = 10 Pa) values provided in WEPP's "gravel road surface on clay loam" were used for all simulations. Soil layer characteristics and initial saturation levels were adjusted to represent plot conditions according to bulk density tests and particles size analyses. Initial saturation levels used in the model were 50, 80, and 90% for the dry, wet, and very wet runs, respectively. For all plots, percent sand was changed to 10% and percent clay to 65% according to particles size analyses conducted on sediment collected from weirs used to measure runoff from the same gas well sites (Havens, 2007).

Default values for the WEPP "Insloped road-unrutted, forest" were used in the management file except for the bulk density parameter. A bulk density value of 1.4 g cm^{-3} was used instead, which was based on the average of six bulk density samples collected at the two gas well pad sites. The observed slopes of each plot were described in each slope file.

A total of 18 WEPP runs were conducted; a dry, wet, and very wet run for each gas well site plot. Model runs were not conducted for reference site plots due to the lack of data generated during the rainfall simulations. Predicted and observed total erosion in t ha^{-1} were evaluated using the Nash-Sutcliffe efficiency (NSE), root mean square error (RMSE)-observation standard deviation ratio (RSR), and percent bias (PBIAS). These statistics are described in detail by Moriasi et al. (2007).

WEPP was also run in continuous simulation to predict average annual sediment yields in mass by area by year, which is useful for comparison to other studies. WEPP runs were conducted for each plot using average k_e and k_i values of the dry, wet, and very wet runs. WEPP's output is the average annual sediment yield in t ha^{-1} yr^{-1} based on a 30-year simulation period.

Results

Dry Run Simulations

Figure 4.4 shows dry run runoff and TSS concentrations for all plots at Sites 2 and 3; no runoff occurred on any of the reference site plots. Time to runoff and summary statistics for steady-state runoff, TSS concentrations, and interrill erosion are shown in Table 4.1. Average time to runoff at Site 2 was 4.55 min. compared to 5.21 min. at Site 3. Runoff neared steady-state within 10 to 15 min. of initial runoff. Mean steady-state runoff at Site 2 and Site 3 was 30.2 and 27.3 mm h^{-1}, respectively. Steady-state runoff was significantly different among plots within the sites ($p<0.0001$), but Sites 2 and 3 were not significantly different from each other ($p=0.4530$).

For all plots, TSS concentrations were highest in the first sample and neared steady-state conditions 5 minutes after runoff started. This "wash off" effect of loose sediment is typical in storm water runoff. The mean steady-state TSS concentration at Site 2 was 3,238 mg l^{-1} compared to 2,316 mg l^{-1} at Site 3. TSS concentrations were significantly different among plots within the sites (p=0.0024) and were also significantly different between the two sites (p=0.0425).

Wet Run Simulations

Figure 4.5 shows wet run runoff and TSS concentrations for all plots at gas well Sites 2 and 3 and Plot 1 at the reference site (Site 3R). Time to runoff and summary statistics for steady-state runoff, TSS concentrations, and interrill erosion are shown in Table 4.2. Average time to runoff was 2.57 min. at both gas well sites. Time to runoff at reference site Plot 1 was much greater at 29.0 min. S-state runoff was reached faster during the wet run compared to the dry run; approximately 5 to 10 minutes after runoff started. Mean steady-state runoff at Site 2 (33.6 mm h-1) and Site 3 (37.8 mm h-1) were greater for wet runs compared to the dry runs. Steady-state runoff was significantly different among plots within the sites (p=0.0077), but sites were not different from each other (p=0.0744). Steady-state runoff differences between gas well site plots and the reference plot were very large; steady state runoff at Site 3R was 8.58 mm h^{-1}.

Mean steady-state TSS concentrations at Sites 2 and 3 were 3,567 and 2,310 mg l^{-1}, respectively. TSS concentrations were significantly different among plots within the sites (p<0.0001), but the sites were not significantly different from each other (p=0.1331). The mean steady-state TSS concentration at the reference site plot was only 52.3 mg l^{-1}.

Very Wet Run Simulations

Figure 4.6 shows runoff and TSS concentrations for all plots at Sites 2 and 3 and Plot 1 at the reference site for the first 58 mm h^{-1} rainfall application of the very wet run. Time to runoff and summary statistics for steady-state runoff, TSS concentrations, and interrill erosion are shown in Table 4.3. Average time to runoff at Site 2 and Site 3 was 1.9 and 1.43 min., respectively. Time to runoff at reference site plot 1 was 21.0 min. Steady-state runoff was reached faster during the very wet run compared to both the dry and wet runs; approximately 5 min. after runoff started at the gas well plots and 10 min. at the reference site plot. Mean steady-state runoff at Site 2 (36.4 mm h^{-1}) and Site 3 (38.3 mm $h\text{-}^{1}$) for the very wet runs was greater than the wet runs. Steady-state runoff was significantly different among plots within the sites ($p=0.0174$), but sites were not different from each other ($p=0.3365$). Steady-state runoff at the reference site plot was only 16.7 mm h^{-1}.

Mean steady-state TSS concentrations were less than wet run concentrations at Sites 2 and 3 and were 3,460 and 2,037 mg l^{-1}, respectively. Wet run TSS concentrations were significantly different among plots within the sites ($p<0.0001$), but Site 2 and Site 3 were not significantly different ($p=0.1437$). The mean steady-state TSS concentration at the reference site plot was 17.7 mg l^{-1}.

Figure 4.7 shows the infiltration rate and TSS concentrations for the entire sequence of rainfall intensities applied during the wet run. The infiltration rate illustrates processes not evident in the runoff rate. For example, an increase in steady-state infiltration rate is evident when the rainfall intensity was increased from 58.7 to 104.2 mm h^{-1}. This type of response is called partial area contribution. Hawkins (1982) suggested that this occurs because there is a distribution of infiltration capacities within the plot due to variability in soil properties. As rainfall intensity increases, more of the total area begins to contribute; however, the “new” areas may have higher infiltration capacities and thus cause an increase in the apparent infiltration rate. TSS concentrations increased slightly when the rainfall intensity was increased, but then slowly decreased to near steady-state conditions of the first 58.7 mm-h^{-1} rainfall application rate.

Dry, Wet, and Very Wet Run Comparisons

Steady-state runoff and TSS standard deviations were relatively small for all plots indicating steady-state conditions were fairly consistent for both runoff and TSS concentrations. Time to steady-state runoff decreased from the dry to wet to very wet runs. Steady-state runoff conditions were significantly different among run types ($p<0.0001$) and multiple comparisons tests (*Student–Newman–Keuls*) indicated that the dry run steady-state runoff values were different than wet and very wet runs. However, the wet and very wet runs were not significantly different from each other. TSS trends were similar among all run types (dry, wet, and very wet), decreasing relatively quickly to a steady-state condition. TSS concentrations were not significantly different among run types ($p=0.8355$).

Modeling Results

Calibrated effective hydraulic conductivity (K_{ef}) values and data derived interrill erodibility values (K_i) are shown in Table 4.4. The results of the observed and predicted sediment yields are shown in table 5. Observed sediment yields ranged from 0.202 to 0.701 t ha^{-1}. Model evaluation statistics for NSE, RSR, and BBIAS were 0.9, 0.3, and 13.7, respectively. These three evaluation statistics are all considered to be “very good” according to recommended guidelines for performance ratings provided by Moriasi et al. (2007). A PBIAS value of 13.7 indicates a slight model underprediction. Graphical evaluation results, shown in Figure 4.8, were also in agreement with the statistical results. A general visual agreement between measured and predicted data indicates adequate model performance over the range of constituents being simulated (Singh et al., 2004).

Discussion

Rainfall Simulations Runoff and Sediment

Research has shown that gravel alters the hydraulic conductivity of a soil (Foltz and Truebe 1995). Flerchinger and Watts (1987) found that, generally, the addition of gravel increases the porosity and increases the hydraulic conductivity of the road, which decreases the runoff. In contrast to this finding, runoff was higher (i.e. hydraulic conductivity lower) on the gravel gas well pad sites compared to the reference site. However, this was not surprising since the soil was cracked at the reference site. The average steady-state infiltration rate for gas well site plots for the very wet run was 21.5 mm h^{-1} compared to 49.9 mm h^{-1} for the very wet run at the reference site. This result may have been due to time of the year the study was conducted; the soil was dry and cracks in the soil were visible, which likely resulted in slower, or non-existent runoff at the reference site plots. Also, hydraulic conductivity values for rangeland soils vary considerably as observed hydraulic conductivity values for tall grass rangeland on clay soils ranged from 18 to 75 mm h^{-1} (Franks et al., 1998).

Average steady-state sediment production from gas well site plots was 22.6 ($\pm$2.1), 28.9 ($\pm$3.1), and 28.7 ($\pm$3.7) mg m^{-2} sec for the dry, wet and very wet runs, respectively. On research plots (33% slope) representing post-construction site conditions (bare, compacted soil) steady-state interrill erosion was 120 ($\pm$98) mg m^{-2} sec (Persyn et al., 2004). In this comparison, sediment production from gas well sites pads appears to be less that sediment production from a typical post-construction site condition. While sediment yields at gas well sites were previously found to similar to those observed at construction sites (Chapter 1 and Chapter 2), this finding suggests that the disturbed area around the site may contribute a greater portion of the total sediment yield compared to the pad itself.

Total sediment yield at the reference site was 0.45 kg ha^{-1} mm^{-1} of runoff, which is much less than sediment yield observed from rangeland plots in other studies. Simanton et al. (1991) and Franks et al. (1998) reported sediment yields that ranged from a few to nearly 160 kg ha^{-1} mm^{-1} of runoff. The reference site plots at this study were smaller, had less than 5% bare soil, had very dense vegetation that had not been clipped, and had not been recently grazed whereas research plots in these other studies were larger, had varying proportions of bare soil and vegetation, and had some degree of recent grazing.

WEPP Modeling

Calibrated effective hydraulic conductivity values (Table 4.4) for gas well site plots were much higher than the default value (16.6 mm h^{-1}) of WEPP's "graveled road surface on clay loam" soil file. Calibrated effective hydraulic conductivity values were also much higher than those reported for gas well pad surfaces in Chapter 2 (0.1 mm h^{-1}) and by Foltz and Elliot (1996) for graveled roads (2 mm h^{-1}). There are three possible explanations for these differences. First, rainfall simulations were conducted during the dry time of year when cracking was evident in soils surrounding gas well pad sites. It is possible the soils beneath the gas well sites were also cracked, which could greatly increase the hydraulic conductivity. The majority of storm water runoff data used to evaluate WEPP in Chapter 2 were collected during wetter times of the year in the spring and fall when soil cracking would be less likely. Second, Medlin stony clays soils have limestone rock strata at each 10 to 20 feet change in elevation (USDA, 1980). These strata may have been exposed at the cut slope when the sites were constructed resulting in soil lenses where water that infiltrates the soil upslope could ex-filtrate at the cut slope. This additional water running onto the pad and eventually through the monitoring weir would contribute to a greater measured runoff volume and thus the hydraulic conductivity value would have to be lowered in the model to account for this additional runoff. This phenomenon appears to be evident in rainfall/runoff hydrographs as runoff continues after rainfall has ceased. Third, the hydraulic conductivity of the pad sites could be a function of rainfall intensity as shown in Figure 4.7. Based on the relationship illustrated by Figure 4.7, for lower intensity rainfalls the hydraulic conductivity could be less.

