United States Environmental Protection Agency Final Report for Catalog of Federal Domestic Assistance Grant Assistance Number 66.463 Water Quality Cooperative Agreement for Project Entitled "Demonstrating the Impacts of Oil and Gas Exploration on Water Quality and How to Minimize these Impacts Through Targeted Monitoring Activities and Local Ordinances"

ID No. CP-83207101-1

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12-17-07

PREPARED IN COOPERATION WITH THE

United States Environmental Protection Agency

The preparation of this report was financed through grants

from the United States Environmental Protection Agency

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INTRODUCTION

Much of the land surface of the Southwest portion of Denton County lies above a geological formation known as the Barnett Shale. In recent years natural gas well development has increased dramatically in the area, which has also increased awareness of potential impacts to the surrounding environment. The research presented in this document is based on a monitoring program designed to evaluate storm water runoff from natural gas well sites. This research was funded through a Water Quality Cooperative Agreement (104b3 grant) provided to the City of Denton by the United States Environmental Protection Agency (USEPA). The study began in October of 2004 and ended in September of 2007. The primary purposes of the study were to characterize storm water runoff associated with "typical" natural gas exploration and production and provide guidance on how to manage these sites *from a regulatory standpoint*. Specific objectives related to these research goals were to: (1) describe a monitoring approach that is practical and effective for small, highly modified, sites, (2) characterize the types and magnitude of pollutants in storm water runoff, (3) evaluate a modeling approach, (4) use research plots and simulated rainfall to characterize runoff specifically from the gas well drilling pad, (5) demonstrate how best management practices could reduce sediment pollution through modeling, and (6) characterize the contents of on-site drilling mud pits during the drilling process. These objectives are discussed in more detail in the following sections. Finally, a discussion is provided on how this research can be related to local natural gas well development policy and decision-making.

SECTION 1. STORM MONITORING PROGRAM

Appendix A describes, in detail, the basis for selecting sites for the study, the use of weirs for measuring runoff, and flow-interval storm water sampling procedures. Three gas well sites and two reference sites were selected for this study (see Figure X, Appendix A). One of the original gas well sites and one of the original reference sites were moved in January of 2007 due to issues related to site access. Sampling small, highly modified gas well sites presented numerous challenges, particularly because a large portion of the site was relatively flat. Typically, in natural drainage basins, small or large, the point to which the water drains is usually evident, whereas at gas well sites areas are often flattened and may drain in numerous directions. Additionally, small elevation changes in site drainage patterns may substantially alter what is believed to be the general runoff area. This can be problematic as slight elevation changes may cause a substantial difference in total runoff area, which can lead to inaccurate loading estimates. These problems were minimized with a site surveys using relatively inexpensive and easy to use survey equipment.

Smaller sites also generate smaller volumes of runoff and respond quickly to changes in rainfall. Due to these two conditions, special attention was given to minimum flow thresholds (enable levels) and flow paced sampling intervals. Prior to establishing these parameters, some knowledge of the site runoff characteristics must be known. The rational formula and SCS method were used in this study to estimate peak discharge and total runoff from the gas well and reference sites.

Sampling begins when the flow depth of the runoff exceeds the minimum flow threshold; sampling ends when the flow depth drops below this threshold. For these sites, the threshold was set as low as possible; as short intense storms were thought to produce a runoff volume would be sufficient to sample and would result in a response where the water level in the weir rises and falls quickly and is relatively low in volume. Thresholds should also be set low to reduce potential error, as increasing flow thresholds has been shown to result in increased error of the true pollutant load.

Setting the appropriate sampling interval was also challenging for the small, highly modified gas well sites, especially since both small and large events were targeted for sampling and the runoff area varied from site to site. Flow intervals in terms of volumetric depth, which is runoff depth over the entire watershed, were used to sample proportionate volumes of rainfall and runoff from multiple sampling sites regardless of the size of the drainage area. The SCS method was useful for evaluating various rainfall/runoff scenarios as the output is in depth units. Flow intervals set either too small or too large can result in missing the first flush of a storm and/or not sampling the end of a storm. Both issues may compromise the ability to sufficiently characterize an event. If analysis of the data is to include “within storm” characteristics, a two-part programming design may be necessary to capture a wide range of storm events (see

Appendix A). A two-part automated sampling program was designed to effectively sample both small and large events. The automated sample is programmed to take a set number of discrete samples, then composites samples to extend the number of total samples that can be taken for storms generating large volumes of runoff. While sampling gas well sites, or other small, highly modified areas, does present several challenges, careful planning followed by continual evaluation of each monitoring event can greatly increase the number, quality, and completeness of samples resulting in a more accurate characterization of the site or drainage area. The application of minimum thresholds and sampling intervals are discussed and illustrated in Appendix A.

SECTION 2. CHARACTERIZING STORM WATER RUNOFF FROM NATURAL GAS WELL SITES IN NORTH CENTRAL TEXAS

This section characterizes storm water samples collected from three gas well sites and two reference sites. All storm water samples were analyzed for a variety of water quality parameters (see Table 1, 2, 3, and 4) but a selected group of conventional water quality parameters (total dissolved solids (TDS), conductivity, hardness, alkalinity, pH, chlorides, calcium, turbidity, total suspended solids (TSS)); metals (As, Cd, Cr, Cu, Fe, Pb, Mn, Ni, Zn); and petroleum hydrocarbons (total petroleum hydrocarbons (TPH), Benzene-Toluene-Ethylbenzene-Xylene (BTEX)) were evaluated in more detail (Appendix B). These data were: evaluated for differences between gas well sites and references sites using summary statistics and nonparametric statistic; 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.

Event Mean Concentrations (EMC) of total dissolved solids, conductivity, calcium, chlorides, hardness, alkalinity and pH were higher at gas well sites compared to reference sites and differences were statistically significant for all parameters except conductivity. Generally, the presence of metals was higher at gas well sites compared to reference sites and EMCs were statistically 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 there were much fewer incidences of EMCs above drinking water standards and only one EMC was above the aquatic life criterion for cadmium. Compared to metal EMCs reported by Hudak and Banks (2006) for local mixed use watersheds, gas well site median EMCs were similar but maximum EMCs 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, all EMCs for TPH and BTEX 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 21.4 to 40.0 t ha^{-1} yr^{-1}, respectively, were comparable to those reported by previous studies aimed at characterizing sediments in construction site runoff. 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 and 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.

SECTION 3. EVALUATION OF WEPP FOR RUNOFF AND SEDIMENT YIELD PREDICTION ON NATURAL GAS WELL SITES

This section describes a modeling approach for estimating sediment yield from gas well sites. The objective of this research was to evaluate Water Erosion Prediction Project (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. A detailed discussion pertaining to the methods and results are available in Appendix C.

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. These results demonstrate 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.

SECTION 4. RAINFALL SIMULATION EXPERIMENTS ON NATURAL GAS WELL PAD SITES: DEVELOPING EROSION PARAMETERS FOR WEPP

This section describes an approach to deriving an interrill erodibility parameter for the WEPP erosion model using rainfall simulation experiments. 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.

A Norton Rainfall Simulator was used to apply multiple applications of simulated rainfall on 9 research plots, 6 plots on two natural gas well pad sites and 3 plots at a nearby reference site. Rainfall was applied at a rate of 58 mm h^{-1} to dry, wet, and very wet soil conditions. A discussion pertaining to the methodology and results from this research is provided in Appendix D.

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.

The interrill erodibility parameter for gas well sites ranged from 443,746 to 1,123,131 kg sec m^{-4}. 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}.

SECTION 5. MODELING EROSION AND SEDIMENT CONTROL PRACTICES WITH RUSLE 2.0: MANAGEMENT APPROACH FOR NATURAL GAS WELL SITES IN DENTON COUNTY, TEXAS, USA

This section 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 Version 2 of the Revised Universal Soil Loss Equation (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. The methodology and a detailed discussion of results if provide in Appendix E.

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.

SECTION 6. PIT SAMPLING DATA

Introduction

Gas well drilling operations require the use of drilling mud, which is usually contained in a lined pit on the drilling site. Drilling mud serves a variety of purposes, including maintaining the stability of the formation and preventing the flow of gas while drilling by balancing the hydrostatic pressure in the well. Drilling mud also lubricates and cools the drill, transports cuttings from the production zone while preventing rock dispersion from blocking the formation, and helps control fluid and solid invasion into the formation.

By local regulation, all drilling mud used within the City of Denton is required to be freshwater based mud, typically referred to as water-based muds, or WBMs. Water-based drilling mud in this area typically consists of bentonite clay (gel) with additives such as barium sulfate (barite) and calcium carbonate (chalk). Caustic soda is also often a component of drilling muds, and various thickeners are used to influence the viscosity of the fluid. Many other chemicals may also be used to enhance or maintain desirable properties during the drilling operation. Drilling mud properties are routinely analyzed by operators to ensure that desirable physical and chemical properties are maintained during the drilling process.

Local operators typically construct drilling mud pits on or adjacent to the pad sites, and near the location of the actual wellhead. Pits usually have an earthen dam and are lined with a plastic liner to ensure pit contents remain in place. Often, pits are divided into two connected “chambers”, which aids in mixing and removal of cuttings during the drilling operation. In many cases, pits within the Denton area are designed to capture rainwater, which has lead to some earthen dams failures and subsequent releases of pits contents during large rain events. Because of this concern, local regulations prohibit gas well drill sites from being located slopes greater than ten percent.

Although there are a variety of potential impacts from drilling mud releases, total petroleum hydrocarbons (TPH), the combination of benzene, toluene, ethylbenzene, and xylenes (BTEX), and benzene itself are listed in the Denton Development Code as contaminants of concern for drilling mud pits. Acceptable concentrations of these compounds were initially established in the Development Code according to the Texas Administrative Code regulations concerning discharges of petroleum contaminated water to waters of the State (30 TAC §321). However, it should be noted that the Texas Commission on Environmental Quality has recently repealed the “petroleum hydrocarbon” components of 30 TAC §321 and replaced them with Texas Pollutant Discharge Elimination System (TPDES) General Permit TXG830000. TXG830000 authorizes discharges of water contaminated by petroleum fuel or petroleum substances from a variety of sources, and discharge standards for TPH, BTEX, and benzene are unchanged from those originally established in 30 TAC. Table 6.1 summarizes the current standards for TPH, BTEX, and benzene, as outlined in Subchapter 22 of the Denton Development Code.

Compound	Concentration Limit
TPH	15 mg/L
BTEX	500 µg/L
Benzene	50 µg/L

Table 6.1. Concentration limits for Total Petroleum Hydrocarbons, BTEX, and Benzene in mud pits, as defined in Subchapter 22 of the Denton Development Code

Drilling muds were sampled from pits using simple grab sample methods. Sampling containers were suspended on specially designed sampling poles and then used to collect samples in the areas closer to the middle of the pits. This method of sample collection was utilized to ensure that influences from the banks of the pits were minimized, and to also ensure that pit contents were as adequately mixed as possible.

<u>Initial Analytical Attempts</u>

Initial analyses were conducted using a Portable Hydrocarbon Analyzer (PHA-100, PetroSense, Inc., 1181 Grier Drive, Building B, Las Vegas, NV, USA 89119). The PHA-100 offers the advantages of quantifying petroleum hydrocarbons in the field, as well as the potential to eliminate the need to use solvents to extract TPH from solutions by using Fiber Optic Chemical Sensor (FOCS) technology. The instrument can be set up to measure petroleum hydrocarbons dissolved in aqueous or vapor phases. Although the instrument does not have the capability to separately analyze the target compounds BTEX or benzene, it does respond strongly to BTEX components. Based on these capabilities, the project team wanted to determine if the instrument could be used to rapidly screen drilling muds for high TPH levels. If successful, a large number of drilling mud samples could be screened using this method, and those samples that exceeded screening criteria could be further analyzed using the standard laboratory methods of Texas 1005 for TPH and EPA 8021B SW 8260B for BTEX.

The PHA-100 calibrated and performed well in the laboratory using a range of standards and spiked water samples. However, TPH concentrations as derived from PHA-100 analyses of initial drilling mud samples did not compare well with concentrations derived for the same samples using standard laboratory methods of solvent extraction followed by gas chromatography (GC). Analyzing TPH in either the aqueous and vapor phases did not make an appreciable improvement in the relationship between FOCS and GC results. The reasons for the discrepancies between FOCS and GC methods are unclear, although discrepancies are likely due at least in part to the complex nature of the drilling mud itself. Drilling muds are a complex mixture of various chemical constituents, suspended in a slurry-like matrix of bentonite clay and water. Such mixtures have the potential to create complex sorption dynamics and interference effects, which could have an influence on TPH concentrations derived from FOCS when compared to solvent based extraction methods followed by GC-FID analyses.

After initial analyses of TPH in drilling mud using the PHA-100, the project team evaluated the performance of Enzyme Immunoassays (EIAs) for drilling mud analyses. EnviroGard™ Petroleum Fuels in Water (TPH) EIA kits were obtained from Strategic Diagnostics, Incorporated (SDI, 111 Pencader Drive, Newark, DE, USA 19702-3322) and utilized to examine mud pit TPH concentrations. EIA methods performed well during laboratory calibration and when comparing results against prepared standards in aqueous phase. However, similar to the FOCS, EIA methods did not give results that were consistently comparable to those obtained using solvent extractions followed by GC-FID analyses when analyzing mud pit samples. The reasons for this are likely due to the same reasons outlined above for FOCS technologies, namely complex sorption dynamics and interference effects.

Because of the concerns for potential bias from either FOCS and EIA methods, the project team elected to analyze all mud pit samples using solvent extraction followed by GC analyses. The Texas 1005 method was used to analyze TPH concentrations, and EPA 8021B was used for BTEX analyses. Using this method ensured the most consistent results, and offered the additional advantages of providing concentration data for total petroleum hydrocarbons based upon size categories the molecules carbon chains (C6-C12: "Gasoline Range" hydrocarbons; >C12-C28: "Diesel Range" hydrocarbons, and >C28-C35: "Oil Range" hydrocarbons. Data for total BTEX concentrations were also divided into concentrations of benzene, toluene, ethylbenzene, and xylenes. All analyses were performed by Xenco Laboratories, 9701 Harry Hines Boulevard, Dallas, Texas USA 75220.

Mud Pit Sampling Results – Liquid Phase Samples

Total Petroleum Hydrocarbons (TPH)

A total of 79 mud pit samples were analyzed for TPH during the grant period. Table 6.2 summarizes the results of these analyses. Overall, more than half of the mud pit samples contained TPH concentrations above method detection limits. In general, TPH concentrations in excess of standards were still relatively low, although there were a few instances of very high concentrations (maximum 25,590 mg/l). This situation is evident when comparing the mean concentration of all mud pit samples, 1,310.8 mg/l TPH, to the median concentration of the sample samples (63.3 mg/l). Overall, TPH data were skewed by these few high values, and are thus best summarized by the median concentration and associated quartiles. Table 6.3 summarizes the number of samples collected that exceeded the regulatory standard of 15 mg/L TPH. As can be seen in the Table, sampled mud pits exceeded the regulatory standard in approximately 46 percent of the collected samples.

