Sediment Delivery from Reopened Forest Roads at Stream Crossings in the Virginia Piedmont Physiographic Region, USA

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ABSTRACT

Efforts to control surface runoff and erosion from forest roads at stream crossings are critical for the protection of aquatic ecosystems in forests. In this research, annual and event-based sediment delivery rates were estimated for reopened legacy roads at forest stream crossings in the Virginia Piedmont physiographic region, USA. Sediment delivery rates were compared among stream crossing approaches with diverse intensities of best management practice (BMP) implementations for surface cover and water control. Model predictions from the Water Erosion Prediction Project (WEPP) were compared to field observations of surface runoff and sediment delivery to evaluate model performance.

Annual sediment delivery rates from reopened (bare) legacy road approaches to stream crossings were 7.5 times higher than those of completely graveled approaches. Sediment delivery rates ranged from 34 to 287 Mg ha^{-1} year^{-1} for the bare approaches and from 10 to 16 Mg ha^{-1} year^{-1} for the graveled approaches. Event-based surface runoff and associated total suspended solids (TSS) concentrations were compared among a succession of gravel surfacing treatments that represented increasing intensities of BMP implementations on reopened approaches. The three treatments were No Gravel (10-19% cover), Low Gravel (34-60% cover), and High Gravel (50-99% cover). Median TSS concentration of surface runoff for the No Gravel treatment (2.84 g L^{-1}) was greater than Low Gravel (1.10 g L^{-1}) and High Gravel (0.82 g L^{-1}) by factors of 2.6 and 3.5,

respectively. WEPP predictions of event-based sediment yield show clear differences among the different road surface treatments, but prediction intervals were wide, reflecting substantial prediction uncertainty.

These findings show that reopened legacy roads and associated stream crossing approaches can deliver significant quantities of sediment if roads are not adequately closed or maintained and that corrective best management practices (BMPs), such as gravel and appropriate spacing of water control structures, can reduce sediment delivery to streams. Watershed management decisions that hinge upon WEPP predictions of sediment yield from forest roads must necessarily take into account a wide range of potential erosion rates for specific management scenarios.

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CONTRIBUTION OF AUTHORS

Dr. W. Michael Aust had the idea for me to write a literature review about soil erosion and sediment delivery associated with forest operations in the Piedmont physiographic region of the United States. Dr. Aust and Dr. Kevin J. McGuire helped me to refine research questions and develop experimental designs for the sediment trap and rainfall simulation studies. Dr. McGuire was instrumental in helping me to use different methods for evaluating uncertainty associated with erosion model predictions, as well as troubleshooting our computer programming in Matlab. Dr. W. Cully Hession and Dr. C. Andrew Dolloff also provided important insights about the direction of this dissertation research during annual committee meetings. In particular, Dr. Hession was helpful with his experiences regarding the rainfall simulator equipment. All committee members provided critical reviews of one or more of the manuscripts in this dissertation. Specific publication information is contained in Chapter 1.

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1.0 Introduction

Forest roads and associated stream crossings are major sources of sediment delivery to streams (Croke et al., 1999; Lane and Sheridan, 2002). Many studies have evaluated the effectiveness of forestry BMPs to reduce erosion, but much uncertainty exists regarding the effectiveness of specific BMPs to reduce sediment delivery to streams and for diverse site conditions. Few studies have quantified the efficacy of best management practices (BMPs) to reduce soil erosion and sediment delivery, as well as BMP implementation costs (Brown et al., 1993; Sawyers et al., 2012; Wear et al., 2013), but both are critical to justify BMP implementation.

By focusing field experiments of erosion and sediment delivery at stream crossings (i.e., where sediment delivery potential is high), the opportunity exists to better approximate reductions in sediment delivery as a result of BMP implementation. Furthermore, improvements in the effectiveness of BMPs to reduce sediment delivery at stream crossings can achieve greater impacts on protecting water quality and aquatic habitat in managed forests. Field experiments can provide valuable information about rates of erosion and sediment delivery from forest roads for a wide range of site characteristics, BMP implementations, traffic types and intensities, and climatic conditions. However, in light of the need for efficient examination of the linkages between BMP implementations and sediment reductions across broad spatial scales, additional approaches must be used alongside field monitoring to guide forest road management for water quality protection.

Therefore, soil erosion models are used to aid in the identification of problem road sections for erosion and sediment delivery to streams, assess potential impacts to water quality, and evaluate BMP effects on reducing sediment delivery. However, model performance must be tested to evaluate the utility of model predictions to inform decision makers about the best ways to manage stormwater runoff and sediment delivery from roads. Erosion model performance and uncertainty associated with model predictions have not been evaluated at the road-stream interface for a wide range of approach characteristics, BMP implementations, and rainfall conditions. Such approaches are critically important because the U.S. Environmental Protection Agency is currently requesting that state forestry organizations evaluate the effectiveness of existing BMPs to reduce sediment delivery from major sources (i.e. roads and stream crossings) and provide guidance for enhanced BMPs. Soil erosion models are needed to help achieve this objective. Thus, the goals of this research are to quantify sediment delivery rates from forest roads at stream crossings for a wide range of approach characteristics, BMP implementations, and rainfall conditions, and to evaluate the utility of erosion model predictions to guide forest road management.