Interrill erodibility values derived from research plots used in the modeling analyses (Table 4.4) were quite variable but were comparable to WEPP's default values (1,000,000 kg sec m^4). Annual average WEPP predictions for gas well plot sites using interrill erodibility values derived from the research plots ranged from 5 to 11 t ha^{-1} yr^{-1}, with an average of 7.4 t ha^{-1} yr^{-1}. Foltz and Elliot (1996) measured sediment in runoff from 61 m long by 4.27 m wide forest road segments covered with low quality aggregate (higher quantities of fine materials). This type of aggregate is similar to the type of aggregate used to construct gas well pads. The three segments were treated with three different tire pressures of logging trucks. Measured average sediment yield in their study were similar to gas well site plots ranging from 6.8 to 34.3 t ha^{-1} yr^{-1} on the three segments. Using parameters derived from rainfall simulations conducted on road segments, Foltz and Elliot (1996) estimated k_e and ki values of 2.0 mm h^{-1} and 3,000,000 kg sec m^4, respectively. Using these values, WEPP predicted average annual sediment yields of 8.7 and 45.5 t ha^{-1} yr^{-1}. Differences between gas well pad sites and graveled road yields could be attributed to the amount of armoring that had occurred prior to each study being conducted. Armoring is the process of wind and water erosion removing the fine material from the surface over time. Foltz and Elliot (1996) conducted their study immediately after constructing the road segments whereas gas well sites had been constructed almost three years prior to this study.

Chapter 1 estimated annual sediment yields ranging of 41 and 28 t ha^{-1} yr^{-1} for Site 2 and Site 3, respectively. In Chapter 2, WEPP predicted annual sediment yields were 38 t ha^{-1} yr^{-1} for Site 2 and 21 t ha^{-1} yr^{-1} for Site 3. In these chapters, sediment was contributed from both the cut slopes and the pad surface at these sites. In this study predicted annual sediment yield from the pad sites averaged 7.4 t ha^{-1} yr^{-1} indicating that a smaller portion of total sediment yield is contributed from the pad area of the site. This is an important finding because best management practices targeted to reduce erosion and sedimentation from the disturbed portions of the site could be more effective in minimizing total sediment yield from these sites.

Conclusion

Rainfall simulations were conducted on natural gas well pad site and rangeland (reference site) research plots in North Central Texas. Rainfall was applied at a rate of 58 mm h^{-1} to dry, wet, and very wet soil conditions. Steady-state runoff was significantly different among gas well pad plots for each run (dry, wet, and very wet), but sites were not significantly different from each other for any of the runs. Steady-state sediment concentrations were also significantly different between Sites 2 and 3 for the dry run, but sites were not different from each other for the wet and very wet runs. For all plots combined, the dry run steady-state runoff rate was significantly different than wet and very wet run runoff, but runoff was not different between the wet and very wet runs. Steady-state sediment concentrations were not significantly different among run types. Runoff only occurred at one reference site plot; both steady-state runoff and sediment concentrations were substantially less at the reference site compared to runoff and sediment from the gas well plots.

Interrill erodibility parameters were derived from runoff and sediment data collected from the rainfall simulations. Measured hydraulic conductivity values (25-70 mm hr^{-1}) were higher than the WEPP default of 16.6 mm h^{-1}.
Interrill erodibility parameters for gas well sites ranged from 443,746 to 1,123,131 kg sec m^4 compared to the WEPP default value of 1,000,000. These parameters should be adjusted accordingly to accurately simulate runoff and erosion from natural gas well pad sites.

Rainfall simulations were modeled with WEPP using calibrated effective hydraulic conductivity and data derived interrill erodibility parameters. Model predictions were evaluated with NSE, RSR, and PBIAS statistics. NSE, RSR, and PBIAS values for sediment yield predictions were 0.9, 0.3, and 13.2, respectively, which are all considered “very good” according to recommended rating guidelines. These results suggest that WEPP can effectively model sediment yield from natural gas well pad sites.

Using model parameters calibrated and derived from this study, WEPP predicted an average annual sediment yield for gas well pads of 7.4 t ha^{-1} yr^{-1}. Initially, pad sites may contribute more sediment yield on an annual basis and may decrease over time as the site becomes more armored from wind and water erosion. This value represents less than half of the total sediment yield from gas well sites previously estimated and modeled in Chapters 1 and 2. Therefore, erosion and sediment control measures that target the surrounding disturbed area, such as the cut and fill slopes would be expected to be more effective for reducing the overall sediment yield from these sites.

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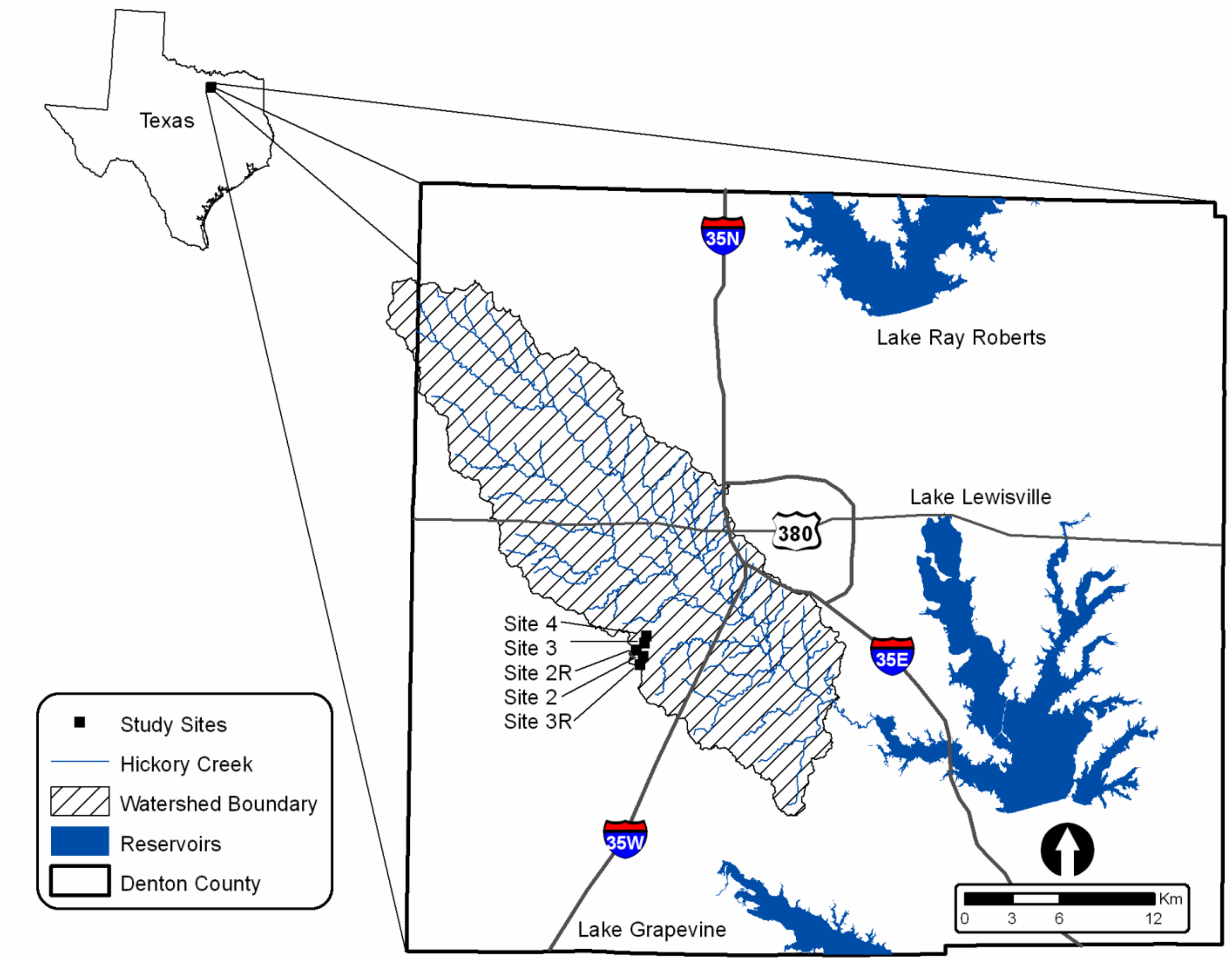


Figure 4.1. Study area - Denton County, Texas.

Figure 4.2. Cracks in the soil at the reference site.

Figure 4.3. Gas well pad site research plot.

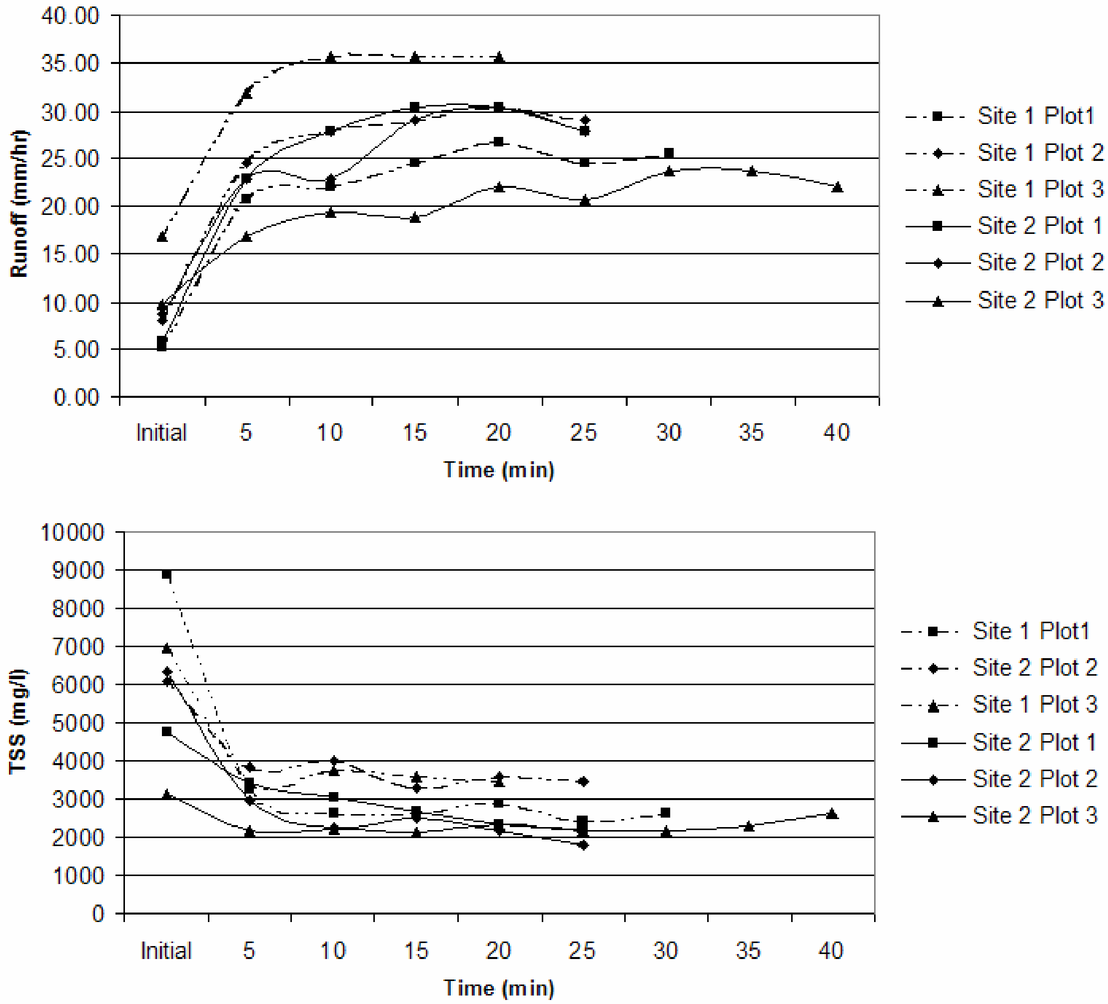
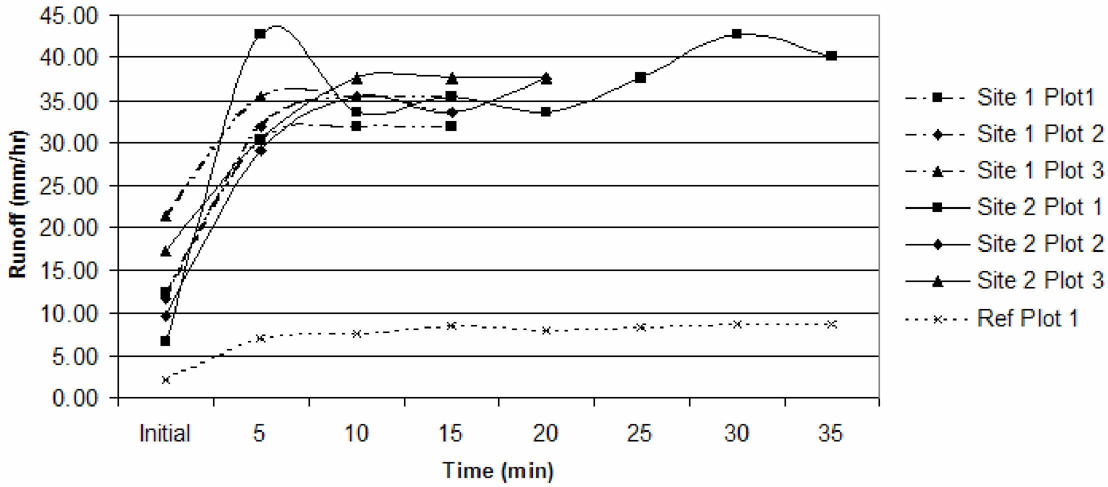