Pit samples that exceeded the TPH standard of 15 mg/l were further evaluated to determine the carbon chain size category of petroleum hydrocarbons within the sample. Generally speaking TPH concentrations were predominantly reflective of hydrocarbons in the >C12 to C35 range. Hydrocarbons in the C6-C12 range exceeded the 15 mg/L standard in approximately 16.6 percent of the samples. Hydrocarbons in the >C12-C28 range, however, were observed in excess of 15 mg/L in 88.9 percent of the samples. Hydrocarbons in the >C28-C35 range were also prevalent, with approximately 67.6 percent of samples containing this hydrocarbon category. Since pit assessment sampling was not designed to differentiate sources of these various hydrocarbon size categories, it is unclear whether site activities, sources from the formation, machinery, or other sources contributed to hydrocarbon concentrations in the pits. However, considering the preponderance of diesel operated machinery and hydraulic equipment at typical gas well sites, as well as the prevalence of hydrocarbon concentrations in the >C12-C28 range ("Diesel Range"), it is likely that contamination was due at least in part to site operations such as maintenance activities, fuel / hydraulic fluid leaks and spills, or similar sources. Hydrocarbons in the >C28-C35 may be from the formation itself, leaking machinery, the intentional disposal of waste oil in the mud pits, or some combination thereof. More specific sampling and site assessment is needed before sources can be more definitively identified.

Benzene, Toluene, Ethylbenzene, and Xylenes (BTEX)

A total of 76 mud pit samples were analyzed for BTEX. Table 6.2 summarizes the results of these analyses. Approximately 26 percent of the mud pit samples contained BTEX concentrations above method detection limits. In general, BTEX concentrations tended to be low, with a few cases of relatively high concentrations (maximum 1.70 mg/l). Data were somewhat skewed, and are thus best summarized by the median and quartiles instead of the mean and standard deviation. The mean BTEX concentration was 0.17 mg/l, while the median concentration was 0.01 mg/l. Table 6.3

summarizes the number of samples collected that exceeded the regulatory standard of 0.5 mg/L BTEX. As can be seen in the Table, only 2.6 percent of the sampled mud pits exceeded the regulatory standard for BTEX.

Generally, xylenes were the most commonly observed BTEX component (detected in 18% of samples), followed by toluene (13%), benzene (11%), and ethylbenzene (8%). The two pit samples that exceeded the BTEX standard of 0.5 mg/l were comprised mainly of xylene (41 and 33 percent, respectively), toluene (27 and 47 percent, respectively), benzene (23 and 18 percent, respectively), and ethylbenzene (9 and 2 percent, respectively). Sources of these components are unclear, but are likely due to maintenance, repair, and cleaning activities associated with site equipment.

Benzene

Benzene was detected above method detection limits in approximately 11 percent of mud pit samples (Table 6.2). As seen in Table 6.3, only 4 percent of collected samples exceeded the benzene regulatory standard of 50 µg/l (0.05 mg/L). Samples above the regulatory standard ranged from a low of 0.13 mg/l to a high of 0.31 mg/L. Sources of benzene are unclear, although it is likely that benzene contamination was due, at least in part, to maintenance, operation, or cleaning of on-site equipment.

	TPH	BTEX	benzene	toluene	ethylbenzene	xylenes
Total samples	79	76	76	76	76	76
No. above detect.	49	20	8	10	6	14
No. below detect.	30	56	68	66	70	62
Mean	1,310.8	0.17	0.08	0.12	0.02	0.11
Standard dev.	4,702.8	0.40	0.11	0.25	0.03	0.16
Median	63.3	0.01	0.01	0.01	0.01	0.04
Min	BD	BD	BD	BD	BD	BD
Max	29,590.0	1.70	0.31	0.81	0.07	0.56
First quartile	9.6	0.01	0.01	0.00	0.00	0.00
Third quartile	259.6	0.10	0.14	0.11	0.02	0.13

Table 6.2. Concentrations of Total Petroleum Hydrocarbons (TPH), BTEX, Benzene, and various length petroleum hydrocarbon chains. Values are expressed in mg/L. “Detect” = detection limit of method. BD = “below detection / recording limit” for the method

	TPH	BTEX	Benzene
Number above regulatory standard	36	2	3
Proportion exceeding standard	45.6	2.6	4.0
Total	79	76	76

Table 6.3. Number and proportion of samples exceeding regulatory concentrations of Total Petroleum Hydrocarbons (TPH), BTEX, and benzene.

Solid Phase Pit Samples

On occasion, samples were collected from pits that were in various stages of evaporative drying or were otherwise being managed to produce pit contents with less liquid content. In some instances, these slurries were considered by the laboratory to be in “solid” phase instead of liquid phase. Solid phase samples were rare, with only 6 samples collected during the grant period that were considered by the laboratory to be in solid phase. Solid phase samples were analyzed using Texas 1005 for TPH and EPA

8021B for BTEX in a manner similar to liquid phase samples, although due to the physical state of the sample, results were reported in concentration units of mg/kg.

In general, solid phase samples exhibited much higher TPH concentrations than were observed in liquid phase samples. The reasons for this phenomenon are unclear, although it is likely that the evaporation of water from these pits served to concentrate petroleum hydrocarbons near the surface of the resulting slurry. Of the 6 total samples, 4 exhibited measureable TPH concentrations, with total TPH concentrations ranged from 62,500 mg/kg to 391,100 mg/kg. TPHs in solid phase samples were predominantly comprised of longer carbon chain molecules in the C12 to C35 range. Only one solid phase sample contained measureable amounts of petroleum hydrocarbons in the C6-C12 range. In general, the highest TPH concentrations observed in solid phases were from carbon molecules in the C12-C28 range, although C28-C35 range molecules were also commonly observed in relatively high concentrations. Solid phase pit samples are summarized in Table 6.4.

Site Name	Date	TPH	Benzene	BTEX	C6-12	C12-28	C28-35
Underwood #1	2/11/05	391,100	BD	BD	BD	214,100	177,000
Guyer #2	11/4/05	190,000	0.02	0.02	BD	81,500	109,000
Burch #1H	11/4/05	BD	0.01	0.01	BD	BD	BD
White #1H	11/4/05	76,600	0.06	0.14	BD	45,800	30,800
Yarbrough #3	1/27/06	BD	BD	BD	BD	BD	BD
Fitts #1	11/3/06	62,500	BD	BD	636	49,800	12,100
Underwood #1	2/11/05	391,100	BD	BD	BD	214,100	177,000

Table 6.4. Concentrations of Total Petroleum Hydrocarbons (TPH), BTEX, benzene, and various length petroleum hydrocarbon chains from solid phase samples. Values are expressed in mg/kg. BD = "below detection / recording limit" for the method.

Summary of Pit Sample Data

In general, BTEX and benzene concentrations were below the regulatory standards set forth in Subchapter 22 of the Denton Development Code. To provide some context, most pits did not exhibit concentrations of either BTEX or benzene that would exceed the standards for discharges into waters of the State under TXG830000. Total petroleum hydrocarbons (TPH) concentrations, however, regularly exceeded concentrations that would be allowable under TXG830000. While this sampling effort was not designed to differentiate sources of TPH for gas well pad sites, the fact that TPH concentrations were detected in excess of the standards set forth in the Denton Development Code in almost half of the pit samples indicates that TPH contamination is a pervasive problem. For this reason, it is recommended that similar standard be utilized for municipalities and other regulatory agencies to ensure that impacts from pit content releases are minimized. However, enforcement of standards for pit contents can be a difficult task, as drilling mud serves both operational and safety functions. However, in mange cases booms, skimmers, and oil adsorbent materials can be used to remove and dispose of TPH within mud pits without jeopardizing operations or safety.

It is important to note that mud pits can contain a wide variety of chemical constituents that were not analyzed as a part of this sampling effort and that could potentially cause impacts if released into the environment. Because pits may capture overland flow during rain events, either by design or by location, it is important to ensure that dam breaches or releases from overtopping are minimized. The City of Denton has dealt with these issues through standards established in Subchapter 22 of the Denton Development Code. In addition to TPH, BTEX, and benzene standards, pits are required to be lined, must be located on slopes less than 10 percent, must have pit contents at least two feet from the top of the pit (2 feet of freeboard), must maintain chloride content less than 3000 mg/l, and must be dewatered within 30 days of well completion and removed within 90 days of well completion. Details of the Development Code may be downloaded from the City of Denton website (www.cityofdenton.com) and are discussed in somewhat more detail in Section 7 regarding general recommendations for municipal gas well environmental regulations.

SECTION 7. REGULATORY INFORMATION

Introduction

The issue of regulating gas well drilling is complex, and should at minimum cover aspects of public health and safety, nuisance issues such as noise and dust, site security, signage, road damage, site operations, fire safety, fluid and waste disposal, tree preservation, and many other issues in addition to storm water / water quality impacts. The issue of pipeline regulations should also be considered, as the pipeline collection, processing, and distribution networks are often extensive and in many cases not well mapped, particularly with regards to small gathering lines that convey gas from individual wells. Numerous regulatory authorities have been established at the Federal, State, and local levels, with different organizations having regulatory oversight depending on the issue at hand. Since covering all aspects of gas well drilling regulations is well beyond the scope and intent of this document, this section will summarize Denton's local regulations concerning environmental issues associated with gas well exploration and development. Many of these local regulations are summarized in Subchapter 22 of the Denton Development Code (DDC), which may be downloaded from www.cityofdenton.com. This section will focus on the current environmental components of Subchapter 22, as well as additional information derived as a result of the research conducted for USEPA CP-83207101-1. Additional local regulations in the DDC that may be of interest to readers include Subchapter 13 related to tree preservation, Subchapter 17 concerning regulation of Environmentally Sensitive Areas, and Subchapter 19 pertaining to Drainage standards. In general, Denton's local environmental regulations for gas wells can be divided into the following categories: drilling locations, tree preservation, and regulations involving site design, construction, and management.

Regulations concerning drilling locations

Regulations involving drilling locations are traditionally associated with site "setback" requirements from residential structures and places of assemblage such as schools, churches, etc. However, from an environmental perspective, regulations concerning locations can be designed to protect surface water resources, various categories of environmentally sensitive areas, and watersheds in general. Subchapter 22 of the Denton Development Code (DDC), for example, does not allow drilling by right for land located within a floodway, or within 1200 feet of the flood pool elevation of either Lake Lewisville or Lake Ray Roberts, two local lakes with water rights that are jointly owned by the Cities of Denton and Dallas.

Prior to code amendments in 2004, gas well drilling was not allowed by right within the "100 year" floodplains of Denton. By definition, these areas were considered to be "Environmentally Sensitive" by the provisions set forth in Subchapter 17 of the DDC, and were also prohibited from some filling activities by drainage regulations. During 2003 to 2004, gas well operators began to petition the City Council and City Manager to allow gas well drilling in the flood fringe, which was defined by the City of

Denton as those areas outside of the year floodway but within the 100 year floodplain. When considering this issue, it is important to note that often the mineral rights within the Barnett Shale have been severed from the surface rights, which creates a situation where the mineral owner and surface owner must work together to determine how site drilling will be accomplished. In general, the State of Texas provides substantial rights to the mineral rights owner, and this fact can cause contention between the mineral owner and surface owner when these parties disagree on how drilling should be conducted. In some cases, surface owners view floodplains as areas that are optimal for drilling, since floodplains are unlikely to be residentially or commercially developable. Mineral owners may also view floodplains as good potential drilling areas because locating pad sites in floodplains can minimize interactions between the installed gas well and future development, and may serve to avoid potential development conflicts with the surface owner during the lease negotiations.

Although locating pad sites in the floodplains may offer some advantages from current or future site development standpoints, there are numerous environmental concerns that must be addressed. City of Denton staff researched the feasibility of locating gas well pad sites in flood fringe areas, and produced a series of recommended amendments to the DDC in order to facilitate this type of development under recommended restrictions. To ensure compliance with local floodplain and environmental standards, the City of Denton requires all operators proposing to locate pad sites in floodplains or other environmentally sensitive areas to obtain a Watershed

Protection Permit (WPP). The WPP establishes a series of additional environmental regulations for those gas wells located in the floodplain fringe, as well as requiring an additional fee to cover the expenses associated with site assessments, additional regulatory oversight, and water quality testing. In general, WPPs require a tree survey of the site and tree mitigation at a rate of 1:1 replacement (based on diameter at breast height or "dbh") for 100 percent of the trees removed from the site. Storage tanks and separation facilities that serve a single wellhead can be located in the flood fringe only if these facilities are a minimum of 18 inches above the established base flood elevation, plus a surcharged depth for encroachment to the limits of the floodway that is equal to a maximum of one percent probability of being equal or exceeded in a year. These regulatory restrictions are required to be supported by an engineering study, and must demonstrate that the proposed activity will have no adverse impact on the carrying capacity of the adjacent waterway and will not cause any increase in the elevations established for the floodplain. When the Special Flood Hazard Area on the site in question is designated as "Zone A", or is undesignated, applicants are required to base floodplain calculations on methods outlined in the Denton Drainage Design Criteria manual. Details can be obtained from Subchapter 22 of the Denton Development Code and associated information.

In general, the slope of a given property, the erodibility of the properties soils, and the proximity of that property to surface water conveyances are all important considerations for minimizing gas well impacts to surface water resources. Flat, heavily vegetated areas that are located long distances from surface water resources tend to be less of a concern than those areas close to streams or lakes, located on highly erodible soils with little vegetation, and situated on steeper slopes. Regardless of whether a municipality decides to allow drilling in the floodplain fringe or not, management practices should be designed to ensure that areas with greater storm water / surface water impact potential are managed appropriately.

Tree preservation

Tree preservation at gas well sites in Denton is complicated by the fact that, in many cases, mineral rights have been severed from surface rights in the Barnett Shale. This creates an interesting dynamic for local tree preservation and mitigation strategies. In general, tree preservation and mitigation requirements within the City of Denton are based on the type of development (residential versus non-residential) and the size of the development in question. However, since gas well development is based on a surface lease that is often negotiated between mineral rights and surface owners, there are some discrepancies as to the size of the property considered for mitigation when gas well development impacts trees. In other words, is the size of the property and associated tree mitigation requirements calculated based on tree clearing activities for only the pad site and roads, or is the size dependent upon the size of the property negotiated in the 20-40 acre (usually) lease agreement? The issue is further complicated by the design of the gas well pad site itself, since pads developed in treed areas must remove the trees to facilitate pad development. The need to remove trees to construct the pad, coupled with the discrepancies associated with property size and ownership designations due to the unique surface owner - mineral rights / surface lease relationship cause complications when considering tree preservation. For example, in cases where the mineral owner and surface owner are not in agreement with site development plans, the mineral right owner could, in effect, claim credit for trees preserved on other portions of the surface owner's property that happens to exist within the mineral owners lease, and thus avoid paying mitigation fees for any trees removed during gas well development activities. The ability for the surface owner to develop his property in the future could thus be influenced by tree preservation decisions made by the gas well developer.

In the interest of promoting equitable tree preservation for both the surface and mineral owners, the City of Denton chose to require mitigation for all gas well pads and associated clearing activities associated with roads, pipelines, etc.. Gas well developers are thus required to mitigate at a rate of 25 percent for all trees removed from the property, and mitigation is required to be in the form of payments to the City of Denton's tree fund instead of mitigation through on-site planting. This approach ensures that the gas well developer pays an appropriate fee for the negative impacts to treed areas, and ensures that the surface owner is not penalized for tree removal activities associated with the gas well lease. Details for this process can be found in Subchapter 13 of the DDC.