1.1 Objectives and Organization

This dissertation is organized into six chapters. The first chapter provides an outline for this dissertation and identifies some major problem areas regarding the control of stormwater and sediment delivery from forest roads at stream crossings. Chapters two through five are designed to be stand-alone manuscripts that have been, or will be, submitted for peer review publication. The final chapter will summarize study findings from the preceding chapters, as well as their implications for the protection of water quality in forested environments.

The second chapter is a literature review of studies that investigated erosion and sediment delivery associated with forest operations in the Piedmont physiographic region of the U.S. The objectives of this chapter were to consolidate and organize study findings by forest operation, including recent unpublished graduate theses, evaluate BMP performance for specific operations, and identify future research needs to reduce sediment delivery to streams. This manuscript was written by Kristopher Brown, with contributions from Dr. W. Michael Aust, and Dr. Kevin J. McGuire. This manuscript was published as a United States Department of Agriculture Forest Service, Northern Research Station General Technical Report, NRS-P-117 (Brown et al., 2013). The right to include the published manuscript in this dissertation is retained by the authors (USDA Forest Service, 2013).

The third chapter presents the results of a field experiment that estimated annual sediment delivery rates from reopened (bare) and completely graveled forest roads at stream crossings in the Virginia Piedmont physiographic region, USA. The main objective of this study was to quantify annual rates of sediment delivery from stream-crossing approaches with minimum surface cover (newly-reopened, bare legacy roads) and maximum surface cover (existing gravel roads) using sediment traps and repeated measures. Sediment delivery rates were compared by surface type (bare, graveled) to evaluate the sediment reduction efficacy of graveling the entire running surface component of forest roads at stream crossings. An additional objective was to evaluate the extent to which observed sediment delivery rates were governed by surface cover, length, and slope of the stream-crossing approaches, as well as rainfall amount, rainfall intensity, and the number of days since road reopening or construction (i.e., time since disturbance). Road approach length, slope, and surface coverage represent management factors that can be controlled during pre-harvest planning, construction, and implementation phases. This study investigated their effects on soil erosion and sediment delivery and evaluated the effectiveness of related BMPs in minimizing sediment delivery to streams. A secondary objective was to evaluate model predictions of sediment delivery from forest roads at stream crossings using the Universal Soil Loss Equation modified for forestland (USLE-forest) and the Water Erosion Prediction Project (WEPP). This manuscript was written by Kristopher Brown, with contributions from Dr. W. Michael Aust, and Dr. Kevin J. McGuire. This manuscript was published in *Forest Ecology and Management* (Brown et al., 2013). The right to include the published manuscript in this dissertation is retained by the authors (Elsevier, 2013).

The fourth chapter focuses on three research questions in the Piedmont physiographic region of southwestern Virginia, USA: (1) How does gravel cover affect road surface runoff?; (2) What are the total suspended solids (TSS) concentrations in road surface runoff during rainfall events for reopened stream-crossing approaches?; (3) What sediment reduction can be achieved using different graveling intensities on stream-crossing approaches and what is the cost of reducing sediment by graveling? Rainfall simulation experiments were conducted on the entire running surface area associated with six reopened stream-crossing approaches. Event-based road surface runoff and associated TSS concentrations were compared among a succession of treatments that represented increasing intensities of BMP implementation. The three treatments were (1) No Gravel: existing conditions following reopening by bulldozer blading (10-19% cover); (2) Low Gravel: gravel application beginning at the stream and continuing uphill for 9.8 m (34-60% cover); and (3) High Gravel: gravel application that doubled the length of the Low Gravel treatment (50-99% cover). The cost-effectiveness of the gravel treatments was evaluated by relating the costs associated with BMP implementation to reductions in sediment. This manuscript was written by Kristopher Brown, with contributions from Dr. Kevin

J. McGuire, Dr. W. Michael Aust, Dr. W. Cully Hession, and Dr. C. Andrew Dolloff. This manuscript was accepted for publication in *Hydrological Processes* in April 2014 (Brown et al., 2014). The right to include the published manuscript in this dissertation is retained by the authors (John Wiley & Sons, 2014).

In the fifth chapter, the Water Erosion Prediction Project (WEPP) was used to predict event-based sediment yield and runoff for rainfall simulation experiments on six reopened stream-crossing approaches that each underwent different intensities of BMP implementation (i.e., different proportions of gravel on the road surface above the stream-crossing). WEPP was calibrated for each of these stream-crossing approaches for three different BMP intensities using a Markov Chain Monte Carlo (MCMC) approach to explore parameter uncertainty and identifiability/sensitivity, prediction uncertainty, and overall prediction performance based on a simple least squares objective function. The objectives of the study were to determine overall prediction performance of WEPP and its ability to distinguish between different BMP intensities.