Figure 4.4. Dry run runoff and TSS.



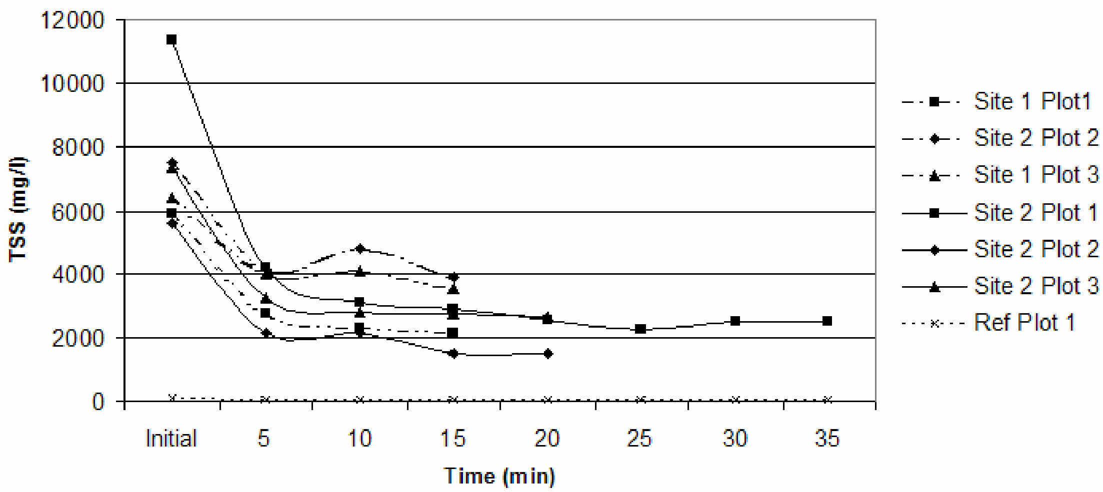


Figure 4.5. Wet run runoff and TSS.

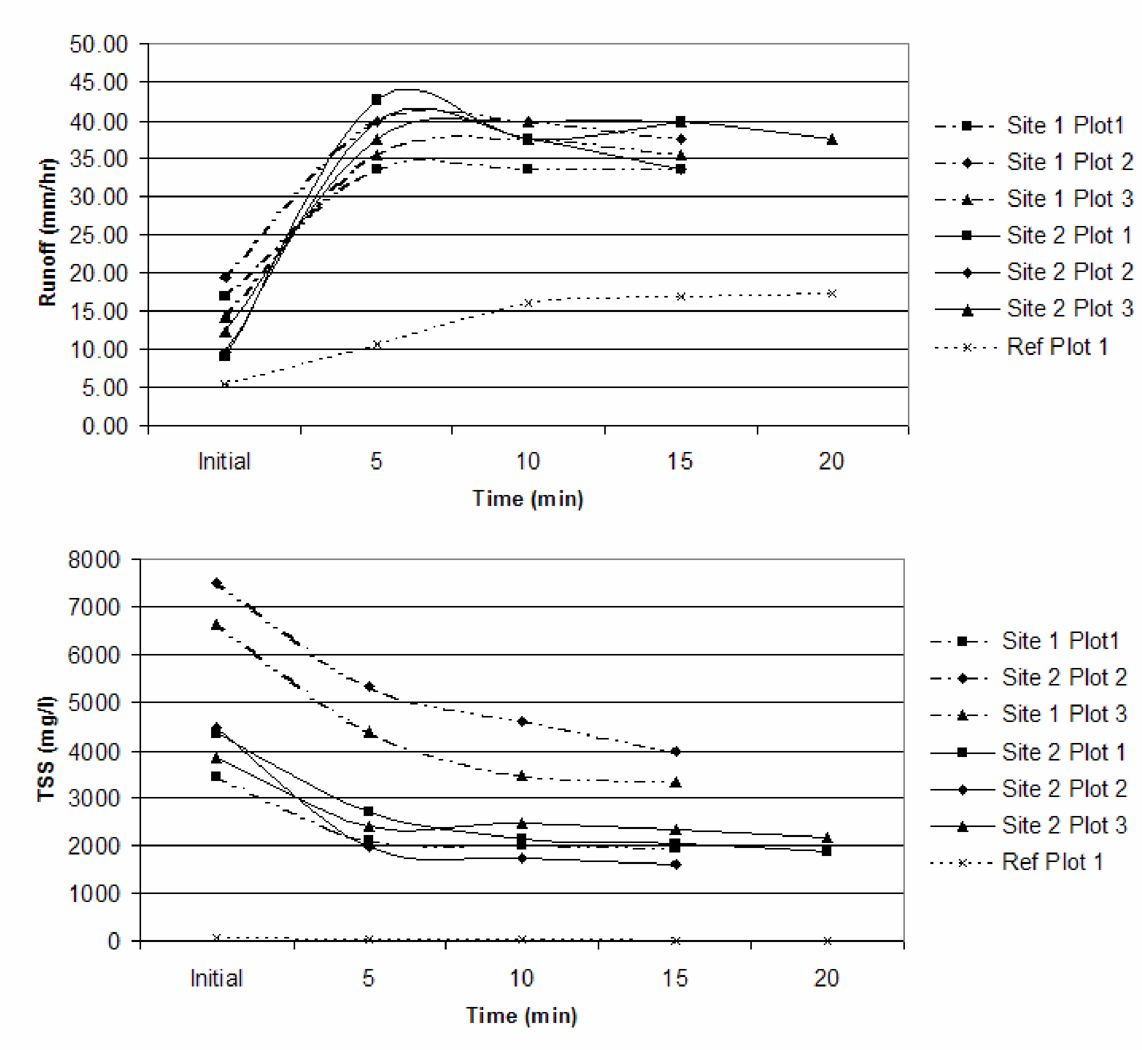


Figure 4.6. Very wet run runoff and TSS.

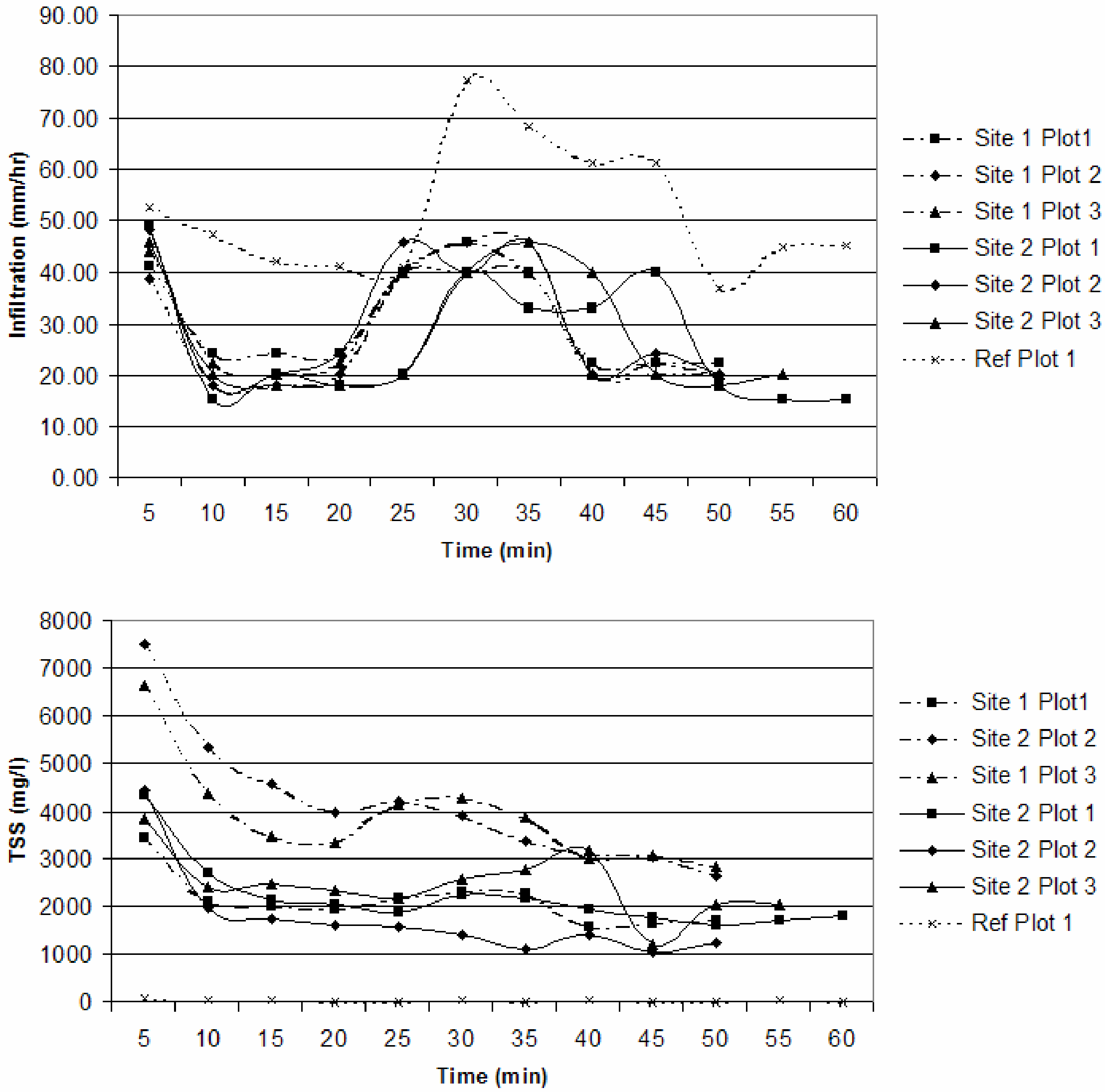


Figure 4.7. Very wet run increased rainfall intensity infiltration rate and TSS.