Tree and vegetation removal for floodplain and other environmentally sensitive areas requires additional considerations. In these areas, the removal of trees may have minimal direct influence on future surface development of the site, but may represent a loss of critical habitat and negative influence on surface water resources. To ensure that tree removal in floodplains and other environmentally sensitive areas is minimal, the City of Denton requires tree mitigation for all of these areas at a rate of 1:1 replacement value for all trees removed from the site, regardless of species or size. Mitigation is required to be accomplished through planting replacement trees within a floodplain, either on site or off-site, or by payment into the tree fund at the current rate of $125 per caliper inch of tree removed. The 1:1 mitigation rate tends to become very expensive if more than a few trees are removed, and requirements for newly planted trees in terms of irrigation and survivability tend to be fairly onerous. For these reasons, very few gas well developers have developed within treed floodplain sites.

Site Design, Construction, and Management Considerations

Site design, construction, and management options are important considerations when attempting to minimize environmental impacts. The body of research contained within this report suggests that gas well pad sites can create significant erosion and sedimentation concerns that are comparable to typical residential or commercial construction sites. As discussed within this document, construction sites are regulated through a series of storm water programs at the Federal, State, and local levels. Gas well development operations, however, have been categorically exempted from Federal and State storm water regulations. Currently, only local regulations can be imposed by municipalities to deal with storm water concerns. Based on the research contained within this document, it is recommended that all municipalities strongly consider adding both erosion and sediment control provisions to local codes. Sediment impacts from gas well development and production sites can be substantial if unmanaged and unregulated.

Although gas well pad sites have been demonstrated to produce storm water runoff containing substantial amounts of sediments, research in this document suggests that there are efficient, effective ways to control impacts from sediments in storm water runoff from gas well pad sites. Research efforts in the latter phases of the grant were directed towards producing simple optimization models that allow gas well developers and municipal regulators the ability to find the most cost effective combination of management practices for dealing with erosion and sedimentation at gas well pad sites while meeting a minimum management goal. Although details of these approaches are contained within the research presented elsewhere in this document, general summary statements are possible. In general, utilizing both erosion and sediment control methods at sites tended to give the best combination of control and cost. However, the best combination of these types of management practices is heavily influenced by the type of soil and slope. Currently, the City of Denton requires compost berms to be installed around the down slope portion of gas well pad sites. However, during the course of this research, the project team found that compost berms by themselves may not be the best management strategy. Newer technologies such as compost "socks" tended to offer more stability, ease of installation, and durability when compared to traditional compost berms. It was also apparent that additional management practices were needed to ensure site stabilization from both erosion and sediment control perspectives, not just sediment controls alone. Based on these findings, the City of Denton staff intends to modify existing DDC regulations to require both erosion and sediment management practices for gas well development sites. Recommendations for the type of management practices will center on the use of both erosion and sediment control strategies, and will rely on the previously mentioned optimization approaches to determine the best combination of performance and cost.

Research demonstrated that total petroleum hydrocarbons are a concern for mud pits, although benzene, toluene, ethylbenzene, and xylene concentrations tended to be relatively low. Results are summarized in Section 6. Because of TPH concerns, sampling mud pits and establishing municipal regulatory standards may be warranted. However, mud pit contents are complex and did not appear to be amenable to analyses via rapid "field-based" methods like Fiber Optic Optic Chemical Sensor (FOCS) technologies or rapid laboratory methods such as Enzyme Immunoassays (EIAs) when compared to solvent based extraction methods followed by GC-FID analyses. If a municipality considers developing mud pit standards for TPH and BTEX, thought should be given to these analytical challenges. In the absence of enforceable standards for the pit contents themselves, municipalities should at least consider pit design standards that will minimize the likelihood of pit contents escaping the mud pits and potentially impacting surrounding areas. Design considerations include the use of liners, construction of pits in areas where concentrated sheet flow will not cause the pits to be overtopped, requiring the use of freshwater based muds only, and maintenance of minimum freeboard distances between the elevation of the pit contents and the elevation of the top of the mud pit dam. Mud pits should only be allowed on the site as long as needed for the drilling operation, and should be removed as soon as possible after drilling is completed. Other options, such as closed loop drilling, could be imposed to eliminate the use of open mud pits altogether.

Site operation standards can be used to create a cleaner overall site, which should provide benefits to both workers as well as minimizing impacts to storm water and site soils. It is important to note that the sampled gas well pad sites assessed in this study generally showed relatively low concentrations of contaminants in runoff, but appeared to be influenced by site operations. With this in mind, it is recommended that municipalities consider simple site management standards for incorporation into local regulatory requirements. For example, drip pans or oil absorbing materials should be placed underneath all tanks, containers, and other equipment that has a potential to leak. Chemical materials should be stored on pallets or other appropriate devices to prevent contact between the ground and containers, and should be protected from storm water and other weather elements. Depending on the type and quantity of materials, secondary containment and other similar strategies may be appropriate. A hazardous materials management plan should be created for all sites, and all materials should be adequately labeled, contained, and have appropriate material safety data sheets available. The overall goal for the site should be to devise a plan that ensures that all chemical materials can be stored as safely as possible on the site, and any accidental spills, leaks, or discharges of materials can be remediated as quickly and safely as possible.

<u>Monitoring Considerations</u>

Monitoring gas well operations can produce valuable information on site characteristics and loadings, as well as providing the information needed to assess contamination and pursue enforcement actions. However, municipalities responsible for regulating gas well operations may not have the resources, equipment, or expertise necessary to implement a large scale monitoring program. The research contained in this document demonstrates that there is a need for certain types of municipal regulations and potentially a need for municipal monitoring. Monitoring storm water runoff from small scale gas well pad sites can be difficult, and requires a relatively large amount of complex equipment to accomplish the volume based sampling needed to assess site loadings. However, much of the concerns associated with storm water runoff are related to erosion and sedimentation issues, which can be dealt with through regulations, appropriate erosion and sediment management practices, and municipal inspection by trained individuals. In most cases, impacts from sediment leaving a site are visually apparent, and can be thus regulated through site visits and the application of properly designed municipal regulations. Management practices similar to those used at residential and commercial construction sites are often sufficient to meet target sediment reduction goals.

Monitoring mud pits may be warranted, especially for excessive concentrations of total petroleum hydrocarbons. However, as noted within this document, monitoring mud pit contents can be difficult, and should be done using standard laboratory procedures involving extractions followed by GC-FID. Field monitoring devices such as FOCS or "quick" laboratory methods such as EIA did not perform well for mud pit samples. Because of these issues, municipalities interested in monitoring mud pit contents should plan to have samples analyzed by an outside laboratory if the municipality does not have the ability to perform extractions and GC-FID analyses in house. These analyses can be costly, and should be considered for addition to gas well permit fees. Overall, many of the concerns related to mud pit contents can be minimized through properly designed regulatory standards and municipal oversight. With properly designed regulations, municipal inspection, and proper enforcement capabilities, mud pits can likely be regulated so that impacts are minimized without conducting routine sampling of the mud pit contents themselves.

The City of Denton, Texas has an active watershed monitoring program that conducts spatially and temporally dense sampling of Denton's surface water resources.

Although this monitoring program is not a component of USEPA CP-83207101-1, the interaction between this monitoring network and gas well issues does warrant discussion. The watershed protection program sampling program is comprised of approximately 70 monitoring stations that are dispersed across the watersheds of the City of Denton. These stations are sampled once a month during periods of normal flow, and resulting samples are analyzed for a wide variety of constituents.

On several occasions, this monitoring network has been useful for detecting contamination from gas well pad sites, often when contamination was not readily apparent from visual inspections of the sites or in times between gas well pad site assessments. In these cases, simple specific conductance (“conductivity”) of the water samples in receiving streams indicated possible contaminations events. Since specific conductance can be analyzed using inexpensive field meters, technicians were able to trace the contamination back to the source by simply walking upstream and periodically analyzing water samples. In many cases, mud pit discharges and fracturing water discharges have appreciable amounts of dissolved salts, which can be easily detected using conductivity meters. As long as releases from gas well pad sites are not highly diluted by existing stream flows, impacts from discharges can be detected using standard field conductivity meters.

If resources allow, municipalities should consider regular assessments of surface water conductivity in areas where gas well pad sites are located. However, because of dilution effects the spatial scale of monitoring and overall stream sizes should be maintained as small as resources allow. Measuring water quality in smaller streams that are in close proximity to gas well operations provides the best opportunities for detecting pollution and being able to trace this pollution back to the source. Analysts should also have a good understanding of the natural variability of conductivity, as values can vary widely due to precipitation, groundwater / surface water interactions, snow melt, road icing operations, and other similar influences. However, due to the ease of instrument operation, relatively low instrumentation costs, and the potential ability to detect gas well discharges, regular receiving water monitoring using conductivity should be strongly considered.

Conclusions

The regulation of gas well drilling and production operations is complex, and should be designed to ensure adequate public and occupational safety issues as well as ensuring environmental impacts are minimal. Subchapter 22 of the Denton Development Code (www.cityofdenton.com) contains a substantial amount of information that may be useful to municipalities interested in establishing local gas well regulations. At minimum, environmental regulations should establish criteria for site locations, set standards for tree preservation, and establish minimum standards for erosion and sediment controls. Since gas well pad sites have the potential to produce storm water sediment loads that are comparable to traditional construction sites, these sites should be required to implement erosion and sediment control practices as standard components of site development. Pad sites also have the potential to produce other contaminants associated with equipment and general site operations. However, proper site management, equipment maintenance, and hazardous materials management and containment can help minimize these sources of contamination. Mud pits and, to a lesser extent, fracture water pits deserve special consideration and management. Although a regular monitoring program coupled with associated regulatory standards may be the best way to ensure that pollution potential is minimized for these pits, municipalities may not have the staff, resources, or expertise to implement such a program. However, implementing operational standards such as requiring pit liners, maintaining adequate freeboard for pit contents, restricting locations to relatively flat slopes, and designing pits so that they do not capture large amounts of rain water can all aid in minimizing the potential release of contaminants without necessarily relying on a regular monitoring program. Regular watershed monitoring using specific conductance can, under the right circumstances, offer a relatively inexpensive and rapid method for detecting contaminant discharges and tracing these discharges back to the source.

APPENDIX A. COLLECTING STORM WATER FROM OVERLAND FLOW AT GAS WELL EXPLORATION AND PRODUCTION SITES

Research Design and Methods

Study Area and Site Description

The City of Denton is located in north central Texas, at latitude N 33° 12' 49.7" and longitude W 97° 09' 03.4". Basin topography varies from level to gently rolling, while physiography is divided between the Grand Prairie and Eastern Cross Timbers regions. The gas well exploration and productions sites under consideration for this study were all located in the Grand Prairie region of the Hickory Creek watershed, which drains into Lake Lewisville (Figure 1). Lake Lewisville is a source of drinking water for a large population of north central Texans. Annual normal rainfall is approximately 99 centimeters, the majority of which normally occurs during the spring months of April through May and the fall months of September through October.

Gas well sites in this area are all similarly constructed, comprised primarily of a tightly packed rock pad approximately one acre in size surrounded by an additional two to four acres of disturbed soil. Each site drains an area of approximately two to four acres. The slope of a typical site ranges from 3 to 5 percent. Soils in the area generally consist of a clay layer near or at the surface and have high runoff potential due to slow infiltration rates. The layout of a typical site is illustrated in Figure 2.

Site Selection

Prior to site selection local operators were solicited for participation in the study. Of the eight that were asked to participate in the study, only one expressed interest. However, because the operator expressing interest happened to be one of the largest in the United States and had the greatest presence in the area, the site selection team was comfortable exclusively using their sites for the study. The team felt that based on the company's reputation this would be a best-case scenario. Furthermore, having all the sites constructed and operated by the same company would help to minimize site-to-site differences. The site selection team included hydrologists, oil and gas industry representatives and water quality experts. Five sites were to be constructed within first few months of the three-year project (Figure 1), three of which were to be chosen for the study. In choosing the sites, the team evaluated topography, soil type and the means of adjacent areas to be used for reference sites.

Topography was initially characterized by field visits. Sites were assessed according to their slope, runoff directionality and similarity. Sites that naturally drained to one down slope portion of the site were better suited for the study. Site 4 was eliminated from consideration due to the potential run-on from upslope of the site. Site 5 was questionable due to its relative flatness; it was unclear whether or not there would be one down slope area to where the site would drain. Geographical Information Systems were employed as a means to delineate drainage areas for the reference sites and provide detailed soil information without sampling the sites. Soil information used to conduct the analysis was obtained from the Natural Resources Conservation Service (NRCS) Soil Survey Geographic (SSURGO) database. SSURGO is the most detailed level of soil mapping done by the NRCS and consists of map data, attribute data, and metadata. The five sites consisted of only two different soil types. After reviewing soil parameters important to the study, such as erodibility, water capacity and some of the more common chemical characteristics, it was determined the difference in the two soil types was minimal. Finally, the sites were evaluated for their ability to provide a suitable reference site. In addition to its questionable runoff directionality, Site 5 did not have a location that would be appropriate in size for a reference site and was also eliminated from consideration. Therefore, Site 1, 2 and 3 were chosen for the study.

Two reference sites were also included in the study, one of which was located between Site 1 and Site 2, which are only 1000 meters apart. The other reference site was located adjacent to Site 3. Reference Sites 1 and 2 are approximately 7 to 8 acres in size. Larger reference sites were necessary to generate enough runoff for sampling, as the runoff characteristics of the reference sites are different than the gas well sites. Reference Site 1 is located in a densely vegetated prairie dominated by native grasses. This site has a much higher infiltration rate due to uptake from the vegetation and a higher storage capacity. Reference Site 2 is located in rangeland covered with thickets of mesquite.

Weir Design

Following site selection, a collection system was designed and constructed. Components of the system for collecting overland flow included a structure to concentrate and measure flow and a means to convey runoff into the structure. Measuring discharge with a structure is based on the relationship between water level and discharge. As water passes through and opening of a known size and shape, the height of water flowing through the opening is used to calculate flow. The two most commonly used devices to concentrate and measure flow are flumes and weirs. Weirs were chosen for this study because they are the simplest to build and install, inexpensive, and highly accurate (Driscoll, 1986). One disadvantage to using a weir is that they are not self-cleaning. Sediment and debris must be removed from the weir after storm events.

A sharp-crested 90^0 v-notch weir was selected for the gas well sites because it provides accurate measurements at both low and high flows. Sharp-crested weirs are most commonly used in small catchments and are especially suited to accurately measure low flows (0.0001 m^3/s) and small changes in discharge (Maidment, 1993). Accurate measurements at both low and high flows are important as storm events in the area can vary dramatically with regards to runoff characteristics. Moreover, during a dry year the number of storm events available for sampling may be limited. For the smaller storms the device must be able to measure flows as low as 0.0001 m^3/s whereas for larger storms flow may be as high as 0.1237 m^3/s. Weirs were built according to specifications for a partially contracted weir (DOI, 1997). Because the sites are relatively small and flat and will experience relatively small volumes of runoff, the minimum depth below the v-notch of 1.5 feet for the fully contracted weir could not be met. Weir materials consisted of wood and PVC. To protect the weir from the elements, the outer shell was coated with a marine grade gel-coat to seal the wood from moisture and the inside of the weir was covered with a PVC material. In a previous study, PVC was found not to contaminate storm water samples (Schleppi, 1998).