This study focuses on three research questions in the Piedmont physiographic region of southwestern Virginia, USA: (1) How well does WEPP predict event-based runoff and erosion from forest roads at stream crossings?; (2) Can WEPP simulate changes in soil erodibility as a function of successive rainfall events, as well as the treatment effect of increasing gravel surface cover on the stream crossing approaches?; (3) Can distinct ranges of model input parameter values be identified in association with the different road surface treatments? This manuscript was written by Kristopher Brown, with contributions from Dr. Kevin J. McGuire, Dr. W. Cully Hession, and Dr. W. Michael Aust. This manuscript will be submitted to a peer-reviewed journal for publication consideration.

The sixth chapter is the dissertation conclusion. This chapter summarizes what was learned about sediment delivery rates associated with reopening legacy forest roads at stream crossings, the effectiveness of various intensities of gravel surfacing BMPs to reduce sediment delivery, and the utility of the WEPP model to estimate BMP effectiveness from experimental forest roads. The implications of our research findings are discussed relative to forest land managers, academicians, and State and Federal forestry agencies, and suggestions for future research are provided. This chapter was written by Kristopher Brown.

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2.0 SEDIMENT ASSOCIATED WITH FOREST OPERATIONS IN THE PIEDMONT REGION

Kristopher R. Brown[a], W. Michael Aust[b], and Kevin J. McGuire[c]

2.1 Abstract

Reduced-impact forestry utilizes Best Management Practices (BMPs) during operations to reduce soil erosion and sediment delivery to streams and to maintain or improve site productivity. However, the efficacy of specific types of BMP implementation is not widely documented. This review will synthesize recent research that investigated contemporary BMP implementation and effectiveness in water quality protection associated with the following forest management operations: forest roads and skid trails, streamside management zones, harvesting, site preparation, and stream crossings. The review concentrates on studies conducted in the Piedmont region and serves to facilitate integration and comparison with forestry BMP effectiveness research from the western U.S. The Piedmont region of the eastern U.S. is home to ubiquitous intensive forest management, where sediment delivery to upland streams from forest operations has water quality implications for downstream water bodies. In addition, the decision by the U.S. Court of Appeals of the Ninth Circuit Court that forest roads are point sources of pollution (Boston, 2012), which would later be reversed by the Supreme Court in March of 2013, highlights the importance of forest roads to water quality, forest operations, and national water

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policy decisions. General results indicate that the most serious water quality issues are associated with bare soil conditions that are hydrologically connected to streams by roads, skid trails, and/or concentrated flows. Future research should determine sediment delivery ratios for forest road and skid trail approaches to stream crossings in order to develop and implement management strategies for minimizing sediment that has the highest probability of reaching the stream.

2.2 Introduction

Sediment is one of the most frequently cited water quality concerns associated with forestry operations (Riekerk et al. 1989; Grace 2002; Stuart and Edwards 2006) and is consistently ranked among the top 10 causes of river and stream impairment in the United States (USEPA, 2003). Streams flowing through forested land generally have lower sediment concentrations relative to agricultural or urban areas, owing largely to the presence of the forest floor. The forest floor is composed of leaf litter and woody debris, which act to prevent soil erosion in a variety of ways. The forest floor covers bare soil and prevents sediment detachment from rainfall droplets. The forest litter layer, humus, and mineral soil have high infiltration capacities that are rarely exceeded, even by intense rain events. When surface runoff does occur, litter decreases the velocity of overland flow and acts to trap sediment. Typical forest operations include access road construction and maintenance, installation of water control structures and stream crossings, harvesting and thinning, skidding, building log decks, fireline construction, burning, and site preparation. Each of these operations increases the percentage of bare soil within a watershed, thus increasing soil erosion and the potential for sediment delivery to streams. Forest cover removal generally results in short-lived streamflow increases as a result of decreased evapotranspiration (McGuire and Likens 2011). Increases in stormflow volumes and peakflows can accelerate within-channel erosion.

Forest operations commonly occur within the drainage areas of zero-, first-, and second-order streams. These headwater streams may be ephemeral, intermittent, or perennial. Headwater streams compose over two-thirds of the cumulative drainage length of river basins and serve to link riparian and upland habitats to downstream ecosystems by providing streamflow, physical habitat, allochthonous organic material, and aquatic life (Benda et al., 2005; Freeman et al., 2007). Therefore, headwaters can govern downstream hydrologic conditions and water quality on a regional scale. For example, Dodds and Oakes (2008) stress the importance of riparian buffers in headwater reaches for the protection of downstream aquatic ecosystems.