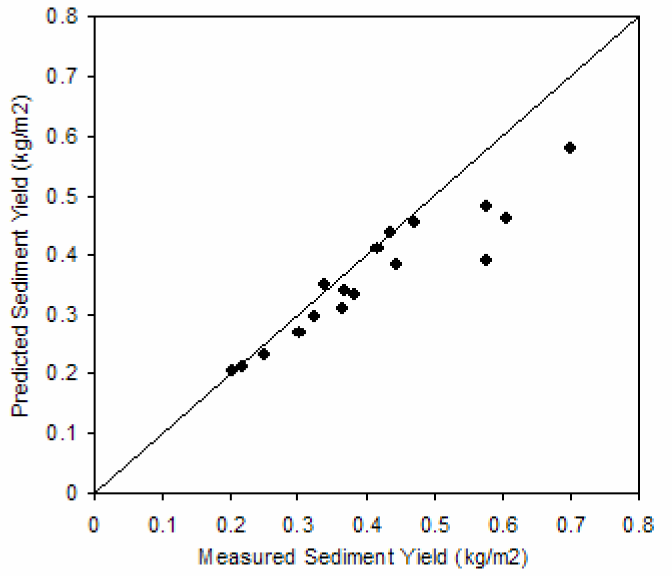


Figure 4.8. Scatterplot of observed and predicted sediment yield table.

Table 4.1. Dry Run Runoff and Sediment Characteristics

Site	Plot	Time to Runoff	Mean Steady-state Runoff (mm h^{-1})	Mean Steady-state Sed. Conc. (mg l^{-1})	Mean Steady-state Interrill Erosion (mg/m^2 sec)
GW #2	1	4.52	25.6 ± 1.03	2637.0 ± 222.99	18.8 ± 2.3
	2	3.75	29.6 ± 0.8	3472.3 ± 148.0	28.5 ± 1.9
	3	5.37	35.6 ± 0	3604.3 ± 144.04	35.6 ± 1.4
GW #3	1	4.75	29.6 ± 1.53	2416.0 ± 239.63	19.9 ± 2.7
	2	6.13	29.1 ± 1.33	2153.7 ± 365.0	17.5 ± 3.3
	3	4.72	23.2 ± 0.94	2378.3 ± 237.4	15.3 ± 0.9

Table 4.2. Wet Run Runoff and Sediment Characteristics

Site	Plot	Time to Runoff	Mean Steady-state Runoff (mm h^{-1})	Mean Steady-state Sed. Conc. (mg l^{-1})	Mean Steady-state Interrill Erosion (mg/m^2 sec)
GW #2	1	1.82	31.5 ± 0.88	2504.3 ± 315.27	21.0 ± 2.2
	2	2.47	33.7 ± 1.78	4300.0 ± 460.95	40.4 ± 6.1
	3	3.38	35.5 ± 0	3895.3 ± 278.64	38.5 ± 2.8
GW #3	1	2.37	40.1 ± 2.51	2430.3 ± 130.4	27.1 ± 3.0
	2	3.43	35.6 ± 1.98	1735.3 ± 374.7	17.2 ± 3.8
	3	2.0	37.6 ± 0	2763 ± 79.2	28.9 ± 0.8
Ref	1	29.0	8.58 ± 0.24	52.3 ± 9.71	0.12 ± 0.02

Table 4.3. Very Wet Run Runoff and Sediment Characteristics (first 58 mm h^{-1} rainfall application rate)

Site	Plot	Time to Runoff	Mean Steady-state Runoff (mm h^{-1})	Mean Steady-state Sed. Conc. (mg l^{-1})	Mean Steady-state Interrill Erosion (mg/m^2 sec)
GW #2	1	2.30	33.7 ± 0	2012.0 ± 82.5	18.8 ± 0.7
	2	1.9	39.2 ± 1.36	4627.7 ± 660.39	50.6 ± 8.6
	3	1.5	36.3 ± 1.21	3742.6 ± 560.48	37.6 ± 5.2
GW #3	1	1.15	38.4 ± 1.36	2014.0 ± 129.15	21.5 ±1.6
	2	1.78	37.1 ± 3.19	1775.3 ± 190.07	18.4 ± 3.5
	3	1.35	39.2 ± 1.36	2322.7 ± 155.57	25.3 ± 2.5
Ref	1	21.0	16.7 ± 0.69	17.7 ± 3.79	0.08 ± 0.01

Table 4.4. Slope, Effective Hydraulic Conductivity, and Interrill Erodibility

Site	Plot	Slope (%)	Dry Run		Wet Run		Very Wet Run	
			K_{ef} (mm h^{-1})	K_i (kg sec/m^4)	K_{ef} (mm h^{-1})	K_i (kg sec/m^4)	K_{ef} (mm h^{-1})	K_i (kg sec/m^4)
GW #2	1	0.6	68	734583	29	670048	37	560478
	2	1.6	49	842732	29	1043606	34	1123131
	3	1.9	41	843045	28	911109	30	875400
GW #3	1	.7	55	663042	31	666975	27	552718
	2	1.2	60	550717	35	443745	29	453974
	3	2.2	77	537061	28	623849	25	524491

Table 4.5. Observed and Predicted Sediment Yield (tonnes per hectare)

Site	Run	Plot	Observed (t ha^{-1})	Predicted (t ha^{-1})
GW #2	Dry	1	0.367	0.340
		2	0.470	0.453
		3	0.606	0.460
	Wet	1	0.250	0.229
		2	0.436	0.438
		3	0.576	0.391
	Very Wet	1	0.217	0.211
		2	0.576	0.482
		3	0.417	0.411
GW #3	Dry	1	0.338	0.349
		2	0.303	0.269
		3	0.383	0.333
	Wet	1	0.701	0.580
		2	0.251	0.229
		3	0.444	0.384
	Very Wet	1	0.322	0.296
		2	0.202	0.202
		3	0.365	0.309

CHAPTER 5

MODELING EROSION AND SEDIMENT CONTROL PRACTICES WITH RUSLE 2.0: MANAGEMENT APPROACH FOR NATURAL GAS WELL SITES IN DENTON COUNTY, TEXAS, USA*

Introduction

Nonpoint source pollution from agricultural and urban runoff is the leading source of impairment to the Nation's waters, and sediment is the single most widespread pollutant affecting the water quality in rivers and streams (USEPA 2000). Initiated by soil erosion, sedimentation is the result of soil and other particulates settling out of the water column and depositing in streams, rivers, lakes, and other water bodies. Although the movement of sediment into water bodies is a natural process, its rate and severity can be amplified by land disturbing construction activities.

Erosion rates from construction sites can be up to 40,000 times greater than undisturbed conditions (Wolman and Schick 1967) and in the United States may result in up to 5 billion tons of sediment reaching receiving streams each year (Willet 1980). Sediment washed from construction sites results in both local and cumulative downstream impacts from suspended and deposited sediment (Harbor 1999). Eroded sediment from construction sites can also transport associated fertilizers, pesticides, fuels, and other contaminants and substances commonly spilled at construction sites (Risse and Faucette 2001). In developing areas, construction activities are by far the leading source of sediment with sediment yields ranging from a few tonnes to over 1000 tonnes per hectare per year (t/ha/yr) (USEPA 2002). Regardless of the type of construction activity (i.e. residential, commercial, or highway), construction projects where topsoil is disturbed or cleared of vegetation are particularly subject to erosion problems (Faucette et al. 2006).

Natural gas exploration and production is a type of land disturbing activity that requires construction of well sites, access roads, and pipelines. These construction activities have the potential to accelerate soil loss due to land cover disturbance, increased slopes, and flow concentration. In 2006, almost 30,000 natural gas wells were drilled nationwide (API 2007), which is a substantial number considering that each well site disturbs approximately 1 to 2 ha of land surface (Wachal et al. 2007). While typical construction activities are regulated by the federal national pollutant discharge elimination system (NPDES) program - which requires erosion and sediment control best management practices (BMPs), storm water pollution prevention plans, and increased monitoring and site inspections - oil and gas field operations and construction activities are exempt from federal NPDES permitting requirements (USEPA 2006).

Because gas well sites have the potential to impact water quality in receiving water bodies, it is important to understand how these sites can be managed with erosion and sediment control best management practices (BMPs) to minimize potential impacts. In lieu of resource intensive monitoring, computer modeling provides a cost effective, practical way to objectively assess various methods that are available to reduce off-site sediment movement from land disturbance and construction activities (McClintock and Harbor 1995). In this context, previous researchers have used ANSWERS (Dillaha et al. 1982), SERS (Ross et al. 1993), SEDCAD (McClintock and Harbor 1995; Harbor et al. 1995), WEPP (Laflen et al. 2001; Wachal and Banks 2006, Moore et al. 2007), and SED2 (Gharabaghi et al. 2006a) to analyze erosion, sediment transport, and BMP effects. This paper used version 2 of the revised universal soil loss equation (RUSLE 2.0) to analyze erosion and the use of BMPs specifically for natural gas well sites.

The objectives of this research were to (1) evaluate the relative effectiveness of six BMP alternatives for natural gas well sites using RUSLE 2.0 and (2) demonstrate a practical approach for quantitatively evaluating BMP alternatives according to site-specific soil erodibility and slope conditions, site management goals, and BMP implementation costs. The RUSLE 2.0 model integrates both theoretical and empirical understanding of erosion and sediment transport processes and was designed specifically as a conservation planning tool for both agricultural and non-agricultural settings (i.e. mining and construction sites) (Foster et al. 2001). This paper illustrates a non-agricultural use of the conservation management capabilities of RUSLE 2.0 that are applicable not only to natural gas well sites, but also to other types of land disturbing construction activities.

Methods

Study Area and Site Description

The study area lies above the Barnett Shale formation in Denton County, in North Central Texas (Figure 5.1). The Barnett Shale is an organically rich geologic formation that may contain the largest onshore natural gas formation in the United States (Shirley 2002). The runoff potential for soils in the area is generally high (low infiltration capacity). The erosion hazard for surface soils ranges from low to high (erosion k factor ranging from 0.17 to 0.43), although the majority of soils in the area are moderately erodible (k = 0.32) (USDA 1980).

Moderately erodible soils tend to be located on upland prairies and have clay or clay loam surface layers. Area soils with low erodibility are usually loamy sandy soils on gently sloping upland ridges, while highly erodible soils consist of fine loamy soils located in low-lying areas near streams and valley fills. Together the sandy loams and fine loamy soils account for less than 10% of the total land area, whereas moderately erodible clay loams account for the remaining total land area. Annual normal rainfall for the region is approximately 99 cm, the majority of which normally occurs during the spring months of April through May and the fall months of September through October (USDA 1980).

Area topography tends to be flat to gently rolling. Construction of a drilling pad site on the gentle hillslopes in the region typically results in site profiles consisting of a cut slope, pad surface, and fill slope, which are approximately 60-100 m in length (Figure 5.2). The pad surface is relatively flat and is used for drilling activities. The term “cut slope” generally refers to the face of an excavated bank required to lower the ground to a desired profile. In contrast, a “fill slope” refers to a surface created by filling an area with soil. The pad is constructed with an all-weather surface of Grade 1 Flex Base approximately 0.3 m in depth. Flex Base is a gravely aggregate commonly used for temporary roads, base material underneath asphalt and concrete paving, and construction pad caps. The area of the pad surface is typically 0.5 ha, but can be much larger if multiple well heads are drilled from the same pad. Similarly, the soil on the cut and fill slopes covers an area of approximately 0.5 ha. There can be additional land disturbance surrounding the cut slope, pad surface, and fill slope depending of specific site conditions and construction practices.