The size of the weir was determined based on the modeled peak discharge of the drainage area. The rational method, a well-known rainfall-runoff model was utilized to estimate peak discharge from the study sites. The model is simple but appropriate for estimating peak discharges for small drainage areas (Chow, 1964). The formula for the rational method is as follows:

$$Q = F\ C\ I\ A$$

where Q = maximum rate of runoff (cfs or m^3/s); C = runoff coefficient or fraction or rainfall that becomes runoff; I = average rainfall intensity (in./hr. or mm/hr.); and, A = drainage area (acre or hectare). F is conversion factor that is usually omitted when English units are used, but for metric units F equals 0.278. Selecting the runoff coefficient should include consideration of the nature of the surface, surface slope, surface storage and the degree of saturation (Gray 1970). Runoff coefficients for urban areas, as recommended by the American Society of Civil Engineers and Water Pollution Control Federation are provided in the Handbook of Hydrology (Maidment 1993); agricultural runoff coefficients can be referenced in the Handbook of the Principles of Hydrology (Gray, 1970). Rainfall intensity is equal to the time of concentration and is estimated from rainfall intensity-duration-frequency (IDF) data. The time of concentration is the time at which the entire drainage area begins to contribute to runoff. Small areas with short times of concentration could result in rainfall intensities that are unrealistically high because the IDF relationship is applied assuming that the duration is equal to the time of concentration. As rainfall duration tends toward zero, the rainfall intensity tends toward infinity. Therefore, a minimum time of concentration of ten minutes is recommended (TXDOT, 2004). Specific IDF data for Texas counties are available in the Hydraulic Design Manual (TXDOT, 2004) where intensity takes the form of the following equation:

$$I = b / (t_c + d)^e$$

where I = rainfall intensity (in./hr. or mm/hr.); t_c = time of concentration (min); and, e, b, d = coefficients for specific IDF relationships listed by county. Based on local rainfall data acquired from the Rainfall Frequency Atlas of the United States (Hershfield, 1961), the weir was sized to accommodate a 2-year 24-hour rainfall event. Collecting storm water from an event of this size was thought to be reasonable based on the scale of these sites.

Once peak discharge volumes were calculated and the weirs were sized, the structures were constructed and placed near the edge of the each site at the lowest elevation. To convey all the runoff from the site through the weir a structure similar to a silt fence, but with a thick impermeable material, was installed along the down slope portion of the site. To minimize water loss, the fencing material was put in a 6-inch trench and backfilled. At the weir, the fencing material overlapped the sides of the weir and was held in place with heavy gauged staples. The pressure of the water against the material helps to create a tight seal.

Storm Water Sampling

It is recommended that prior to the implementation of a sampling program some information pertaining to runoff characteristics be obtained and studied (King and Harmel, 2003). Automated storm water sampling generally requires setting both a minimum flow threshold and an appropriate sampling interval. The minimum flow threshold for this study was based on recommendations from previous research (Harmel et al., 2002). The appropriate sampling interval was estimated according to a site-specific rainfall-runoff ratio. The rainfall-runoff ratio was then used to back calculate a runoff curve number (CN). Once the CN value was obtained, the runoff depths for a range of storms were calculated according the Soil Conservation Service (SCS) Runoff

Curve Number method (Cronshey, 1986), also referred to as the NRCS method. Although standard coefficient values can be applied to simple models to estimate runoff, site-specific rainfall and runoff data is almost always more effective in determining runoff characteristics of a specific study area. The following methodology illustrates a procedure used in estimating the appropriate sampling interval for collecting storm water samples from small sites.

Minimum Flow Threshold

Prior to determining a sampling interval, automated storm water sampling requires setting a minimum flow threshold for beginning and ending sampling. This is a crucial number, since minimum flow thresholds set too low will enable sampling for small storms in which no significant pollutant loads will be transported. However, if flow thresholds are set too high, an entire event or a substantial portion of an event will be missed. Increasing flow thresholds has been shown to result in increased error of the true or total pollutant load (Harmel et al., 2002). For small watersheds, errors were shown to be substantial even for small increases in minimum thresholds. For this reason, Harmel and others recommend setting minimum flow thresholds of 0.001 to 0.04 m^3/s (0.1067 to 0.4297 m head). However, these recommendations were based on observations obtained from watersheds ranging from 6 to 67 hectares. Drainage areas of gas well sites are substantially smaller (~1 hectare) and will produce much less runoff.

Based on the size of the gas well sites, the minimum flow threshold was set at 0.0004 m^3/s (0.0381 m head).

Flow Interval Sampling

Once the minimum flow threshold has been established the next important consideration is whether to sample based on a time or flow-interval. Flow-interval sampling has been shown to better represent storm loads because more samples are taken at higher flow rates (Harmel, King, and Slade, 2003). In addition, the Event Mean Concentration (EMC) of a storm event can be easily averaged from flow-interval samples. The size of the flow-interval is also an important factor in the characterization of the storm. Although smaller sampling intervals better represent storm loads, they also result in a higher number of samples and thus an increasing amount of analytical time and cost. However, concentrations from flow-interval sampling can easily be composited to reduce the total number of sample while still retaining the necessary resolution for appropriately characterizing the storm event. In all cases, it was is imperative that the sampling interval be set in such a manner as to capture a wide range of storm event sizes, as weather conditions are unpredictable. Setting the appropriate interval is not a trivial task, and requires a good understanding of site conditions in order to sample efficiently. For example, if the sampling interval is set too high, too few samples will be collected to sufficiently characterize the storm. On the other hand, if the sampling interval is set too low, the number of samples will exceed the capacity of the sampler and a portion of the storm will not be sampled.

In order to estimate an appropriate flow-interval the volume of runoff was approximated. Although runoff can be estimated using the rational method, as discussed above, the NRCS method (Cronshey, 1986), commonly referred to as TR-55, may provide more accurate results. While it is also a simple model, the method is somewhat more sophisticated than the rational method as it considers initial rainfall losses due to interception and storage and takes into account an infiltration rate that decreases during the course of the storm. The NRCS method estimates the depth of direct runoff (in./mm) resulting from a given rainfall amount. Heavily influencing the model, the CN value represents the likelihood that rainfall will become runoff, a higher value results in more runoff. For urban areas CN values range from 39 to 95. A CN value of 65 is recommended for construction sites (City of Houston, 2001). One of the initial challenges of this project was to obtain an appropriate CN estimate for the gas well sites. A gas well site is unique in that it partially represents a construction site, but is also largely semi-impervious as the pad site is comprised of tightly packed crushed rock that is designed to drain efficiently. Instead of using the recommended CN value of 65 as recommended for construction sites, the CN value was back calculated based on rainfall-runoff data from a recent storm event.

Rainfall data was compared to the measured runoff at Site 3. At the time of the event Site 3 was the only location constructed and equipped with monitoring equipment. The ISCO storm water sampler used to collect the data was equipped with a pressure transducer that measures the water level (head) above it. The head measurement is automatically converted by the device to a volume according to the volume discharge relationship of a 90^0 v-notch weir. During the event, rainfall and runoff data were collected at fifteen-minute intervals. Rainfall and runoff measured approximately 37.5 mm and 182.37 m^3, respectively. Based on the area of the site, 182.37 m^3 of runoff is equivalent to 22.5 mm of runoff. The formula for the NRCS method is:

$$Q = (P - 0.2S)^2 / (P + 0.08S)$$

where Q = runoff (in. or mm); P = rainfall (in. or mm); and, S = potential maximum retention after runoff begins (in. or mm). S is related to the soil and cover conditions of the drainage area through the CN value by:

$$S = z\ (100 / CN - 1)$$

where z = 10 for English measurement units, or 254 for metric; and, CN = runoff curve number. When the discharge of a storm event is known, the CN value can be back calculated. Substituting 37.5 mm for P and 22.5 mm for Q, S becomes 17.65; Substituting 17.65 in the equation above, the CN value for gas well Site 3 becomes 93.5.

Referencing Table 2.2a in TR-55, a CN value of 93.5 compares closely to gravel roads (CN = 91) or newly graded developing urban areas (CN = 94) for hydrologic soil group D, which is based on the soil condition at the site. Note the substantial difference in the data derived CN value of 93.5 compared to the suggested value of 65 for construction sites. Predicted runoff volumes derived from the two values differ by approximately three orders of magnitude. Had the CN value of 65 been used in the following sampling methodology the majority of storm event discussed later in this paper would not have been sampled. Once the CN value had been established, the NRCS method was used to develop a table providing the depth of runoff for a range of rainfall amounts.

The sampling interval was based on the volumetric depth of runoff (flow-interval), which is the amount of runoff (measured in mm) that will result from a given amount of rainfall. Volumetric depth is a useful sampling metric as it results in sampling proportionate volumes of rainfall and runoff from multiple sites regardless of the size of the drainage area. Harmel and others (2003) summarize the number of samples taken at various flow-intervals for 190 storm events. The data provide guidance on selecting flow-intervals for watersheds less than 1000 hectares. However, since site-specific information was known, the NRCS method was used to calculate runoff depths for storm events ranging from 6.25 to 75 mm or rainfall (Table 1). Three different volume based sampling intervals were considered: 0.5 mm, 1.0 mm, and 2.0 mm. Sampling according to the 0.5 mm flow-interval would result in effectively sampling smaller storm events, but for larger events, the 24 sample capacity of the device would be exceeded early in the storm. Conversely, a 2 mm flow-interval would adequately sample large storms, but could potentially result in collecting few, if any, samples during small storms. Sampling based on a 1 mm flow interval seemed to be the most appropriate choice but there was still a chance that the tail end of a very large storm would not be sampled. Sampling large storms is important for characterizing annual loading, as previous research has shown that as little as three to six storm events per year create as much as 75 percent of the storm runoff and nonpoint source loads (Tate et al., 1999).

The conundrum of choosing between adequately sampling both small and large storm events can be resolved by compositing multiple flow-interval samples into one bottle. A typical ISCO storm water sampler has 24-1000 ml bottles. To be sure to sample both the small and large events, the sampler was programmed to draw 18-1000 ml discrete samples, then composite 5-200 ml samples in each of the remaining six bottles. This would provide a sufficient number of samples to characterize the smaller storms, while also increasing the number of samples allowing for complete sampling for relatively large storms.

Therefore, precipitation, runoff volumes and target sample numbers were used to establish the flow-interval of 1.0 mm. This value is then converted to a runoff volume based on the area of the site and programmed into the ISCO sampler. For a gas well site 2-acres in size, 1 mm of runoff is equal to approximately 8.5 m^3. Under this scenario, once enabled by the minimum flow threshold, the sampling device will continually measure the flow and will draw a sample for every 8.5 m^3 of runoff.

Results

Storm water samples were collected for a rain event that occurred on June 1, 2005. Precipitation at Site 1 and Site 3 were recorded at 20.75 mm and 29.25 mm, respectively. No precipitation was measured at Site 2; the gauge malfunctioned. Storm water samples were collected at Sites 1 and 3 (Figure 4 and 5) but at Site 2, the sampler was enabled, collected one sample, then failed due to extreme sediment loading in the weir. Two factors are thought to contribute to the substantial sediment loading at Site 2. One, Site 2 is steeper than the other sites, and two, the placement of the weir is closer to steepest portion of the site allowing less distance for the sediment to settle out of the runoff before reaching the weir. Raising the height of the intake device may resolve this problem. Neither reference site generated a measurable amount of flow; therefore, the ISCO samplers were not enabled at the reference sites. It was anticipated that the difference in vegetation, infiltration and storage of the reference site drainage areas would result in substantially less runoff; however, to experience no runoff from this event was not expected. The reference sites will be relocated to allow for a larger drainage area.

The volume of runoff at Site 1 (205.63 m^3) substantially exceeded modeled estimates according to 20.75 mm of rainfall. This difference could have been related to a number of factors. Inspection of the site after the event revealed that the installation of a gas gathering line servicing the site altered the upslope hydrology of the site, which may have potentially resulted in two acres of additional runoff influencing the site. Also, the area around the site appears to contain water outcroppings; upslope infiltration may have percolated out at the site and contributed to a portion of the measured flow. Finally, a rainfall event a few days prior had partially saturated the area. These factors will all be considered in subsequent analyses of runoff at this site. The ISCO sampler successfully collected 16 samples before failing on sample 17 though 24. The sampler attempted to sample, but no volume of water was collected.

Runoff volumes at Site 3 more closely matched the predicted volumes of the model. The model predicted 125 m^3 of runoff based on 29.25 mm of precipitation. The device measured 111.05 m^3 of runoff. The ISCO sampler attempted to collect 13 samples, but failed on the first sample. The cause of failure for either of these sites is unknown. Based on measured runoff versus predicted runoff the model appeared to overestimate the runoff volume at Site 3. However, post storm site observations revealed evidence that the conveyance/containment system had been partially breached. It was estimated that as much as 20-30 percent of the runoff did not flow through the weir. This loss also appears to be illustrated in the data as Site 3 drained much faster than Site 1. Adjusting runoff volumes based on post storm site observations, assuming two additional acres of runoff at Site 1 and 20-30 percent loss of runoff at Site 3, it appears that the modeled runoff underestimated flows by as much as 20 percent. However, the estimate was useful for “ball parking” runoff volumes and developing initially sampling strategies.

Total Suspended Solids (TSS) Concentrations

Summary statistics for TSS concentrations at each site are shown below in Table 1. For Site 1, the EMC was 236 mg/l. The highest TSS value was captured during the first sample. After the initial high value, TSS concentration dissipated quickly then increased again when flows increased (Figure 5). The flow/concentration relationship at Site 1 was different than Site 3, as the concentrations seemed to dissipate rather quickly after the initial first flush whereas the dissipation of TSS as Site 3 was comparably slower (Figure 6). However, the data illustrates that the relatively small increases in flow were strong enough to re-suspend the sediments, suggesting that small increases in flow can dramatically influence the loading of sediments. Total sediment load is calculated by multiplying the EMC with the total flow volume. Total sediment loading from this storm event for Site 1 was estimated at approximately 49 kg. Prior to the sampler failing, approximately 70 percent of the storm event was sampled. The EMC was not adjusted to account for percentage of the storm not sampled, therefore total loads for Site 1 may have been over estimated. For Site 3, TSS reached a maximum concentration of 536 mg/l. The EMC was 241 mg/l. Compared to Site 1, TSS concentrations at Site 3 appear to more closely represent the pattern of the hydrograph (Figure 6). This may be attributed to disturbance of the area from the installation of the gas gathering line at Site 1. Based on the adjusted volume accounting for the estimated loss of runoff through the containment system, total sediment loading for Site 1 was estimated around 37 kg.

EMC concentrations for TSS from this storm event are higher than average concentrations of TSS values reported for residential, commercial, industrial, and non-urban land uses (USEPA, 1983). TSS values reported in the NURP study ranged from 67 to 101 mg/l. However, the TSS concentrations measured here do to not seem to be as high as values reported in construction site studies. Based on data from construction site runoff in Maryland, Schueler and Lugbill (1990) reported median TSS concentrations of 680 mg/l. TSS concentrations sampled from 72 storms at 15 highway construction sites in California ranged from 12 to 3850 mg/l with a mean of 499 mg/l (Kayhanian et al., 2001). Nelson (1996) evaluated 70 construction site samples in Birmingham and measured suspended solid concentrations ranging from 100 to over 25,000 mg/l, with a median of about 4000 mg/l. Lower TSS concentrations at gas well sites could be due to the intensity of the storm as suspended solid concentrations have been shown to vary dramatically based on rainfall intensity (Nelson, 1996). The rainfall intensity of this event is not considered to be high. Perhaps another explanation is the difference between a construction site and a gas well site. A gas well site is largely comprised of a tightly packed gravel surface. Gravel is rougher (higher Manning's coefficient) than soil and the particles are heavier, thus it would take much more energy pick up sediment and move them across the gas well sites. Although some comparisons can be made, more storms are needed to adequately characterize sediment loading from these sites.