Reduced-impact forestry utilizes Best Management Practices (BMPs), which have proven to be generally effective in reducing sediment inputs to streams (Aust and Blinn 2004; Ward and Jackson 2004; Wang and Goff 2008). However, BMP implementation does not eliminate sediment delivery to streams altogether. For example, in a review of three paired watershed studies in the eastern United States, Edwards and Williard (2010) calculated that BMP implementation reduced sediment by 53 to 94%. Often, BMP failures that contribute sediment to streams are non-uniformly distributed and occupy relatively small proportions of the total forest operational area. Rivenbark and Jackson (2004) estimated that approximately 0.33 to 0.4 percent of southeastern Piedmont industrial forest land is contributing to SMZ failures at any given time. Much work has yet to be done to understand both the spatial distribution of BMP failures within a watershed or operational area, as well as the causes of BMP failures. For example, slope steepness, surface runoff contributing area, topographical feature type (e.g. gullies and swales), bare soil percentage, and their interactions have been used to aid in the characterization of sediment problem areas (Rivenbark and Jackson 2004).

Evaluation of reduced-impact forestry practices to minimize soil erosion and sediment delivery to streams is particularly relevant for the Piedmont physiographic province of the eastern United States. For more than a century prior to the 1930s, poor agricultural practices associated with row crop agriculture of corn, cotton, and tobacco caused extensive soil loss, gully formation, and aggradation of stream channels across the Piedmont, particularly in the southern states. Trimble (1985) describes an era of "land rotation," whereby exhausted farmland was abandoned and left to regrow, while forested land was cleared for new farms. This practice resulted in a highly eroded landscape, with sediment-laden stream channels and valley bottoms. Soil loss across the Piedmont has been estimated to be 60 cm or more (Trimble 1985). Although this region is now mostly forested, Piedmont streams continue to export these legacy sediments, which confounds the quantification of contemporary land use effects on stream sedimentation (Jackson et al. 2005).

In addition, suspended sediment production from Piedmont forestry operations is high in comparison to mountainous and coastal plain sites of the southeastern U.S. because of the interaction between site preparation intensity and topographic relief (Riekerk et al. 1988) and clay rich soils. Industrial forest operations are ubiquitous throughout the Piedmont. Future demand in the South for forest products is anticipated to increase (Anderson and Lockaby 2011), which underscores the importance of understanding how well reduced-impact forestry practices perform in protecting stream water quality under various scenarios that may include increased stand entry, shorter rotations, and higher overall production over fewer forested areas.

Unlike well-known research sites in the northeast U.S. such as Coweeta, Hubbard Brook, Fernow, and Leading Ridge (Ice and Stednick 2004), the Piedmont physiographic province lacks a cohesive research unit. However, many recent studies have been conducted in this region regarding the effects of contemporary reduced-impact industrial forest operations on soil erosion and sediment delivery to streams. The objectives of this review are to consolidate and organize study findings by forest operation, including recent unpublished graduate theses, evaluate BMP performance for specific operations, and identify future research needs to reduce sediment delivery to streams.

2.3 Forest Operations and Sediment in the Piedmont

2.3.1 Harvesting and Site Preparation

Generally, harvesting itself does not substantially increase soil erosion. However, skid trails, log decks, and roads commonly cover 2 to 10 percent of logged sites (Kochenderfer 1977) and represent the most significant threat to water quality from forest operations due to an increase in erosion potential resulting from bare soil exposure, compaction, and increased surface runoff. Nutter and Douglass (1978) defined "soil-loss tolerance" for traditional agriculture (e.g. row crop agriculture) as the maximum average annual rate of soil erosion that permits a high level of productivity to be sustained economically and indefinitely. Soil-loss tolerance ranges from 4.4 to 11.2 Mg/ha (2 to 5 tons/acre) for intensively managed (fertilized and site prepared), "good" agricultural soils in the Piedmont. The authors contended that most harvest methods would not exceed the soil-loss tolerance for agricultural soils and recommended that following harvest, there should be no site preparation that would expose additional mineral soil on slopes greater than 15 percent. This indicates an awareness of the potential for high rates of soil loss owing to the interaction between slope steepness and intensive site preparation practices that expose bare soil.

Pye and Vitousek (1985) estimated soil erosion rates resulting from clearcut harvest of 22-yr-old loblolly pine (*Pinus taeda*), followed by site preparation in the Piedmont of North Carolina. The study location was characterized by gentle slopes (0 to 10 percent) and clayey, kaolinitic, thermic Typic Hapludult soils. Three blocks (5 ha each) were clearcut, with half of each block either stem-only or whole-tree harvested. One half of each harvest treatment was drum chopped. The other half of each harvest treatment was sheared, windrowed, and the inter-windrow areas were disked. Most of the windrows were burned, while burning was unsuccessful for the drum chopped areas. Finally, the four resultant treatment combinations were halved, with herbicide applied to one half and no herbicide application on the other. The split-split plot experimental design was replicated in each of the three blocks, which resulted in 24 plots. Soil erosion was measured with sediment traps for one year, beginning nine months after site preparation. While the drum chopped plots produced minimal erosion, the windrowed sites produced a mean of 6.8 Mg/ha. This study shows that substantial soil erosion may occur even on gentle slopes when site preparation practices such as windrowing are implemented that remove or bury the forest floor. In addition, it is clear soil erosion may be effectively controlled by forest practices that minimize areas of bare soil.