RUSLE 2.0 Model Description

RUSLE 2.0 is a public domain erosion prediction tool developed and maintained by the United States Department of Agriculture - Agricultural Research Service (USDA-ARS). The model was specifically designed as a conservation planning tool to be used for a wide variety of environments and land use situations ranging from croplands to construction sites (Foster et al., 2001). RUSLE 2.0 is hybrid model that uses the empirical structure of its predecessors (USLE/RUSLE) in combination with a number of process-based erosion equations. The model applies the principal of conservation of mass, including both soil loss and deposition, to estimate sediment yields from single overland flow paths along hillslope profiles. RUSLE 2.0 was intended to be used without calibration since the model retained its fundamental empirical equation based on over 10,000 plot years of natural runoff data and 2,000 plot years of simulated runoff data (Foster et al. 2003). The model has been well validated and includes numerous process-based equations that were developed and calibrated with large data sets ranging from 10 to 30 years (Foster 2003; Foster 2005).

A comprehensive discussion of the RUSLE 2.0 equation is provided by Foster et al. (2003). RUSLE 2.0 computes net detachment on a daily time step applying a variation of the USLE computation:

$$a_i = r_i\, k_i\, l_i\, S\, c_i\, p_i \qquad (1)$$

where a is the net detachment (mass/unit area), r is the erosivity factor, k is the soil erodibility factor, l is the slope length factor, S is the slope steepness factor, c is the cover-management factor, and p is the supporting practices factor, occurring on the *ith* day. The slope steepness factor is the same every day, denoted by the upper case *S*. Deposition occurs when sediment load exceeds transport capacity and is computed by the equation:

$$D = (V_f / q)\ (T_c - g) \qquad (2)$$

where *D* is the deposition rate (mass/unit area), V_f is the fall velocity of the sediment, *q* is the runoff rate, T_c is the transport capacity of the runoff, and *g* is the sediment load (mass/ unit width). Transport capacity is determined by:

$$T_c = K_t\ q\ s \qquad (3)$$

where *s* is the sine of the slope angle and K_t is the transport coefficient computed as a function of cover management variables. Sediment load is then computed from the steady state conservation of mass equation of:

$$g_{out} = g_{in} + \Delta x\ D \qquad (4)$$

where g_{out} is the sediment load leaving the lower end of a segment on the slope, g_{in} is the sediment load entering the upper end of the segment, Δx is the length of segment, and *D* is the net detachment ("+") or deposition ("-") within the segment. The distribution of detachment is a function of soil texture and is computed for five particle classes of primary clay, primary silt, small aggregate, large aggregate, and primary sand.

The main advantage of RUSLE 2.0 over other erosion models is its ability to assess the relative effectiveness of various BMPs represented by the model's cover management practices (c factor) and support practices (p factor). For disturbed sites, cover management practices include whether or not the land is bare, mulch has been applied, or the slope has been recently reseeded. Cover management practices reduce erosion primarily by reducing the erosivity of raindrop impact and surface runoff. RUSLE 2.0 support practices generally decrease sediment yield by redirecting runoff or reducing its transport capacity and, for disturbed sites, include vegetated filter strips, fabric filter fence (silt fence), gravel bags, runoff interceptors, and small impoundments.

Modeling Analyses

RUSLE 2.0 (Version 1.26.3.0) was used to estimate sediment yields for natural gas well sites with and without erosion and sediment control BMPs. The BMPs evaluated include seeding, mulching, erosion blanket, silt fence, vegetated filter strip, and small sediment basin. Each BMP was evaluated for all possible combinations of three soil types with differing erodibility values (k-factor) and three slope profiles; therefore, nine model runs were conducted for each BMP. Erodibility values were based on the range of k-factors of Denton County soils and were classified into the following categories of low (loamy sand, K = 0.18), moderate (clay loam, K = 0.32), and high (silty clay loam, K = 0.43). Slope profiles used in the model runs were based on slopes modified for gas well sites originating from slopes of 1.8% (low), 2.9% (moderate) and 4.5% (high). These slopes profiles represent the typical slope variation for a majority of gas wells in the area. Figure 5.2 provides an example of a hillslope modified for the construction of a gas well site. Profiles representing the modified slopes were entered into RUSLE 2.0 as 9 segments for each slope. RUSLE 2.0 uses a mass balance approach to compute soil loss or deposition for each slope segment. Table 5.1 shows the slope and length of each segment for each modified slope profile. Erosion control BMPs (seeding, mulching, erosion blanket) were modeled on both the cut and fill slopes. Sediment control BMPs (silt fence, filter strip, sediment basin) were modeled at one location at the lowest point of the slope profile.

Both average annual and design storm sediment yields were modeled. Annual average sediment yields provide the best estimate for disturbed sites that are exposed for an extended period of time. Since RUSLE 2.0 is based on long-term data, long-term predictions are generally better and short-term predictions are not as good (Foster et al. 2003). However, for disturbed sites that are exposed for relatively short periods, erosion and sediment control may be more appropriately considered according to a particular design storm. In the context of erosion and sediment control, a design storm is a rainfall event of specified duration, depth, and return interval (i.e., a 24-hour storm of 99 mm has the likelihood of occurring once every 2 years) that can be used to select and size best management practices.

In this study the 1, 2, 5, and 10-year design storms, further referred to as return interval (RI) storms, were based on 24-hour duration events (Hershfield 1961). Each return interval storm was modeled on the day of the year that erosivity was likely to be the highest. Since RUSLE 2.0 is based on long-term average data, the erodibility of the environment was, by default, modeled for average conditions. Therefore, the size of the erosion event for each return interval storm would not be a "worst case" erosion event, rather it would be an erosion event based on "worst case" rainfall and average erodibility conditions for the most erosive time of year.

BMP Efficiency Rating

BMP alternatives were compared to each other according to BMP efficiency ratings.

BMP efficiency ratings provide a relative comparison among BMPs and were determined from modeled sediment yields according to Equation 5 as follows:

$$ER = (SY_{without\ BMP} - SY_{withBMP}) \ / \ SY_{without\ BMP} \qquad (5)$$

where *ER* is the efficiency rating, $SY_{without\ BMP}$ is the modeled sediment yield without any erosion or sediment control protection, and $SY_{withBMP}$ is the modeled sediment yield with erosion or sediment control protection. The ER is essentially the proportion of sediment removed by the BMP that would have otherwise left the site. For example, a BMP ER of 0.70 would mean that the BMP removed 70% of the sediment that would have left the site had the BMP not been in place.

*ER*s can be compared to site management goals to determine whether or not a particular BMP would be suitable for gas well sites. The site management goal is the measure of the acceptable level of reduced sediment yield through erosion prevention and sediment removal. For example, if the site management goal is 0.80, this means that erosion or sediment control BMPs must provide for an 80% reduction in sediment yield compared to yields expected from unprotected site conditions. If a particular BMP, or combination of BMPs, reduces sediment yield by 80%, as determined by the *ER*, then the BMP(s) is/are assumed to have accomplished the site management goal. In the North Central Texas region, a minimum management goal of 0.70 is suggested as a guideline for the adequate design of erosion and sediment control plans (NCTCOG 2003). However, the management goal may be set higher for sites located in areas that may be more sensitive to sediment pollution or to provide a margin of safety.

Results and Discussion

Construction/Disturbance without Erosion or Sediment Control

All combinations of soil erodibility conditions (low, moderate, and high) and slope steepness conditions (low, moderate, high) were modeled assuming the entire cut and fill slopes were completely disturbed and exposed to direct precipitation. The results of modeled sediment yields and *ER*s are shown in Table 5.2. Annual average sediment yields for unprotected sites ranged from 12.1 t/ha/yr (tonnes per hectare) for the low erodibility/low slope condition to 134.5 t/ha/yr for the high erodibility/high slope condition. Predicted sediment yield for the moderate erodibility/high slope condition was 85.2 t/ha/yr, which compares to 54 t/ha/yr estimated by Williams et al. (2007) for a site with similar characteristics. The predicted sediment yield value compares favorably to the estimated value since RUSLE2 is considered moderately accurate if it is within $\pm$ 50% of the true yield (USDA NRS, 2007). Modeled sediment yields were more sensitive to the slope steepness factor than to the soil erodibility factor. Sediment yields increase by about 450% as the slope conditions increased from low to high, compared to a 250% increase as the soil erodibility conditions increased from low to high.

Sediment yields from return interval storms were computed for the moderate slope/moderate erodibility condition, and ranged from 8.1 t/ha for the 1-year RI to 20.6 t/ha for the 10-year RI (Table 5.3). McClintock and Harbor (1995) used SEDCAD to model RI sediment yields from construction sites of similar condition – 1.5 to 2.4 ha in size and soils stripped completely bare and exposed to direct precipitation – that were similar to those modeled for gas well sites. Sediment yields modeled for a 1.5 ha subwatershed (190 m slope length, sandy soils, 3.3% slope) were 13.2, 21.1, and 28.5 t/ha for the 2-, 5-, and 10-year RI, respectively.

As seen in the tables 2 and 3, sediment loadings from such unprotected sites can be substantial. Modeled sediment yields from unprotected sites illustrate that sediment yields are a function of both soil erodibility and slope, which indicates that both of these factors should be considered when developing erosion and sediment control site management plans. In this context, sediment yields from gas well sites within the study area appear to be more sensitive to increases in slope compared to soil erodibility factors. Therefore, when planning and designing erosion and sediment control BMPs greater emphasis should be placed on pre and post-development site topography compared to site-specific soil characteristics. Analyses of RIs demonstrate that even sediment loadings from relatively small events (1-year RI) are substantial enough to warrant protection from potential erosion impacts. Results thus suggest that even if a site is only exposed for a relatively short time frame adequate erosion and sediment protection should still be required.

Construction/Disturbance with Erosion or Sediment Control

Under managed conditions, annual average and RI sediment yields were substantially reduced. Table 5.2 summarizes *ER*s for each erosion and sediment control BMP. *ER*s based on average annual sediment yields ranged from as low as 0.52 for seeding for the high erodibility/high slope condition to as high as 0.93 for erosion blanket and sediment basin for the low erodibility/low slope condition. Return Interval *ER*s for the moderate erodibility/moderate slope condition ranged from as low as 0.68 for silt fence for a 10-year event to as high as 0.87 for an erosion blanket for the 1- and 5-year events. In the following discussion, each modeled BMP is described, discussed in the context of *ER* and differences in *ER*s among soil and slope combinations, and compared to published BMP efficiency (also referred to as effectiveness) values based on laboratory tests and field studies.

Erosion Control with Seeding and Mulching

Seeding establishes vegetated cover on disturbed areas and can be effective in controlling soil erosion once dense vegetation has been established. Under the conditions used in this study (model assumes short grass prairie seed spread with broadcast seeder), seeding *ER*s based on average annual sediment yields remained relatively constant for each erodibility factor (see Figure 5.3 for example of moderate slope condition) but decreased with increased slope (see Figure 5.4 for example of moderate erodibility condition). This is due to more erosion occurring on steeper slopes before vegetation can be established. Using the site management goal of 0.70, seeding should only be considered as an appropriate BMP for gas well sites located on 1.8% (or less) slopes as the *ER* drops below 0.70 for 2.9% (or greater) slopes. In more arid regions, irrigation may also be needed to establish vegetation. Irrigation was not considered in the model runs because typically water is not available for irrigation at gas well sites. Seeding was not included in the RI analysis because RI analyses are only applicable to one point in time, whereas seeding BMPs assume a period of time to establish vegetation.