Acknowledgments

The authors would like to thank Daren Harmel with the USDA-Agricultural Research Service in Temple, TX for his guidance on flow-interval sampling methodologies.

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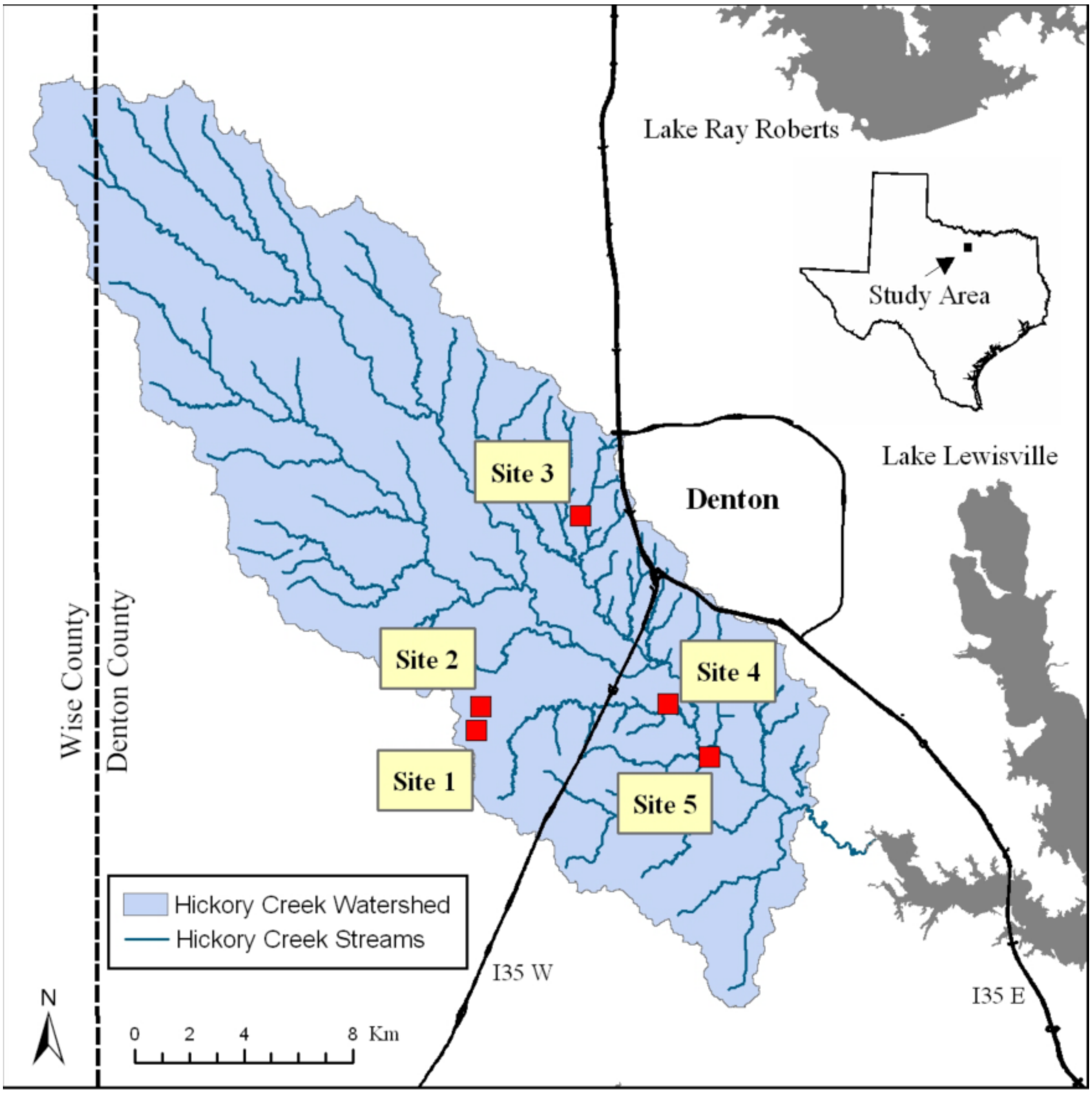

Figure A.1. – Study Area Site Map

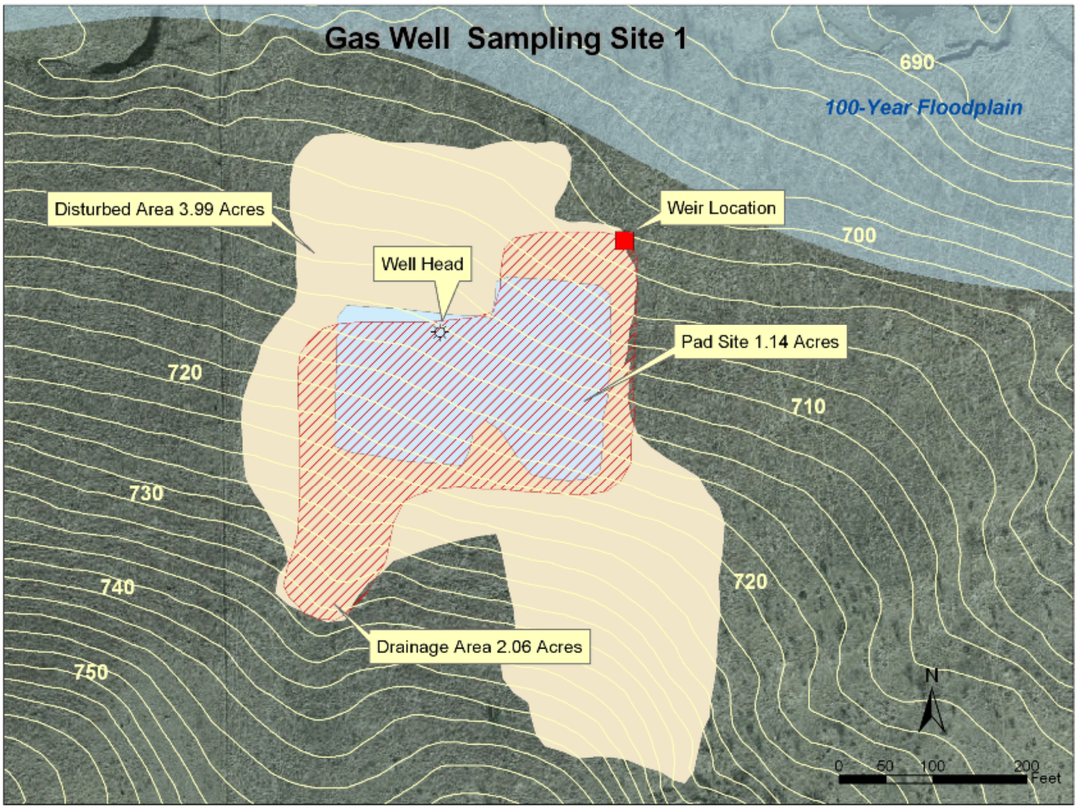

Figure A.2. – Typical Gas Well Site Layout

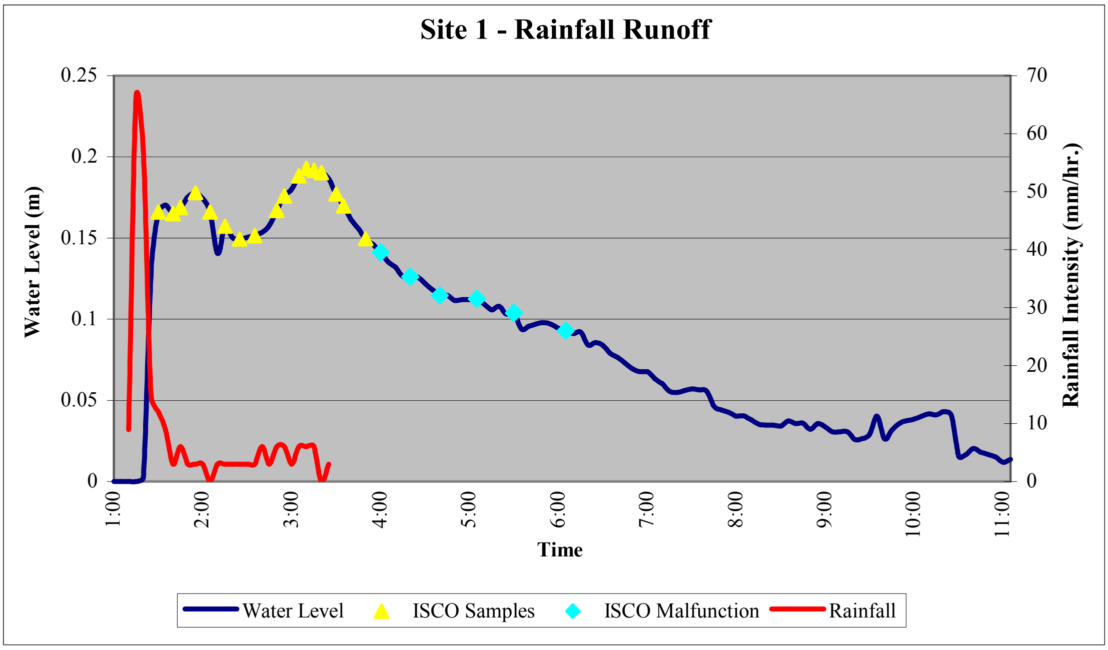

Figure A.3. – Site 1 Rainfall, Runoff, and Sample Times

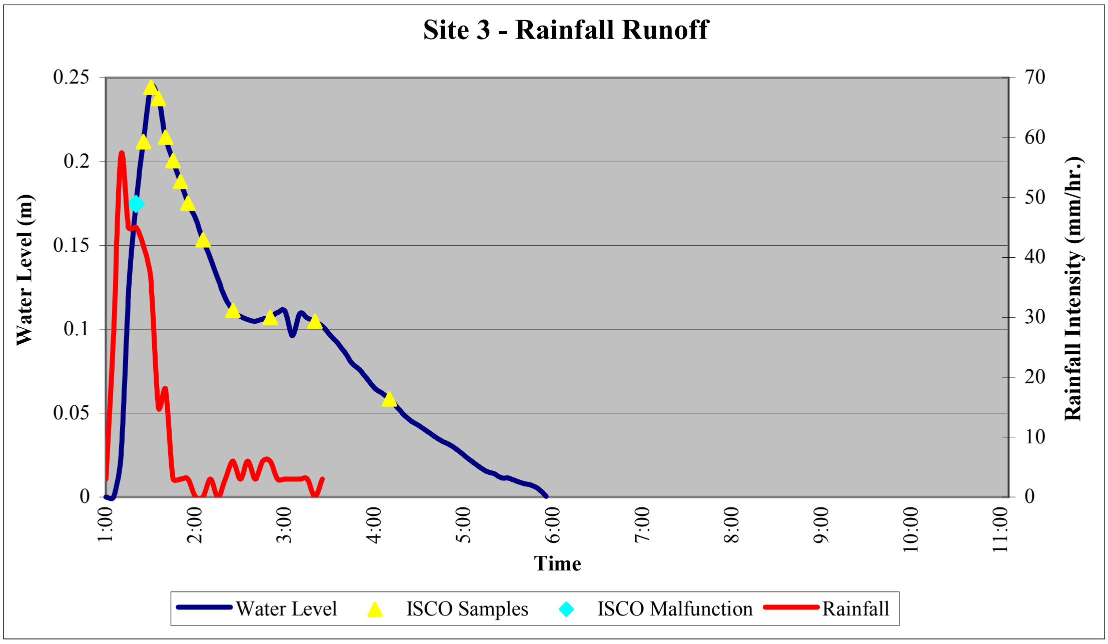

Figure A.4. – Site 3 Rainfall, Runoff, and Sample Times

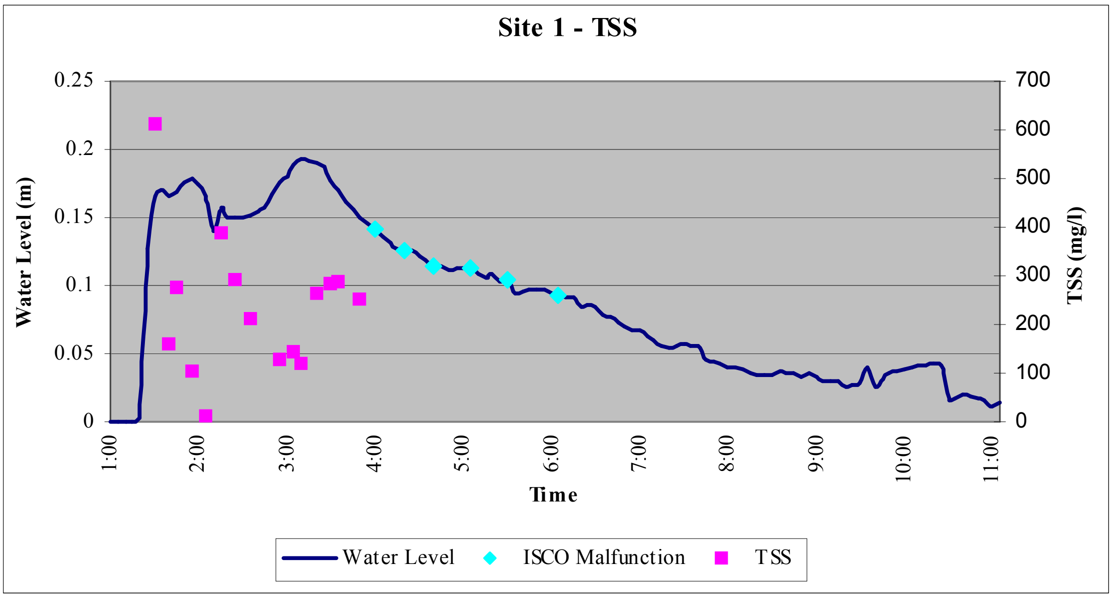

Figure A.5. – Site 1 TSS Concentrations

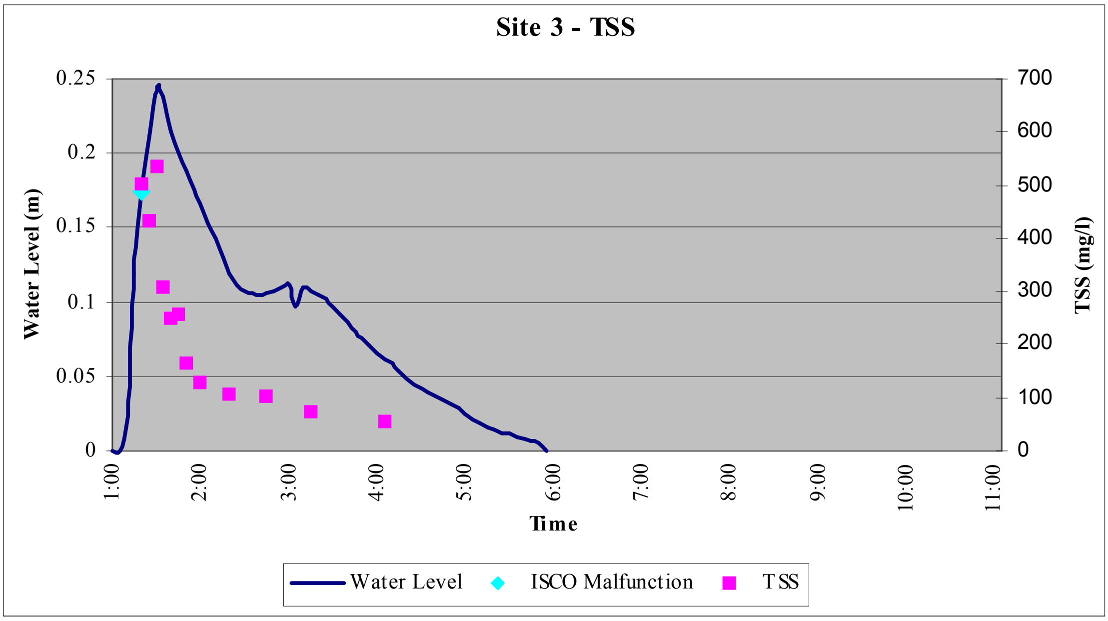

Figure A.6. – Site 3 TSS Concentrations

Table A.1. Number of Samples Based on Volumetric Depth of Runoff

		Number of Samples			Total Volume
Rainfall(mm)	**Runoff (mm)**	**0.50 mm**	**1.00 mm**	**2.0 mm**	**(m^3)**
6.25	0.38	0.76	0.38	0.19	3.09
12.50	3.08	6.00	3.00	1.54	24.98
18.75	7.14	14.00	7.00	3.57	57.86
25.00	11.91	23.00	11.00	5.95	96.44
37.50	22.52	45.00	22.00	11.26	182.38
50.00	33.87	67.00	33.00	16.94	274.31
62.50	45.60	91.00	45.00	22.80	369.28
75.00	57.54	115.00	57.00	28.77	466.01

Table A.2. – TSS Concentrations

	Site 1	**Site 3**
Minimum (mg/l)	14	56
EMC (mg/l)	236	241
Maximum (mg/l)	612	536
Median (mg/l)	251	195
No. Samples	16	12

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

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 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 United States (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). 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 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 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).