In general, mechanized harvest operations compact soils, thus increasing bulk density and decreasing both aeration porosity and saturated hydraulic conductivity (Campbell et al. 1973). Gent et al. (1984) investigated changes in soil physical properties to a depth of 0.3 m for clayey, kaolinitic, thermic, Typic Hapludult soils in the Piedmont physiographic region of North Carolina in response to whole-tree harvesting (low traffic area), skidding (high traffic area), and site preparation methods that included shearing/windrowing/double disking or chopping/burning. Soils were slightly above field capacity during harvest and site preparation. Soil physical properties of skid trail plots were impacted to a greater depth (0.22 m) in comparison with whole-tree harvest plots (0.17 m). Disking restored soil physical properties to pre-harvest levels in the upper 0.07 to 0.12 m of soil. This study is further evidence that the greatest impacts to soils during a typical harvest operation are associated with highly trafficked, bare soil areas such as roads and skid trails. In addition, on sites with steeper slopes, decreased saturated hydraulic conductivity can increase overland flow and therefore, soil erosion potential.

Grace and Carter (2001) quantified the effect of harvesting and site preparation on sediment and runoff yield from a 20-yr-old loblolly pine plantation with sandy loam soils and slopes ranging from 3 to 15 percent in the southern Piedmont in Alabama. Following a 25-ha clearcut, site preparation treatments included: 1) shearing, ripping, bedding, and machine planting on contour; and 2) machine planting on contour. Treatments were compared with an unharvested control site. During the 20-month study period, soil erosion rates were 0.08, 0.16, and 1.02 Mg/ha for the unharvested control, Treatment 1, and Treatment 2, respectively. The more intensively site prepared plot (Treatment 1) was characterized by greater surface cover and roughness than Treatment 2, which provided greater protection against soil loss resulting from several high-intensity rain events. This indicates that rainfall timing and intensity may greatly influence soil erosion rates associated with forestry practices, as well as the apparent effectiveness of BMP implementation. In addition, the results of this study emphasize the importance of forestry operations (e.g. bedding) that maintain or create adequate surface roughness to allow for infiltration and decreased velocity of surface runoff. However, more studies are needed that quantify not only erosion rates from harvest operations, but also sediment delivery to adjacent water bodies, in order to evaluate and improve low-impact forestry practices.

For example, Hewlett (1979) estimated that for a typical forest operational unit, five percent of the detached soil reaches the stream channel.

2.3.1.1 Biomass Harvesting

In response to current woody biomass demand, chipping of logging residues, such as limbs, tops, and other non-merchantable material is being incorporated into some conventional timber harvesting operations to produce biomass fuel chips. Despite many benefits of biomass as an alternative energy source, there is some concern that the use of non-merchantable material for energy production at the expense of erosion control may increase soil loss and stream sediment concentrations. Barrett et al. (2009) used the Universal Soil Loss Equation modified for forest land (USLE-Forest) (Dissmeyer and Foster 1984) to estimate erosion rates on a biomass harvesting case study site in the Piedmont of Virginia. Estimated annual erosion rates for the biomass harvest ranged from 7.2 to 19.3 Mg/ha (3.2 to 8.6 tons/acre) as compared with erosion rates of less than 2.2 to 11.2 Mg/ha (1 to 5 tons/acre) for a similar conventional Piedmont harvest. Some states have already begun making additional recommendations for BMP implementation for biomass harvests. The authors concluded that more research is needed regarding the effects of biomass versus conventional harvesting on soil erosion before additional BMPs for biomass harvesting are recommended.

Soil erosion increases associated with forest roads and trails have been widely identified as the dominant nonpoint source of sediment pollution attributable to forest silvicultural activities (Grace 2002; Grace 2005; Croke and Hairsine 2006; Jordan 2006). Recently, the U.S. Court of Appeals of the Ninth Circuit ruled that logging roads should be considered point sources of pollution, therefore deciding that forest roads cannot be considered exempt from National Pollutant Discharge Elimination System (NPDES) permit requirements of the Clean

Water Act under the Silvicultural Rule (Boston and Thompson 2009; NEDC v. Brown 2011). The Ninth Circuit ruled that forest roads are point sources when runoff is confined and re-routed through well-defined conduits, such as ditches and culverts, which ultimately flow and transport sediment into streams and rivers. Although the ruling currently applies to roads within the jurisdiction of the Ninth Circuit in Oregon, both public and privately owned forest roads throughout the nation may require NPDES permits. This ruling emphasizes the importance of forest roads to water quality, forest operations, and national water policy decisions.