Mulching involves applying plant residues or other suitable materials on disturbed soil in order to protect soil from detachment and erosion. In general, mulching accomplishes this goal by absorbing rainfall impacts and reducing overland flow velocities (McClintock and Harbor

1995). Mulching also helps to encourage plant growth by conserving moisture and moderating temperature (Goldman 1986). Modeled annual average mulching (model assumes native hay with application rate of 4000 lb/ac) *ER*s were equal to or greater than 0.80 for all combinations of soil erodibility and slope categories. As slope categories increased from low to high, mulching BMP efficiencies also increased. Also, mulching *ER*s decreased slightly for moderate soil erodibility condition (clay loam), compared to the low (loamy sand) and high (silty clay loam) soil erodibility condition. These results are due to ground cover being more effective for rill erosion compared to interrill erosion (Foster 2005). Slope steepness has a greater effect on rill erosion, and, silty and loamy soils are more susceptible to rill erosion (Foster et al. 2003). The RI analyses for mulching show that *ER*s are greater than 0.80 for all return intervals (Table 5.3). On steep slopes or on soils that are highly erodible, multiple mulching treatments should be used (USEPA 2002).

Doolette and Smyle (1990) reviewed 200 mulching studies and found mulching reduces soil erosion between 78 and 98%. In contrast, Jennings and Jarrett (1985) found that erosion rates from straw-mulch treatments were only 2 to 27% of that from bare soil conditions. Mulching and seeding can also be used in combination to improve vegetation establishment. Seeding and mulch combinations provide immediate protection by the mulch, and longer-term protection as vegetation becomes established when mulch decays (Harbor 1999). Hydroseeding applications of seed, mulch, water, fertilizer, and tackifier allow for treatment of steep slopes quickly (Harbor 1999) and are commonly used on construction sites (Faucette et al. 2005). Faucette (2006) found hydroseed application on research plots, combined with a mulch berm or silt fence, reduced soil loss on research plots by 99%. However, achievable erosion prevention on the scale of a construction site was estimated at approximately 50% due to logistical difficulties with establishing and maintaining adequate temporary coverage on constantly changing site conditions (Harbor et al. 1995).

Erosion Control with Erosion Blankets

Erosion blankets are also referred to as Turf Reinforcement Mats (TRMs). Erosion blankets typically use synthetic materials to form a high strength mat that helps to both prevent erosion on steep slopes and enhance the natural ability of vegetation to permanently protect soil from erosion by allowing soil infilling and retention (USEPA 1999). Under study parameters, erosion blankets had the highest *ER*s for all combinations of soil erodibility and slope conditions except the low erodibility/low slope and low erodibility/moderate slope conditions (Table 5.2). Since erosion blankets are designed for steep slopes, it is not surprising that erosion blankets performed the best on steep slopes ($ER \geq 0.92$). Erosion blankets also had the highest *ER*s, with all values being equal to or greater than 0.85 regardless of RI (Table 5.3).

Godfrey and Curry (1995) compared numerous erosion blanket products on clay soil research plots and found them to be at least 75% effective. Under simulated rainfall conditions, Benkin et al. (2003a) compared a straw-mulch treatment and three erosion blanket products (bonded-fiber matrix, straw/coconut blanket and wood-fiber blanket) on clay soil research plots and found erosion from straw-mulch plots was roughly one-tenth of that from bare soil plots and erosion from wood-fiber blanket and bonded-fiber matrix plots was one-tenth of that from the straw-mulch plots. Results were similar under natural rainfall conditions (Benkin et al. 2003b). Although erosion blankets perform well on steep slopes, these methods should not be used to prevent slope failure due to causes other than surficial erosion or when flow velocities and shear stress are greater than 15 feet per second and 8 lb/ft2, respectively (USEPA 2002).

Sediment Control with Silt Fences

Silt fences are the accepted standard for containment of silt and sediment on construction sites (Tyler 2001). In general, silt fences reduce sediment yields by slowing runoff velocities and filtering sediment as runoff flows through fence fabric. Silt fences have been the preferred method of erosion control because of their perceived advantages such as more than 6 month effectiveness, strong construction, good ponding depth, greater than 75% removal efficiencies easy assembly, and relatively low cost (Goldman 1986). Many fabrics are available with varying efficiencies based on mesh size, filtration capacity, and strength. For average annual sediment yields, silt fence *ER*s decreased with increasing soil erodibility categories for all slope conditions (see Figure 5.3 for example of moderate slope condition), and remained below 0.70 for all slope conditions when the erodibility condition was high (Table 5.2). ERs decrease as erodibility increases because, for Denton County soils, the percentage of silt and clays (smaller particle size) also increase as erodibility increases. Silt fence is more effective for coarser silts and sand material (NCTCOG 2006). In contrast, silt fence *ER*s tended to increase with increasing slopes. This relationship is due to more erosion occurring on steeper slopes and therefore a greater potential for more sediment to settle out in the ponded water behind the silt fence.

Sediment from construction sites typically consist of a larger percentage of smaller sized particles (clay and silt) because smaller particles are more easily dislodged from compacted soils and are more easily transported (Schueler and Lugbill 1990). Havens (2007) collected sediment from weirs used to measure runoff from three gas well sites in North Central Texas and found the percentage of silt and clay (particles < 62.5 μm) ranged from 63 to 78%. This measure is likely conservative considering a large percentage of the smaller particles would have remained suspended traveling through the weir whereas the larger particles would have had a greater tendency to settle. For RI analyses, silt fence had an *ER* greater than 0.70 for the 1, 2, and 5-year RI, but fell below 0.70 for the 10-year RI (Table 5.3).

RUSLE 2.0 sediment yield predictions assume proper installation and maintenance, which is important to silt fence efficiency and longevity. Silt fences should only be used in areas where sheet flow occurs and should be reinforced with a rock check dam or sand bag berm if concentrated flow occurs. Proper construction requirements include a maximum drainage area of 0.10 hectare or less per 30.5 linear meters of fence, maximum flow to any 6.1 meter section of 0.03 m^3/s, a maximum distance of flow to a silt fence of 30.5 meters or 15.2 meters if the slope exceeds 10 percent, and a maximum slope adjacent to the fence of 2:1 (NCTCOG 2006). Over time, efficiency decreases and breach potential increases if sediment deposits behind the fence are not removed.

Total suspended sediment removal from silt fences in laboratory settings ranged from approximately 85 to 100% (Crebin 1988; Kouwen 1990). Kouwen (1990) may have overstated the removal efficiency that could be expected at a typical construction site due to the use of a sediment slurry that contained solids that are much larger (200 μm) than those typically found at construction sites (Barrett et al. 1998). In field studies, silt fence efficiency was much more variable. Horner at al. (1990) investigated removal efficiencies on research plots and found the silt fences removed 86% of sediment from runoff. Barrett et al. (1998) collected runoff samples upstream and downstream of silt fences at six construction sites and found efficiencies ranged from -61 to 54%. Poor removal efficiencies were attributed to difficulties of in situ sampling at construction sites and a high percentage of silt and clay-sized particles, which ranged from 68 to 100% with a median value of 96%.

Sediment Control with Vegetated Filter Strips

Filter strips provide a physical separation between the disturbed site and water body or property boundary. Vegetated filter strips (VFS) are low-gradient vegetated areas that filter overland sheet flow. Their effectiveness is dependent on vegetation type, soil infiltration rates, flow depths, and travel times (USEPA 2002). Filter strip *ER*s were higher for the low erodibility condition (loamy sand) compared to the moderate erodibility condition (clay loam) and high erodibility condition (silty clay loam) because loamy sand has a greater fall velocity compared to clay and silt particles and RUSLE 2.0 computes deposition mainly as a function of fall velocity (Foster et al. 2003). *ER*s were greater than 0.70 for all combinations of erodibility and slope conditions (Table 5.2). Filter strips also had *ER*s greater than 0.70 for all RI years (Table 5.3). Maintenance of filter strips requires inspection to ensure that channelized flows do not occur and may require sediment removal (USEPA, 2002a).

VFSs have been studied extensively in field settings. In a review of 16 studies investigating VFS performance for feed lots (Koelsch 2006) suspended sediments were commonly reduced by 70 to 90% and variations were due to site-specific conditions such as vegetation, slope, soil type, and geometry of the filter strip. Koelsch (2006) also reported that most solids were removed within the first few meters of the filter strip. Han et al. (2005)

collected runoff from a filter strip treating highway runoff and found it was effective in removing more than 85% of the incoming suspended sediment. Particles greater than 125 µm appear to be easily trapped by vegetative treatment systems, but trapping efficiency decreased for particles less that 60 µm and become poor between 6 and 32 µm (Meyer et al. 1995; Deletic 1999, 2001).

In a modeling demonstration where the goal was to reduce sediment yield by 75%, optimal filter lengths were 1 to 4 m for sandy clay compared to 8 to 44 m for clay (Munoz-Carpena 2004). Gharabaghi et al. (2006b) investigated the sediment removal rates for various combinations of filter strip widths and vegetation types and found sediment removal efficiency increased from 50 to 98% as the VFS increased from 2.5 to 20 m. Approximately 62% of clay sized particles and up to 95% of silt sized particles were trapped in the first 5 m of the filter strip. These studies indicate filter width is an important factor in filter strip efficiency.

Sediment Control with Sediment Basins

Sediment basins are designed structures that promote settling of sediment from reduced flow velocities. Basins are usually installed at the low point of the site prior to full-scale grading and remain on site until the disturbed area is fully stabilized. Dewatering of the basin is typically achieved through a single riser and drainpipe or by passing the water through the gravel of a rock check dam. Sediment basins are popular with developers because they require less maintenance than other erosion and sediment control practices and can be integrated as permanent storm water management facilities (Harbor 1999).

Modeled annual average sediment basin *ER*s ranged from 0.77 to 0.93 (Table 5.2) and tended to be highest for the low erodibility (loamy sandy soils) condition (see Figure 5.3 for example of moderate slope condition). RUSLE 2.0 computes the sediment delivery ratio from a mixture of eroded primary particles and aggregates, and consequently sandy soils produce poorly aggregated sand-sized primarily particles that are easily deposited (Foster 2005). *ER* values did not decrease with increased slopes as would be expected if the basin was designed to capture, and then slowly release, all the runoff from the site. Sediment basin *ER*s decreased with increasing RI (Table 5.3) due to the decreases in residence time that result from increased runoff volumes of increasingly larger rainfall events.

*ER*s determined from the modeling results were greater than would be expected as sediment basins are generally designed to remove 50 to 75% of sediment that enters the structure (Goldman et al. 1986) and even the best designed sediment basin seldom exceeds a removal rate of 75 % (USEPA 2002). RUSLE 2.0 not only assumes sediment basins are well designed, but also assumes that basins are well maintained and perform at maximum efficiency (Foster 2005). McClintock and Harbor (1995) modeled a variety of sediment basin sizing scenarios and found trap efficiencies for a common design standard (127 m^3/ha of storage) was only 26, 21, and 20% for the 2-, 5-, and 10-year RI storms, respectively.

Variability of sediment basins removal efficiencies is high among field studies. The City of Austin, Texas (1999) reported TSS removal efficiencies of 46 and 17% for wet and dry basins, respectively. Kayhanian et al. (2001) also reported wet basins were more efficient that dry basins with TSS removal efficiencies of 96 and 64%, respectively. Generally, sediment basins have poor trapping efficiencies for fine sediments (Nighman and Harbor 1997), however, Gharabaghi et al. (2006a) monitored sediment ponds at construction sites and measured sediment removal efficiencies greater than 90% even though 50% of the particles were less than 3.75 µm (clay sized particles). Variability among studies is likely due to basin design, maintenance, sampling error, and particle size differences of the measured sediment.

Comparison of BMP Efficiencies and Costs

Based on modeled average annual sediment yields, erosion blankets and mulching were the most effective practices for the moderate and high soil erodibility conditions and moderate slope condition; both practices had an *ER* greater that 0.80 (Figure 5.3). Filter strip and sediment basin were the next most efficient BMPs, with *ER*s of 0.79 and 0.77, respectively. While *ER*s of filter strip and sediment basin were not much different between moderate and high erodibility conditions, the BMPs tended to perform better on the low soil erodibility condition (see Figure 5.3 for example on moderate slope condition). Silt fence was adequate for low ($ER = 0.84$) and moderate ($ER = 0.74$) erodibility but not for high erodibility condition ($ER < 0.70$). For the moderate slope condition (Figure 5.3), the seeding *ER* was less than 0.70 for each soil erodibility condition and therefore should not be considered as a viable BMP on sites with the 2.9% slopes or greater unless seeding is applied in combination with a complimentary BMP such as mulching or an erosion blanket.