<u>Analytical Procedures</u>

Water quality parameters analyzed under the monitoring program, along with the analytical methods and detection limits, are summarized in Table 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:

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

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).

<u>Data Analysis</u>

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 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 4. Table 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 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 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 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 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 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 South Eastern 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).

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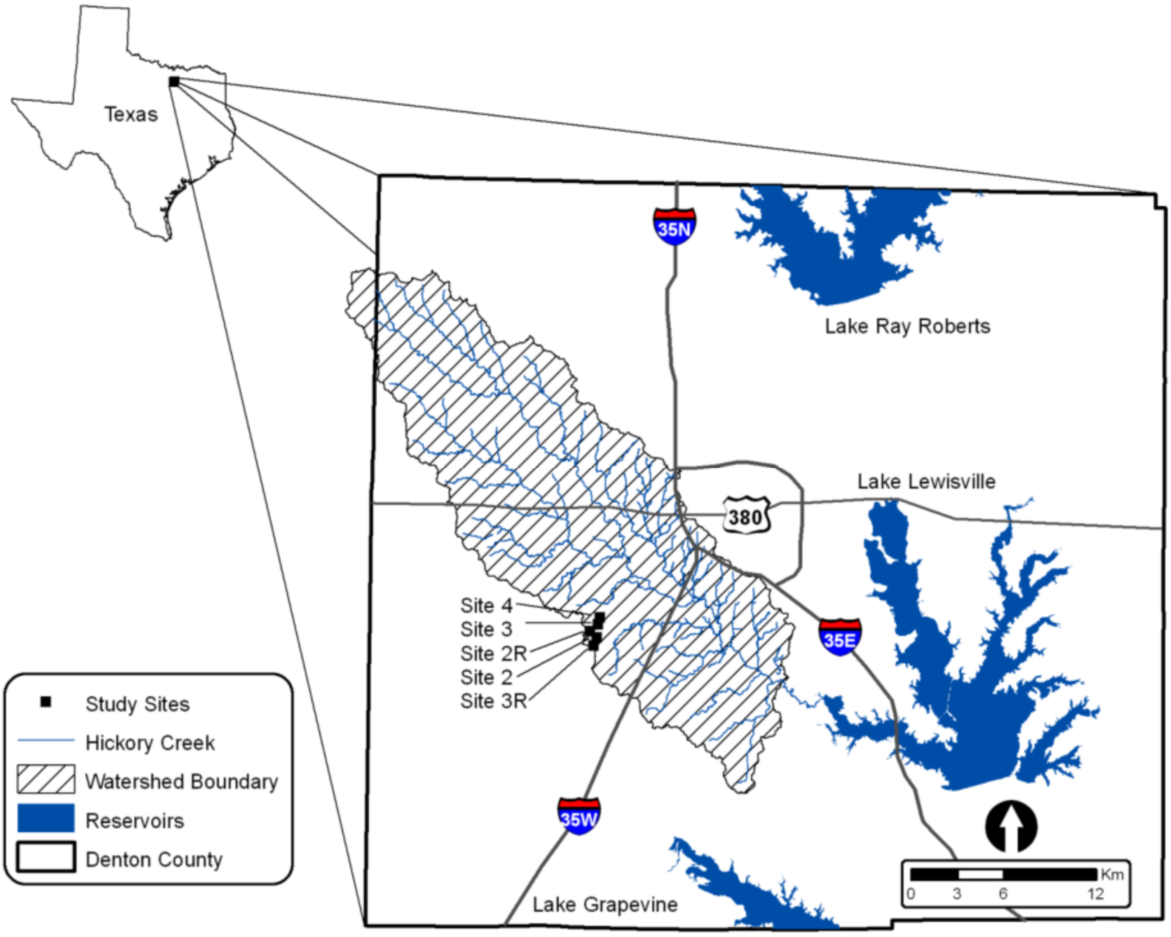

Figure B.1. Study Area - Denton County, Texas

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

Table B.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 B.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 B.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 B.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 B.5. Physical and Chemical EMC Summary

		Gas Well Sites			Reference Sites	
Parameter	Variable	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 B.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 B.7. Drinking Water Standards, Aquatic Life Criteria, and Local Conditions (Values in mg l^{-1})

Parameter	Standard	Aquatic Life Criteria	Runoff from Local Watersheds[a]	
			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

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

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 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 1.

<u>Site Monitoring</u>

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 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 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.

<u>Model Description</u>

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).

<u>Input Parameters</u>

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 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 4.

Soil parameters for the cut slopes were obtained from WEPP's Medlin soil series input file. Soil information for any soil in the U.S. 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 4. Default and calibrated WEPP soil parameters are listed in Table 5.

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 –∞ 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:

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

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:

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

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).

<u>Measurement Uncertainty</u>

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 6. 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 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). 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 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 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 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.

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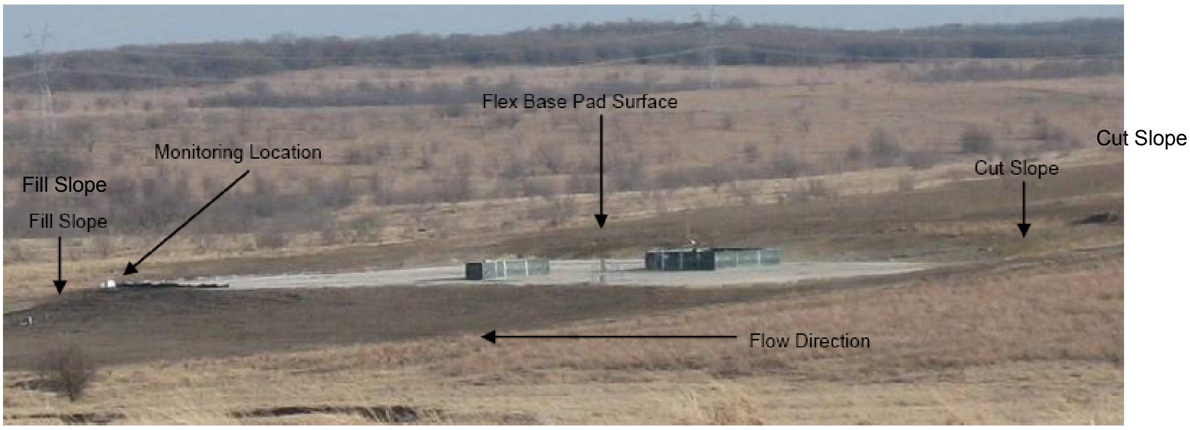

Figure C.1. Gas well pad surface (GW1) on modified hillslope

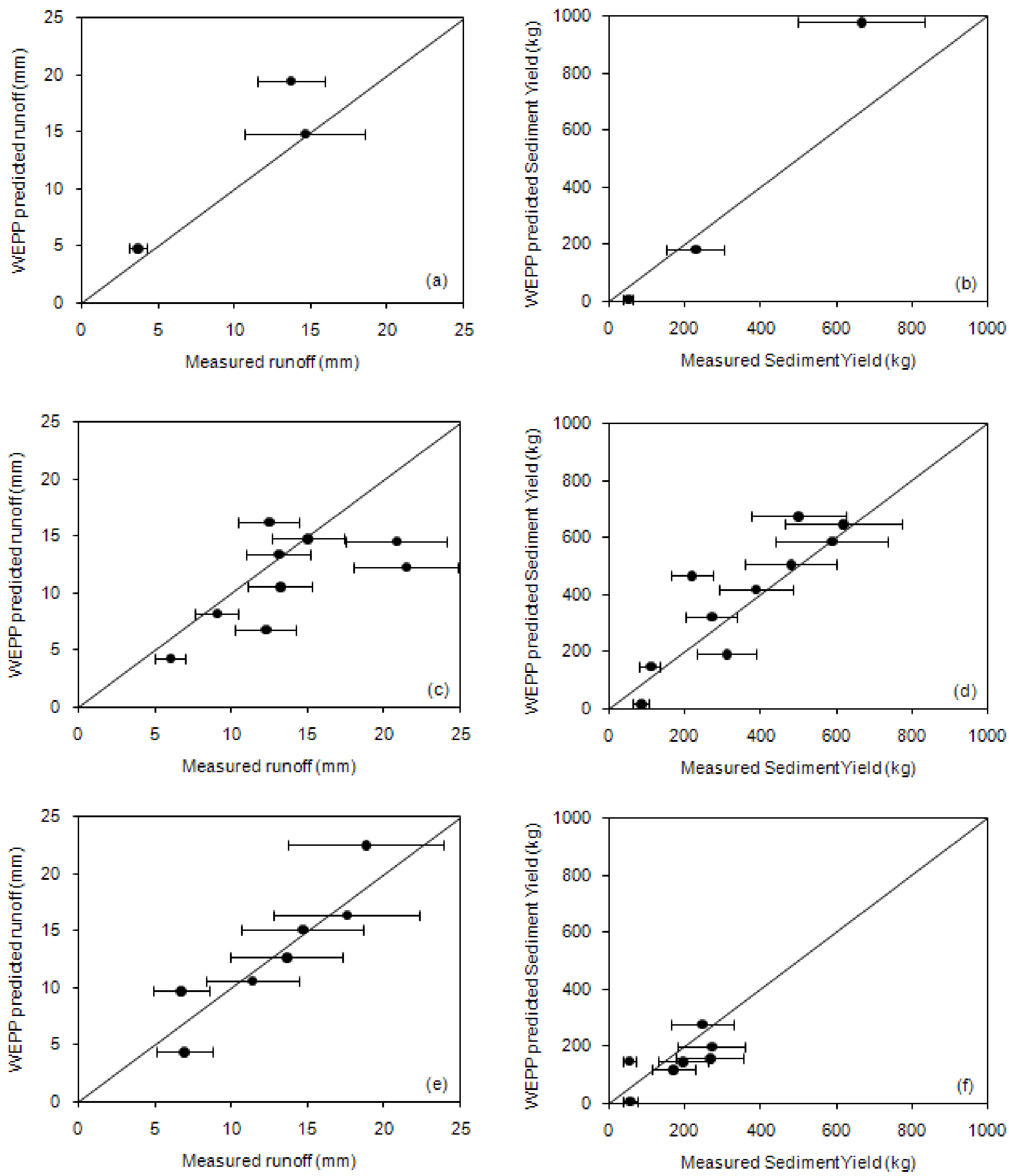

Figure C.2. Scatterplots of measured and predicted runoff (mm) and sediment yield (kg) modified with Modification 1 to include the uncertainty range (PER) for each measured value: (a) calibrated runoff (PER = ±16%, ±27%); (b) calibrated sediment yield (PER = ±25%, ±33%); (c) GW1 runoff (PER = ±16%); (d) GW1 sediment yield (PER = ±25%); (e) GW2 runoff (PER = ±27%); (f) GW2 sediment yield (PER = ±33%)

Table C.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	

Table C.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 C.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 C.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 C.5. 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 C.6. Measured and Predicted Runoff and Sediment Yield

Site	Sampling Date	Runoff (mm) Meas.	Runoff (mm) Pred.	Sediment Yield (kg) Meas.	Sediment Yield (kg) 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 C.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)

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

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 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 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 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 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 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 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 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 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 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 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. 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 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 (±2.1), 28.9 (±3.1), and 28.7 (±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 (±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) 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 7. Based on the relationship illustrated by Figure 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) 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.

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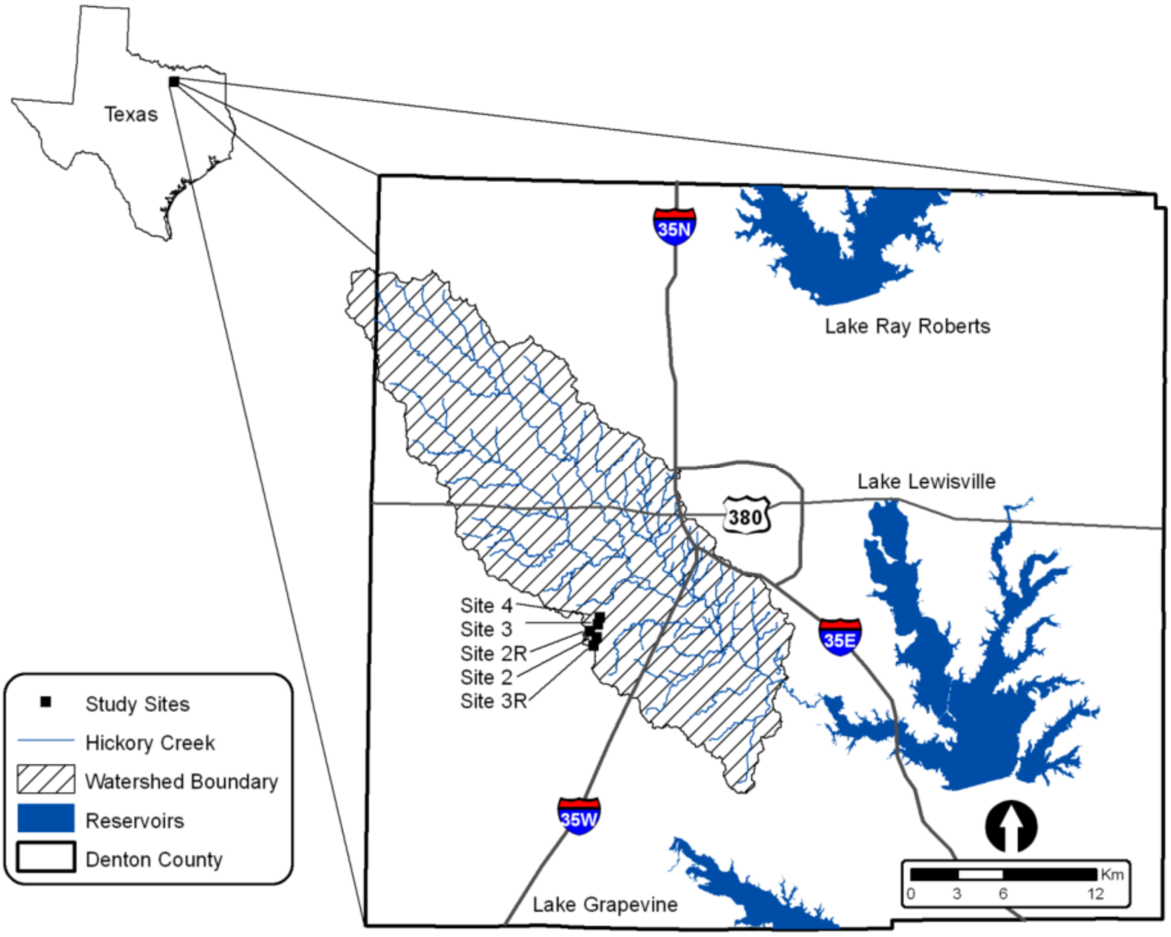