2.3.2 Roads, Skid Trails, Stream Crossings, and Streamside Management Zones

2.3.2.1 Roads

Forest roads are an integral component of forest harvesting operations and timber harvests are conducted on approximately 4,000 km^2 (1 million acres) of Virginia forest every 4-5 years (Barrett et al. 2012, VDOF 2012). The potential for water quality degradation due to forest roads is widely recognized (Luce 2002). The degree of water quality impacts resulting from forest road erosion depends on the delivery ratio of soil erosion to streams. Sediment is primarily delivered to streams through surface overland flow. Hydrologic connectivity between the road and stream networks depends on factors such as gully formation (Wemple et al. 1996; Croke and Mockler 2001) and mean annual precipitation (NRC 2008), but is inversely proportional to water control road features, such as waterbars, turnouts, and relief culverts (NRC 2008). Lakel et al. (2010) and Ward and Jackson (2004) found sediment delivery ratios from forest operations (including roads) to be approximately 10-25%, but forest roads alone can have higher delivery ratios. Dymond (2010) examined the influence of forest roads on water yield and concluded that road density effectively increased watershed stream density and stream flashiness. This implies that roads disproportionately increase water yields and sediment. In a catchment modeling study of road effects on hydrology in two heavily logged, small catchments on the western slopes of the Cascades, Storck et al. (1998) used the Distributed Hydrology-Soil-Vegetation Model (DHSVM) and found that forest roads increased peak flows for the largest storm events by approximately 17%. However, Surfleet et al. (2010) found that roughly 25-50% of DHSVM-simulated storm volumes and peak flows for road ditches were outside the uncertainty bounds of a generalized likelihood estimation procedure. This indicates substantial variability in modeled road runoff and emphasizes the need for studies that evaluate uncertainty in both model input parameters and predictions to evaluate model performance in accurately representing field hydrologic and soil erosion processes.

Road contribution of sediment to total export at the watershed and basin scale is highly variable. Turton et al. (2009) studied sediment yield to streams for unpaved roads in Oklahoma and estimated that roads may contribute up to 35% of the total sediment load for a large watershed (715 km^2). However, Sheridan and Noske found that near-stream unsealed forest road surfaces contributed only 4.4% of the total sediment load for a 135 km^2 watershed in southeastern Australia. Gravel application to bare road surfaces substantially decreases soil erosion (Kochenderfer and Helvey 1987).

2.3.2.2 Skid Trails

Wade et al. (2011) used a Randomized Complete Block Design to evaluate several skid trail closure techniques and ground cover BMPs for their performance in bare soil stabilization and erosion control. The study location was in the Virginia Piedmont physiographic region, with 10 to 15% slopes and sandy clay loam, fine, kaolinitic, mesic, Typic Kanhapludults. Treatments included: 1) Water bars (control); 2) Water bars plus seeding; 3) Water bars, seeding, and straw mulch; 4) Water bars plus hardwood slash; and 5) Water bars plus pine slash. Sediment was captured at the base of the plots by geotextile sediment filtration bags and weighed following rain events and at monthly intervals to obtain sediment weights. Three soil erosion models were used to compare measured soil erosion with modeled soil erosion, including USLE, the Water Erosion Prediction Project for Forest Roads (WEPP), and the Revised Universal Soil Loss Equation v.2 (RUSLE2). Mean annual erosion rates (Mg/ha) for the treatments were 137.7, 31.5, 8.9, 5.9, and 3.0 Mg/ha for the control, seed, hardwood slash, pine slash, and mulching treatments, respectively. In general, USLE, WEPP, and RUSLE2 correctly predicted the order in which treatments afforded the best erosion control, which demonstrates their utility in BMP evaluation. Results indicate that for areas of high erosion potential, water bars alone may be a poor choice for water quality protection due to their lack of soil stabilization. The best choices appear to be application of logging slash (see also Sawyers et al. 2011) or mulching. Slash may be the most advantageous choice because it is readily available on harvest sites and has a slower decomposition rate than straw mulch.

2.3.2.3 Stream Crossings

Sediment delivery is of particular importance at forest road stream crossings (Lane and Sheridan, 2002), which represent the most direct pathway for overland flow and sediment to stream channels. Therefore, sediment delivery ratios for forest road approaches to stream crossings should be determined in order to implement management strategies for minimizing sediment that has the highest probability of reaching the stream. The 2010 Virginia Department of Forestry (VDOF) BMP audit indicated that improper BMP implementation at stream crossings was the most important problem identified from forest operations in Virginia (Lakel, pers. comm.).