For the moderate erodibility condition (Figure 5.4), all BMPs, except seeding for the moderate and high slope conditions, produced *ER*s greater than 0.70. However, since the seeding *ER* for the low erodibility soil was 0.71, this BMP could be considered appropriate for sites with a low slope condition. The relative order of BMP ERs is the same for all slope categories with erosion blanket and filter strip being the most effective for all conditions.

Figure 5.5 illustrates the comparison between ERs based on average annual sediment yields for the most common condition (moderate slope/moderate erodibility) and the two extreme conditions (low slope/low erodibility and high slope/high erodibility). This comparison shows that for the low erodibility/low slope condition, any of the BMPs would achieve the management goal of 0.70. Seeding would not provide adequate protection for the moderate erodibility/moderate slope condition and neither silt fence nor seeding would meet the site management goal of 0.70 for the high erodibility/high slope condition.

For developers and site managers, the most important factor in managing site runoff is typically cost. BMP unit costs available in the literature (USEPA 2002) were adjusted to 2007 dollars using the consumer price index. The per unit price (i.e., cost per cubic yard of erosion blanket) was multiplied by the total site area or length (for silt fence) to calculate a BMP site cost. BMP site costs were compared to BMP *ER*s (Figures 5.6, 5.7, and 5.8) in the context of two site management goals, 0.70 and 0.80, to illustrate how costs and efficiency can both be used to select the most cost effective BMP that meets specific site management goals. For the low slope/low erodibility site condition (Figure 5.6), all modeled BMPs met the 0.70 site management goal, of which seeding is the least expensive option. However, if the site management goal was 0.80, seeding would not be adequate and the most cost effective alternative would be silt fence. On moderate slopes with moderately erodible soils (Figure 5.7) silt fence would be both adequate and the least expensive BMP under a site management goal of 0.70, but for a site management goal of 0.80 only mulching or erosion blanket would be adequate. Mulching would be the most cost effective option to meet site goals under these conditions. For sites with a high erodibility/high slope condition (Figure 5.8), neither seeding nor silt fence would be adequate to meet a site management goal of 0.70. Under this condition, the vegetated filter strip would be the most cost effective option to meet either a site management goal of 0.70 or 0.80.

It is important to note that the site manager must also consider the length of time the site will require erosion and/or sediment control along with the associated maintenance and/or replacement costs (not included in the cost analysis). Considering these factors, a more efficient option might be to choose an alternative that is initially more expensive but is more permanent and has less maintenance costs such as a filter strip or mulching alternative.

Implications for Gas Well Sites

Generally, vegetated filter strips provide the most efficient, cost effective, BMP for sites located in North Central Texas. While seeding is the least expensive BMP it is only an alternative for low slopes with low soil erodibility. Silt fence is also a relatively inexpensive option but will not meet site management goals for all conditions. Also, theoretical/modeled efficiency for silt fence is much higher than efficiency observed at construction sites. Silt fences should be installed properly, regularly inspected, and properly maintained in order to provide adequate protection for a disturbed construction site.

For gas well sites, a compost filter sock can be used as an alternative to silt fence. A number of studies have reported that compost filter socks are at least as effective, and in many cases more effective, than traditional erosion and sediment control BMPs (McCoy 2005; Tyler and Faucette 2005). Compost in the filter sock can also improve the quality of runoff by absorbing various organic and inorganic contaminants, including motor oil (Tyler and Faucette 2005).

Modeled *ER*s were high for sediment basin under all site conditions but implementation costs are high compared to other alternatives that would meet site management goals, such as mulching or erosion blankets. Like silt fence, theoretical sediment basin efficiency modeled by RUSLE 2.0 assumes basins are well designed and perform at maximum efficiency, which is typically not the case at construction sites. Also, when the size of a gas well site lease would provide adequate space for the installation of a sediment basin, the area could probably be more efficiently used for a vegetated filter strip, which is nearly as efficient and less expensive. Erosion blankets also provide high ERs for gas well site conditions but are also relatively expensive. While filter strips, and in many cases, silt fences - if installed properly, regularly inspected, and properly maintained - are generally both cost effective and adequate to meet erosion and sediment site management goals for most conditions in North Central Texas, other areas with different slope and soil conditions and/or site management goals may require BMPs in combination, BMPs that have higher *ER*s, and BMP/BMP combinations that are more expensive.

Conclusion

This study presents an applied approach to evaluating sediment yields from modified field-scale geomorphologic features using natural gas well sites in North Central Texas as a case study. RUSLE 2.0 was used to evaluate reductions in modeled sediment yields for six best management practices. On unprotected sites, the model predicted average annual sediment yields ranged from 12.1 t/ha/yr for sites with low erodibility (sandy loam soils) and low slope (1.8% slope) conditions to as high as 134.5 t/ha/yr on sites with high erodibility (silty clay loam soils) and high slope (4.5% slope) conditions. Sediment yields were substantially reduced through best management practices by a minimum of 52% and up to a maximum of 93%. Generally, mulching and erosion control blankets had the highest *ER*s; however, from a cost efficiency standpoint, silt fences or filter strips were shown to be less expensive options for achieving site management goals in most cases. RI analyses illustrated that even small return intervals have the potential for high erosion and off-site sediment movement. The variation of *ER*s based on different combinations of soil erodibility and slope conditions demonstrates that, in the context of managing sediment migration from these sites, several management scenarios are possible and the most effective management strategies depend on individual site characteristics. Furthermore, comparison of modeled BMP sediment yield reductions and observed reductions illustrate that modeled BMP efficiencies are typically best case scenarios. Due to the flexibility of the model, the approach outlined in this manuscript can be applied to complex or simple slopes, can evaluate a wide variety of BMPs, and can be easily customized for specific site characteristics or geographical regions. Future analyses could consider evaluations of multiple BMP combination alternatives. In order to minimize sedimentation impacts from construction sites on receiving systems, planners, watershed managers, and regulatory agencies responsible for storm water quality should consider local and/or site-specific conditions when evaluating construction site management plans and when developing erosion and sediment control strategies, policies, and guidance documents.

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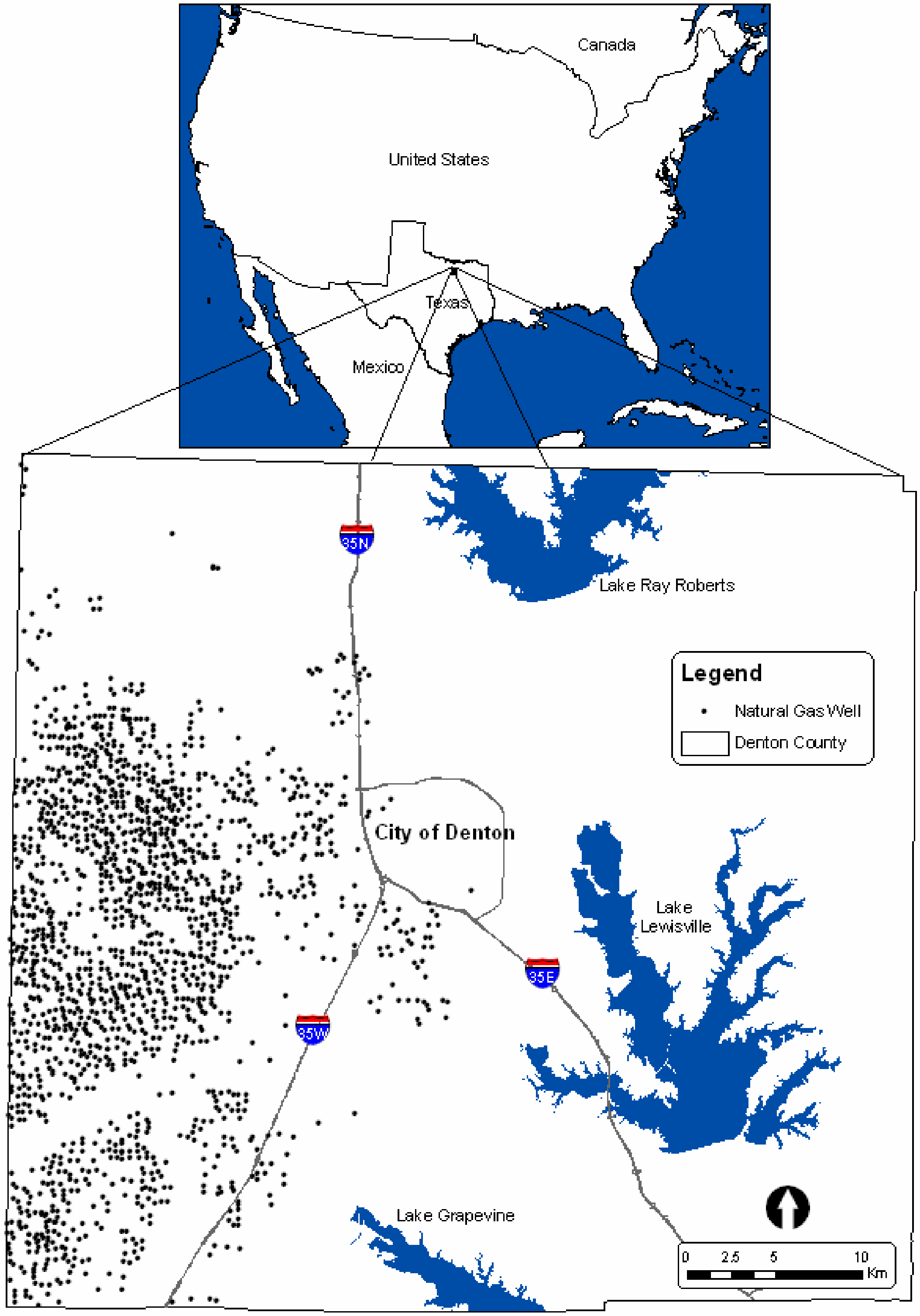

Figure 5.1. Study area Denton County, Texas, USA.

Figure 5.2. Gas well site on modified hillslope.

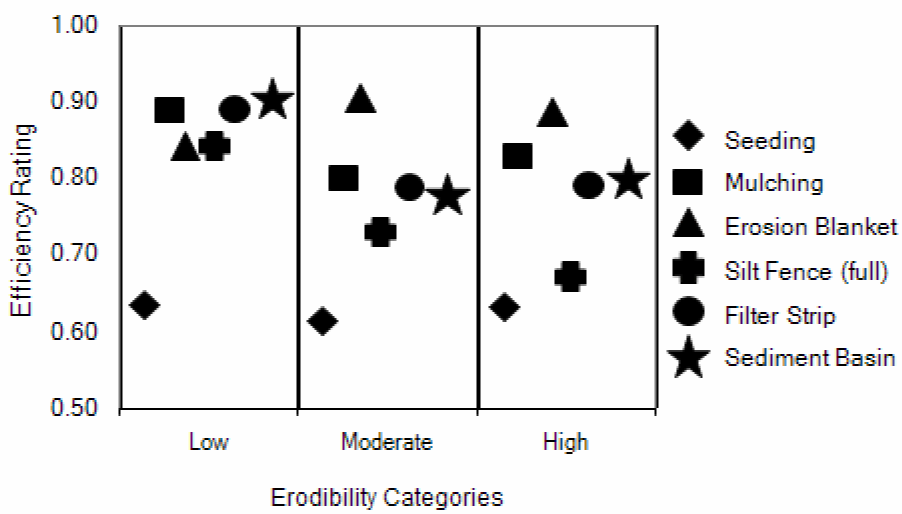

Figure 5.3. Average annual BMP ERs for moderate slope condition.