Figure D.1. Study Area - Denton County, Texas

Figure D.2. Cracks in the soil at the reference site

Figure D.3. Gas well pad site research plot

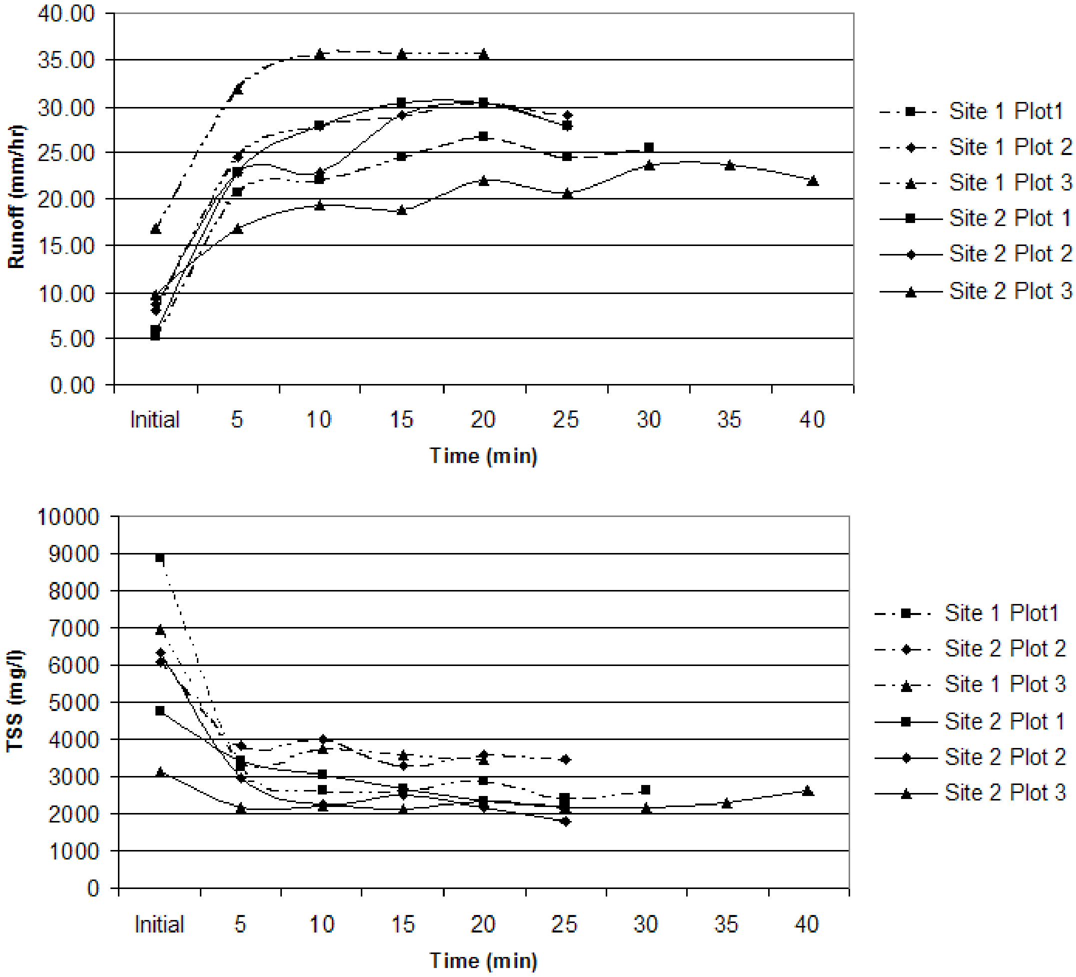
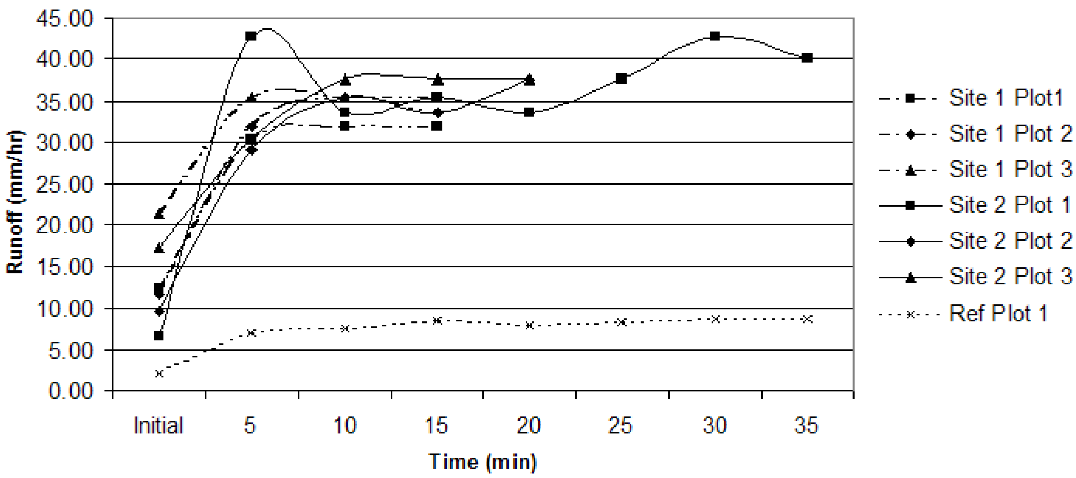


Figure D.4. Dry run runoff and TSS



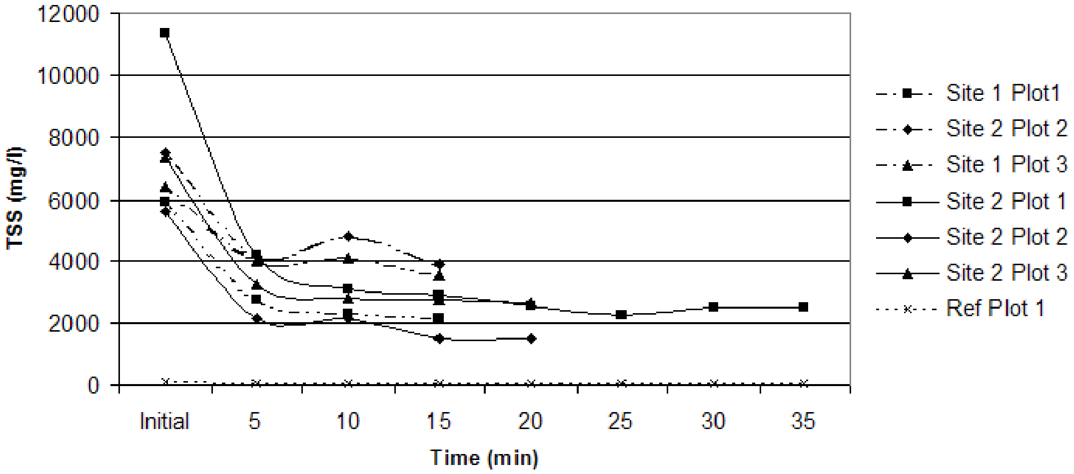


Figure D.5. Wet run runoff and TSS

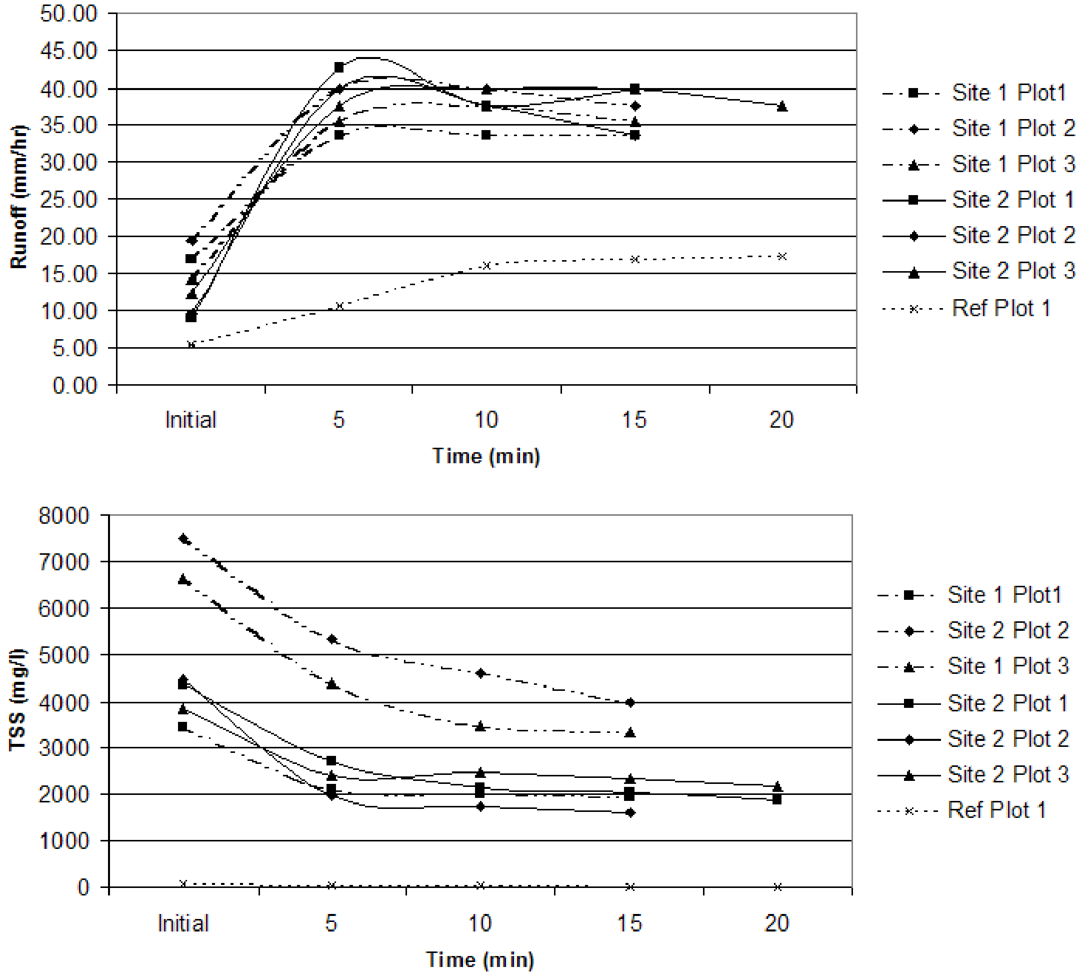
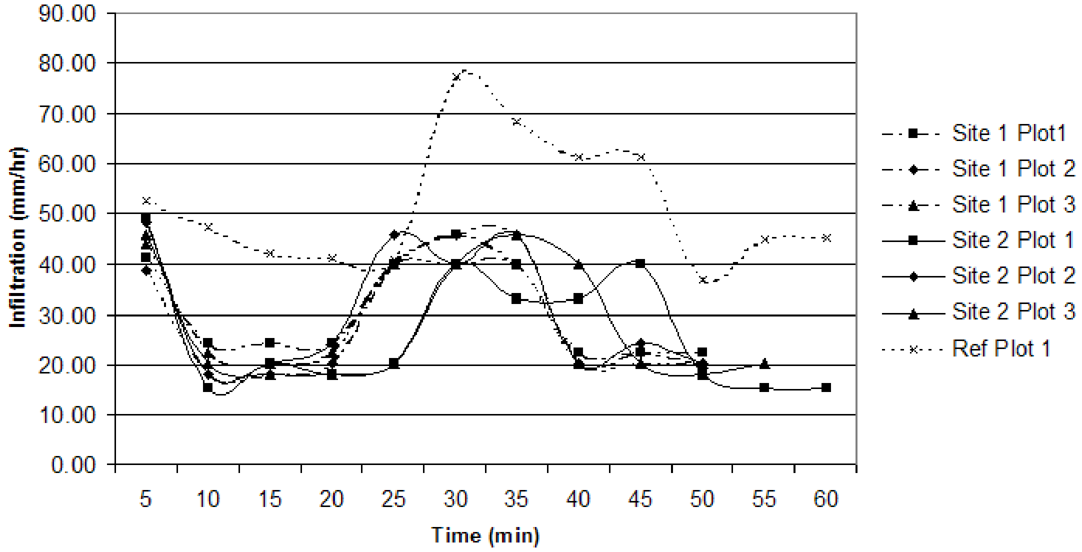


Figure D.6. Very wet run runoff and TSS



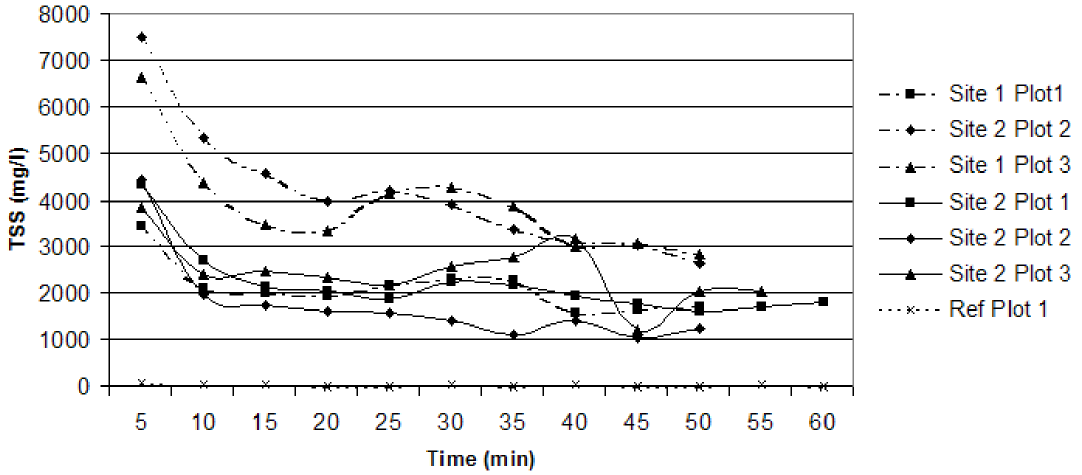


Figure D.7. Very wet run increased rainfall intensity infiltration rate and TSS

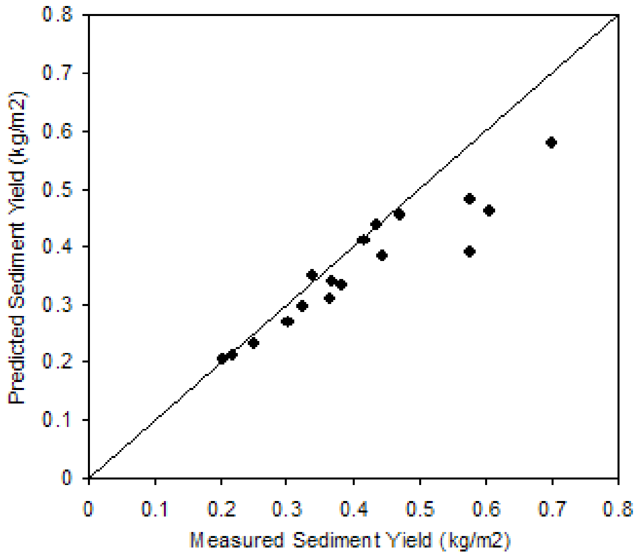


Figure D.8. Scatterplot of observed and predicted sediment yield

Table D.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 D.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 D.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 D.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 D.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

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

Methods

Study Area and Site Description

The study area lies above the Barnett Shale formation in Denton County, in North Central Texas (Figure 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 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 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 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 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 ± 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 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 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 3 for example of moderate slope condition) but decreased with increased slope (see Figure 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 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 2). Since erosion blankets are designed for steep slopes, it is not surprising that erosion blankets performed the best on steep slopes (*ER* ≥ 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 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).