Forest road approaches to streams, as well as stream crossings, have the potential to deliver the greatest quantity of sediment to streams during forest operations (Swift 1986; Taylor et al. 1999; Carroll et al. 2008). Installation of crossing structures requires heavy equipment trafficking over sensitive stream banks, riparian zones, and potentially in the stream channel itself. In addition to sedimentation from equipment, sediment can run directly to streams from forest road approaches. Fords introduce sediment to streams as vehicles drive over the stream bed. Culvert installation, which involves excavation and fill work, can introduce 10 or more times the amount of sediment than a logging operation (Swift 1985; Taylor et al. 1995). Taylor et al. (1995) make a strong case for the use of portable longitudinal glued-laminated (glulam) deck timber bridges for stream crossings on temporary low-volume roads. Advantages of portable timber bridges include their light weight and ease of fabrication, transport, installation, and removal. Because portable timber bridges are re-usable (up to 10 times or more), installation cost is comparable with that of a permanent corrugated metal culvert at $2,550 per installation (Taylor 1995). In addition, a major advantage for water quality protection is that glulam bridges may be installed and removed with skidders or hydraulic knuckleboom loaders without operating the equipment in the stream channel (Carroll 2008).

McKee et al. (2010) surveyed Virginia logging contractors from the major physiographic regions of Virginia (Mountains, Piedmont, and Coastal Plain) to better understand the most typical stream crossing types installed, the total cost associated with purchasing and installing stream crossings, as well as type and cost of closure BMP implementation. The authors found that more stream crossings are used for skidders than for log trucks across all physiographic regions. Bridges are most commonly used for stream crossings in the Piedmont, while culverts predominate in the mountains. Costs associated with stream crossings range from most to least expensive as follows: steel bridges, wooden bridges, culverts, and fords. The most commonly used stream crossing closure BMPs include a combination of waterbars, seeding, and mulch. Additional BMPs include covering roads with slash and the installation of water turnouts. These BMPs have been shown to be generally effective in water quality protection (Carroll 2008). The cost of BMP closure implementation ranged from $445 to $655/crossing, with greater costs associated with BMP installation in the Mountain region.

Carroll (2008) evaluated upstream and downstream water quality, including sediment concentration, for 23 operational stream crossings in the Virginia Piedmont. Stream crossing structures included portable bridges, culverts backfilled with poles, culverts backfilled with earth, and reinforced fords. Water quality was monitored during four operational phases that included pre-installation, post-installation, during harvest, and post-road closure. Overall, this study found portable bridges to be the most effective for water quality protection, but that performance is also governed by road standards and approach characteristics. Importantly, this study found that the increased SMZ removal associated with permanent stream crossings may result in greater stream temperature increases.

2.3.2.4 Streamside Management Zones

In a watershed scale experiment, Lakel et al. (2010) evaluated the sediment trapping efficacy of various SMZ widths under different levels of thinning following forest harvesting and site preparation in the Piedmont physiographic region of Virginia. The study examined SMZs widths of 7.6-, 15.2- and 30.4-m in which no thinning occurred, as well as 15.2-m, thinned SMZs. All SMZ widths performed equally well in trapping sediment. This indicates that SMZ effectiveness is controlled by other factors in addition to width. Several keys to SMZ effectiveness in trapping sediment include the presence of an intact forest floor and slope steepness, suggesting that SMZ width prescriptions should be based on a site by site basis. This implies that through better understanding of the processes that control soil erosion, as well as BMP effectiveness in minimizing erosion and sediment redistribution, both water quality and site productivity objectives may be optimally achieved.

Lakel et al. (2010) also provided important data on soil erosion to sediment delivery ratios, determining that 3 to 14% of sediment from the harvested area reached the SMZ. This study examined not only SMZ sediment trapping effectiveness, but also quantified the amount and percentage of soil erosion and sediment delivery from harvest site preparation, roads, skid trails, decks, and firelines. As is most commonly found, highly compacted and bare soil areas such as roads, skid trails, and firelines contributed the most sediment to the SMZ. These areas of high erosion potential often represent a small percentage of the total operational area, but contribute the most sediment per unit area.

Rivenbark and Jackson (2004) examined the spatial frequency and physical characteristics of ephemeral concentrated flow paths entering SMZs for 30 clearcut and site prepared industrial forest operational units in the Georgia Piedmont. The impetus for this study was to aid in the understanding of where and why BMPs fail to prevent sediment from being transported to stream channels. Breakthroughs were defined as surface overland flow (and sediment) pathways that invaded the SMZ and reached the stream channel. Areas of convergence (swales) and gullies comprised about 50% of all breakthroughs, while concentrated runoff from roads and skid trails was identified as the cause of 25% of the breakthroughs. In general, large contributing areas (mean = 0.4 ha), minimal litter cover, and steep slopes characterized the locations where breakthroughs occurred. In some cases, overland flow traveled more than 100 feet through SMZs before reaching the stream channel. The authors concluded that improvements to increase BMP effectiveness include maximizing ground cover, improving road runoff dispersal, increasing resistance to probable surface overland flow paths, and selective increases of SMZ widths in problem areas.