Figure 5.4. Average annual BMP ERs for moderate erodibility condition.

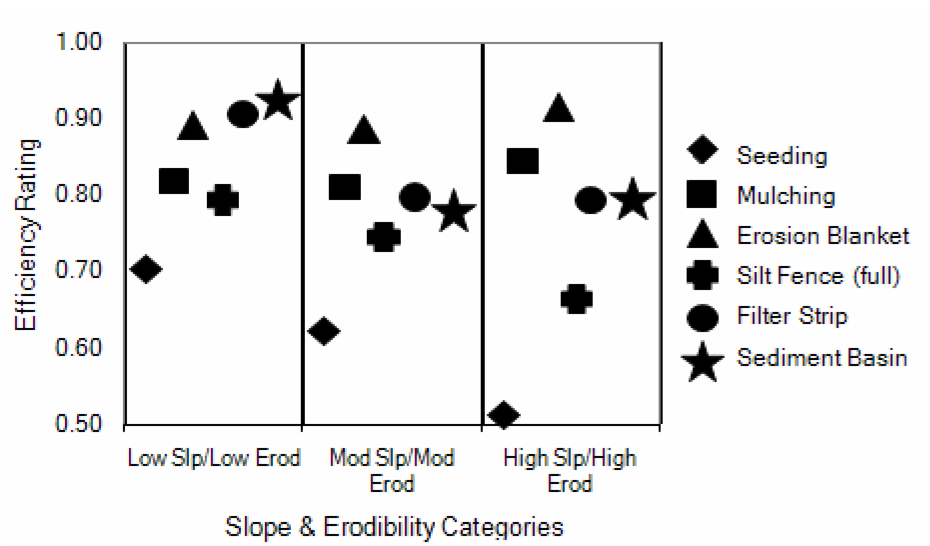

Figure 5.5. Average annual BMP ERs for low, moderate, and high combined factors.

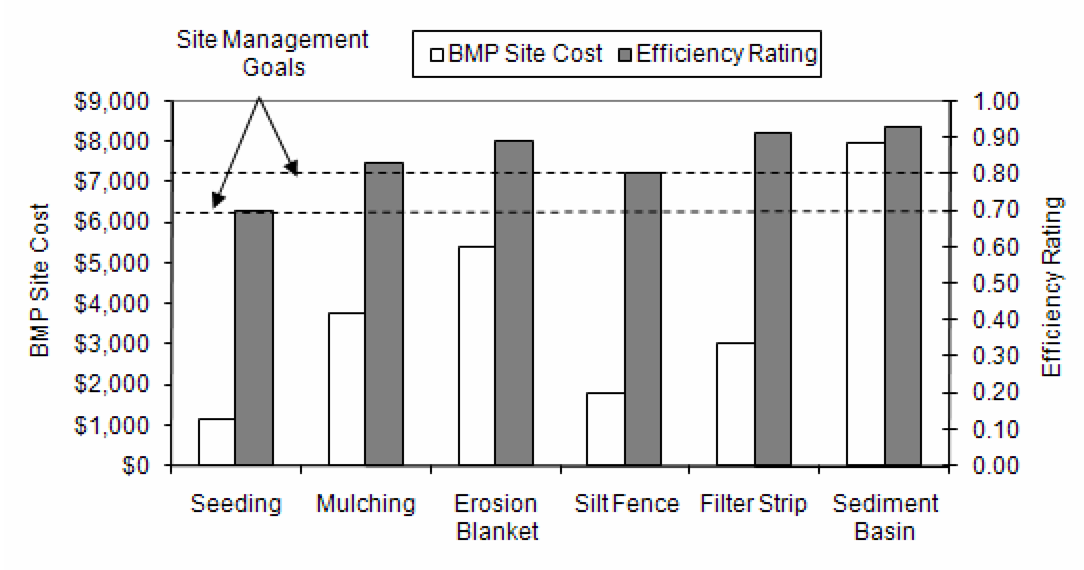

Figure 5.6. BMP cost/ER comparison for low erodibility/low slope condition.

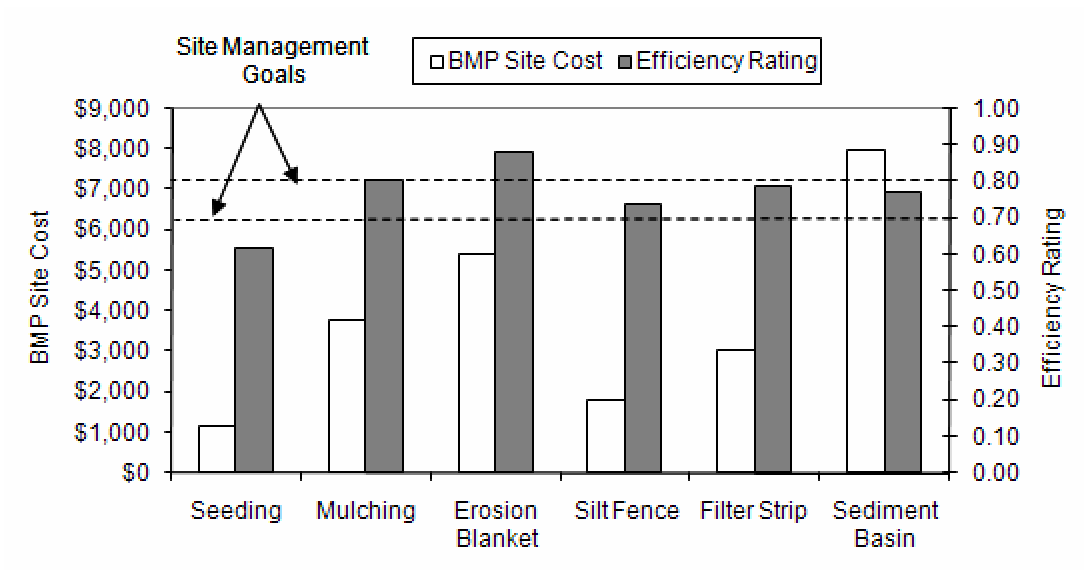

Figure 5.7. BMP cost/ER comparison for moderate erodibility/moderate slope condition.

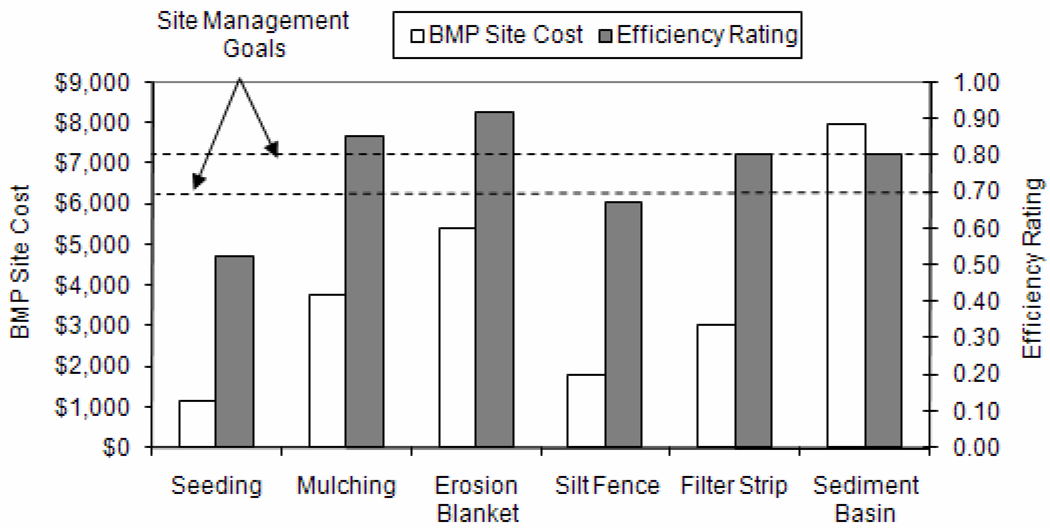

Figure 5.8. BMP cost/ER comparison for high erodibility/high slope condition.

Table 5.1. Modified Slope Profile Segments

	Modeled Slope Profiles								
	Cut Slope Segments[1]				Pad Site[2]	Fill Slope Segments[1]			
Low	3.5%	2.25%	2.0%	1.75%	1.5%	2.0%	3.0%	2.5%	1.0%
Moderate	10.0%	8.0%	6.0%	3.0%	1.5%	3.0%	6.0%	4.0%	2.0%
High	20.0%	15.0%	10.0%	5.0%	1.5%	5.0%	10.0%	8.0%	4.0%

[1] slope segment 4.6 m; [2] slope segment 61 m

Table 5.2. Average Annual Sediment Yields (tonnes/hectare) and *ER* Results

	Best Management Practice	Low Erodibility		Moderate Erodibility		High Erodibility	
		SY[1]	ER[2]	SY	ER	SY	ER
Low Slope	Unprotected	12.1	-	19.5	-	29.1	-
	Seeding	3.6	0.70	5.6	0.71	8.5	0.71
	Mulching	2.1	0.83	3.8	0.80	5.2	0.82
	Erosion Blanket	1.3	0.89	2.5	0.87	3.4	0.88
	Silt Fence	2.5	0.80	5.4	0.72	10.8	0.63
	Filter Strip	1.1	0.91	4.3	0.78	6.7	0.77
	Sediment Basin	0.8	0.93	4.5	0.77	7.2	0.75
Moderate Slope	Unprotected	24.7	-	38.1	-	60.5	-
	Seeding	9.0	0.64	14.6	0.62	22.2	0.63
	Mulching	4.0	0.84	7.6	0.80	10.1	0.83
	Erosion Blanket	2.7	0.89	4.7	0.88	6.3	0.90
	Silt Fence	4.0	0.84	10.1	0.74	19.7	0.67
	Filter Strip	2.7	0.89	8.1	0.79	12.8	0.79
	Sediment Basin	2.5	0.90	8.7	0.77	12.8	0.79
High Slope	Unprotected	56.0	-	85.2	-	134.5	-
	Seeding	26.9	0.52	42.6	0.50	65.0	0.52
	Mulching	6.5	0.88	13.2	0.84	19.7	0.85
	Erosion Blanket	3.8	0.93	7.0	0.92	10.5	0.92
	Silt Fence	7.4	0.87	22.4	0.74	67.3	0.67
	Filter Strip	5.4	0.90	17.0	0.80	26.9	0.80
	Sediment Basin	4.9	0.91	19.5	0.77	26.9	0.80

[1] Sediment Yield (tonnes/ha); [2] Efficiency Rating

Table 5.3. Return Interval Sediment Yields and *ER*s –Moderate Erodibility on Moderate Slopes

Best Management Practice	1-yr RI		2-yr RI		5-yr RI		10-yrRI	
	SY[1]	ER[2]	SY	ER	SY	ER	SY	ER
Unprotected	8.1	-	11.0	-	16.8	-	20.6	-
Mulching	1.4	0.83	2.1	0.81	2.9	0.83	3.8	0.82
Erosion Blanket	1.1	0.87	1.6	0.85	2.2	0.87	3.1	0.85
Silt Fence	2.0	0.76	2.9	0.73	4.7	0.72	6.5	0.68
Filter Strip	1.5	0.81	2.2	0.80	3.8	0.77	4.9	0.76
Sediment Basin	1.8	0.78	2.7	0.76	4.4	0.76	5.6	0.73

[1] Sediment Yield (tonnes/ha); [2] Efficiency Rating