<u>Sediment Control with Silt Fences.</u> 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 3 for example of moderate slope condition), and remained below 0.70 for all slope conditions when the erodibility condition was high (Table 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 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 2). Filter strips also had *ER*s greater than 0.70 for all RI years (Table 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 2) and tended to be highest for the low erodibility (loamy sandy soils) condition (see Figure 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 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.

<u>Comparison of BMP Efficiencies and Costs.</u> 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 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 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 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 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 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 6, 7, and 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 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 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 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.

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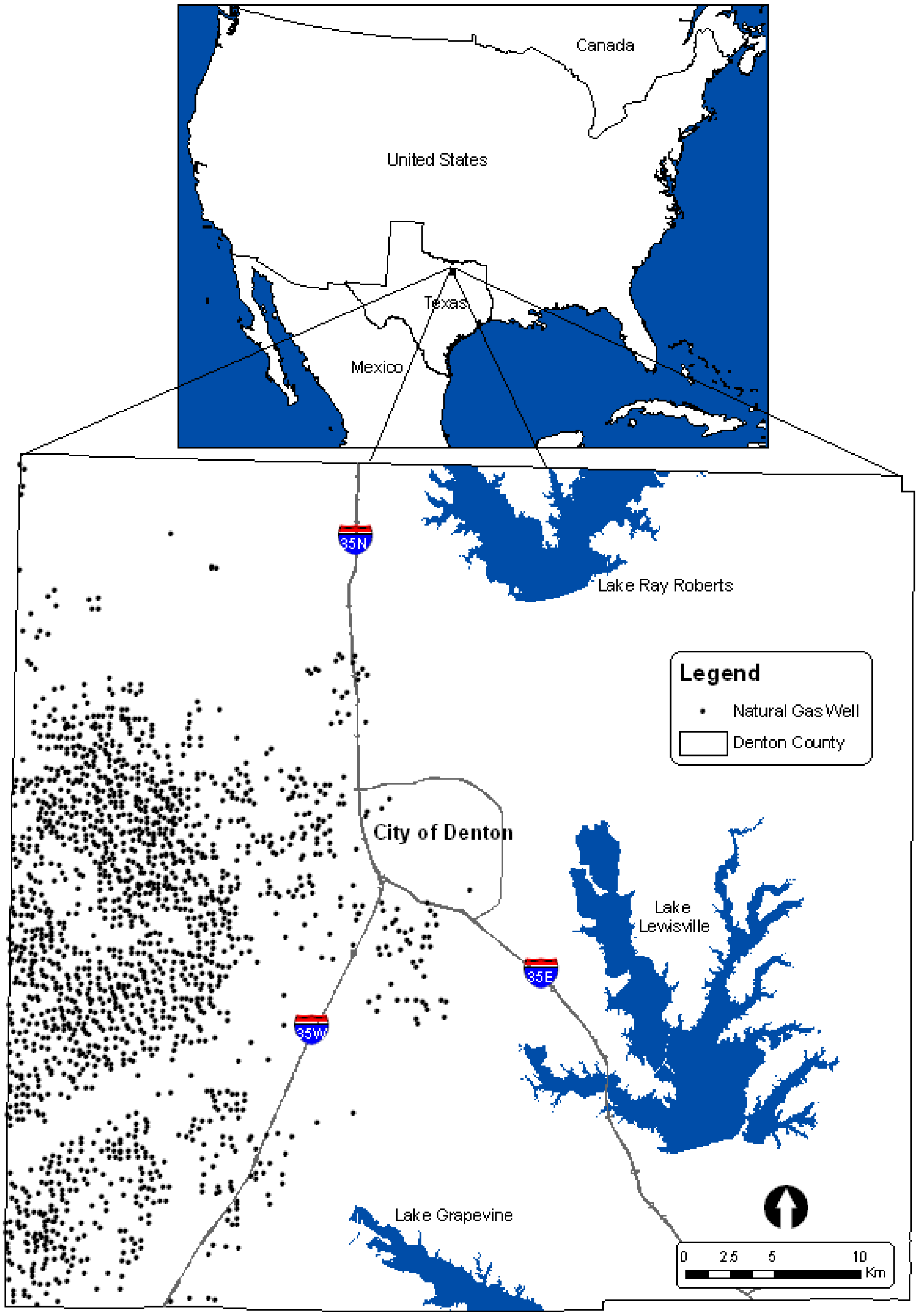

Figure E.1. Study area Denton County, Texas, USA

Figure E.2. Gas well site on modified hillslope

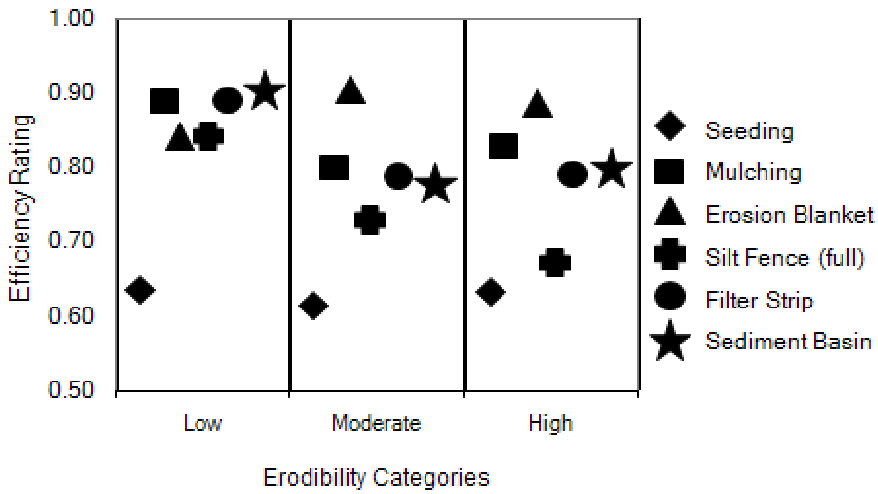

Figure E.3. Average annual BMP ERs for moderate slope condition

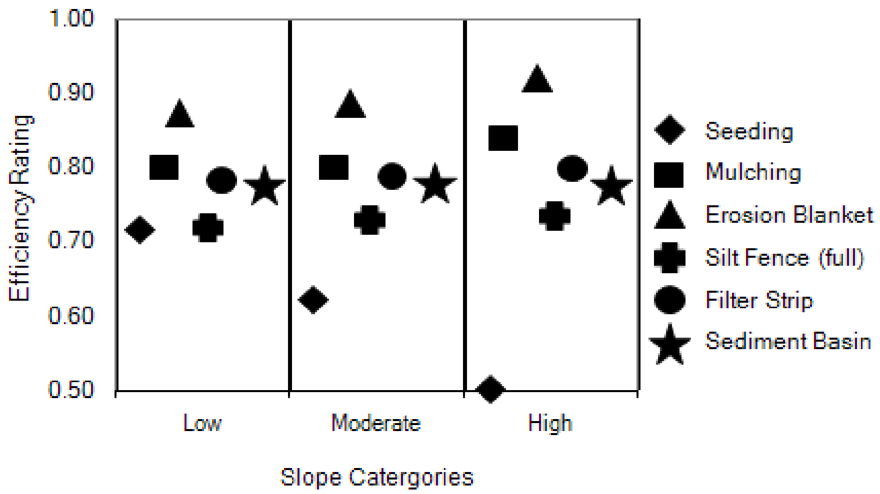

Figure E.4. Average annual BMP ERs for moderate erodibility condition

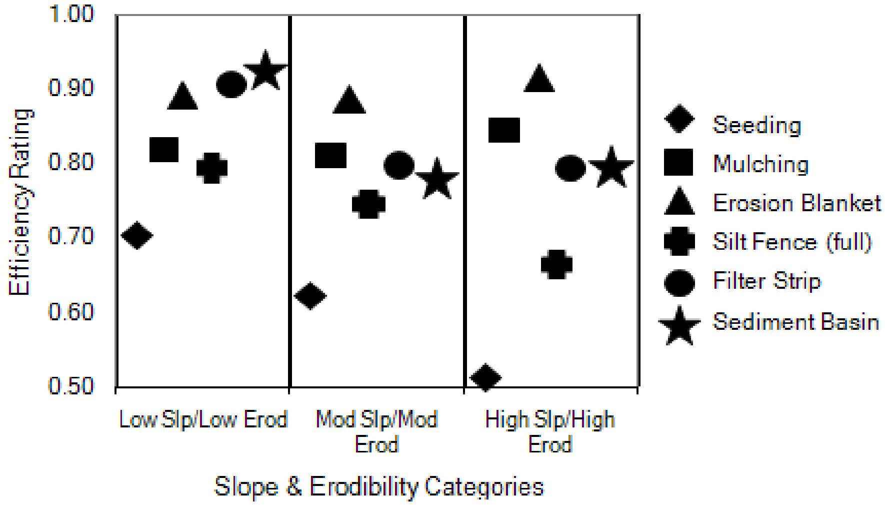

Figure E.5. Average annual BMP ERs for low, moderate, and high combined factors

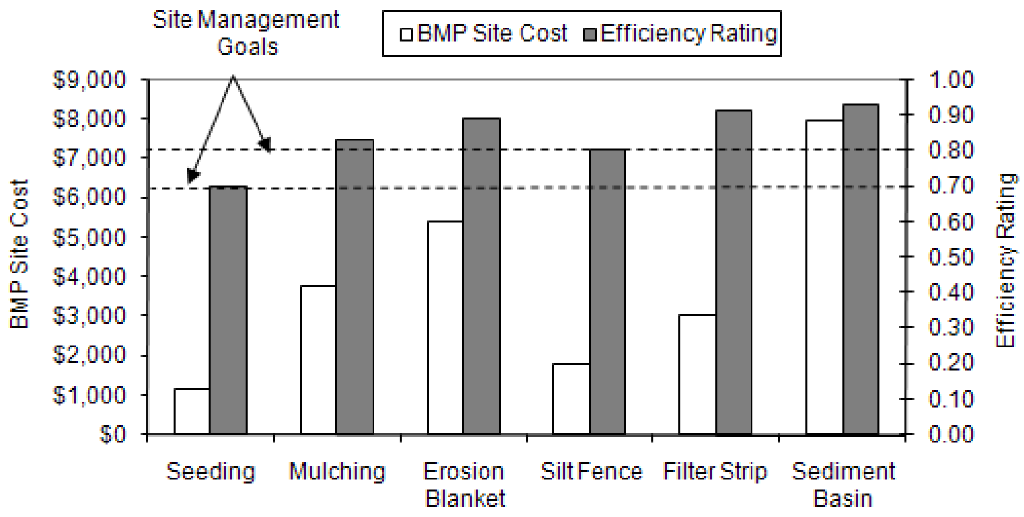

Figure E.6. BMP cost/ER comparison for low erodibility/low slope condition

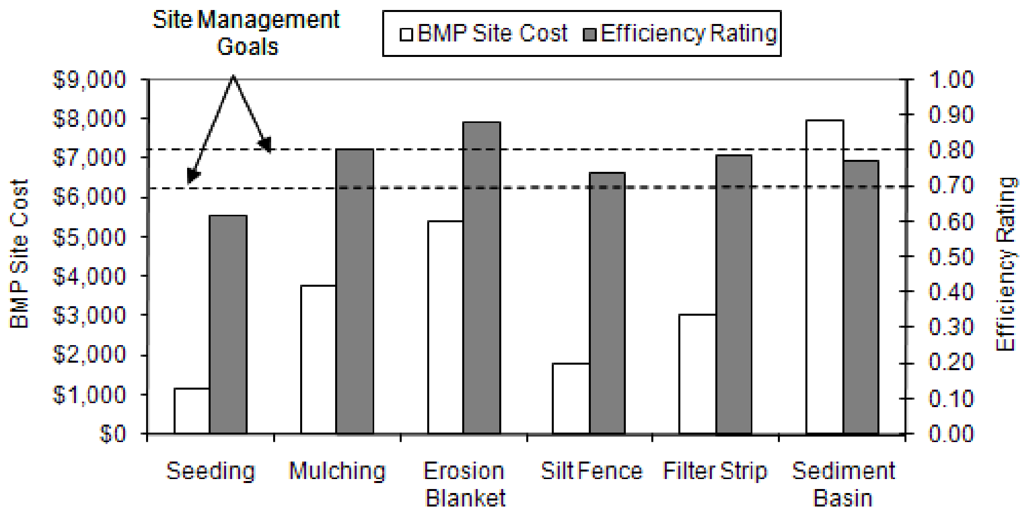

Figure E.7. BMP cost/ER comparison for moderate erodibility/moderate slope condition

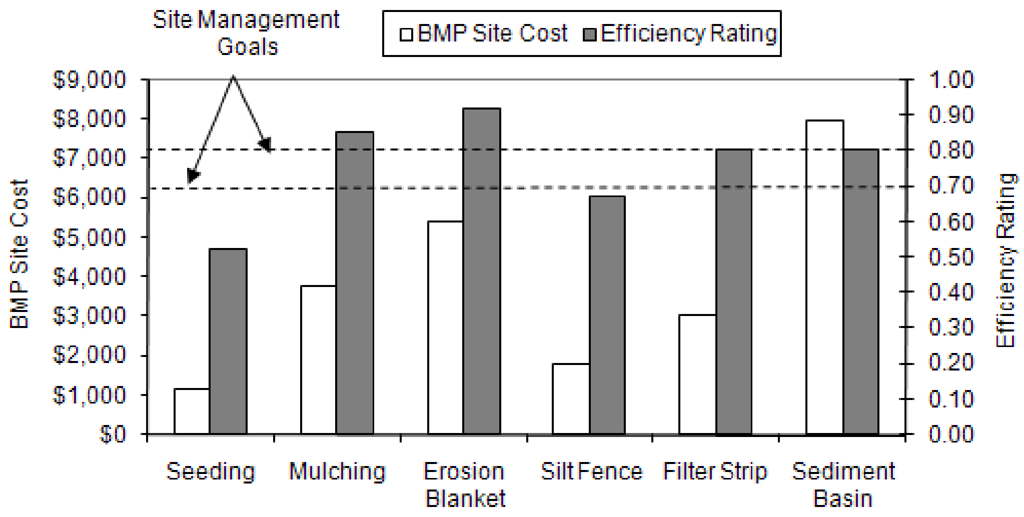

Figure E.8. BMP cost/ER comparison for high erodibility/high slope condition

Table E.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 E.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 E.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