Swift (1986) examined sediment transport distances below forest roads during and nine months after construction in the Appalachian Mountains of western North Carolina. The objectives of this study were to evaluate the effectiveness of filter strip standards in the southern Appalachian Mountains and to test the efficacy of mulch or grass on fill slopes and of obstructions to flow within filter strips. Guidelines for filter strip widths in the eastern U.S. originated from the Trimble and Sartz (1957) experiment, where the slope distance of sediment transport was determined from 36 open-top culverts on partially graveled roads at the Hubbard Brook Experimental Forest in New Hampshire's White Mountains. Slope steepness below the road was used to make recommendations for filter strip widths to effectively trap sediment (Trimble and Sartz 1957).

Swift (1986) found that grassed fill slopes, filter strips with intact forest litter cover, and brush barriers, such as logging slash and hay bales, were most effective in reducing sediment deposit length. Importantly, this study showed that filter strip width may be reduced if the aforementioned BMPs are implemented correctly, again implying that increased filter strip width does not necessarily mean that more sediment will be trapped. Filter strip width recommendations should be made on a case by case basis and should take into account slope steepness, forest litter layer condition, erosion potential from man-made (e.g. road ditches and culverts) and natural (areas of convergence, like gullies and swales) drainage structures, soil erosivity, and climate. In addition, the duration of soil exposure should be as short as possible and limited to periods of minimum rainfall intensity.

2.4 Research Needs

Anderson and Lockaby (2011) identified four categories of research gaps related to forest operations and stream sedimentation, including timber harvesting effects on water yield and water quality, temporal and spatial variability of sediment delivery, sediment and water yield from roads, and assessing the effectiveness of BMPs. Increases in stormflow volumes and peakflows following harvest operations may increase within-channel erosion, particularly in streams that are heavily impacted by legacy sediments. Legacy sediments confound evaluations of contemporary forest management practices because it is difficult to separate water quality impacts from past and present land use (Trimble 1974; Jackson et al. 2005). Incorporation of tracers (isotopic and radionuclide) to track sediment movement and statistical tools (Aikaike's Information Criterion and multivariate approaches) in current and future forestry studies could greatly improve understanding of different sediment sources (Anderson and Lockaby 2011).

We know that the majority of sediment associated with forest operations is generated from relatively small problem areas that are often non-uniformly distributed throughout the operational area. Often, these problem areas are associated with stream crossings and their associated skid trail and haul road approaches. Better quantification of soil erosion to sediment delivery ratios under various levels of BMP implementation is critical to improve the efficacy of reduced-impact forest practices. We also know that when BMPs are properly installed during forest operations, water quality generally remains unimpacted (Aust and Blinn 2004). However, more cost-benefit analyses of BMP implementation are required to achieve the major objectives of sustainable productivity and water quality protection at minimal cost.

In addition, more research is necessary to understand the impact of major rainfall events on the performance of BMPs in reducing soil erosion and sediment redistribution. Many studies are short in duration (<4 years) and may be heavily influenced by one or more major storm events. Conversely, a lack of rainfall may give undue credit to BMP effectiveness in protecting water quality. Simply put, rain events govern study findings (Anderson and Lockaby 2011). Rainfall simulation studies allow researchers to test BMP effectiveness over any desired range of rainfall conditions.

Much of the research specifically regarding forest roads and water quality in the United States has originated from the west (Reid and Dunne 1984; Anderson and Potts 1987; Litschert and MacDonald 2009), with far fewer studies investigating sediment delivery from forest roads in the east, with the exception of research from Coweeta and Fernow. Intensive forest management is ubiquitous throughout the Piedmont Physiographic Province. Therefore, an extensive network of forest roads and stream crossings exist in this region, where sediment delivery to upland headwaters has important water quality implications for the protection of downstream water bodies (Freeman et al. 2007). It is impractical, if not impossible, to monitor stormflow and soil erosion from all road and skid trail crossings through field experimentation. Therefore, models that are readily applicable to land management programs and that accurately represent hydrologic and soil erosion processes at the catchment-scale are critical to predict site- and regional-scale impacts to water quality. These hydrologic and soil erosion models may be used to assist land managers in identifying high-risk areas for erosion and implementing appropriate Best Management Practices (BMPs) for water quality protection. However, models must be evaluated to determine their utility in accurately representing field hydrologic conditions and sediment delivery ratios across a broad range of conditions.

Several State and Federal agencies, including the Virginia Department of Forestry and USDA Forest Service are interested in determining the applicability of the Water Erosion

Prediction Project (WEPP) model for predicting sediment production from forest roads. WEPP is a physically-based soil erosion and hydrologic model developed by the United States Department of Agriculture Natural Resources Conservation Service and Forest Service that estimates soil loss and sediment yields from hillslope erosion at the small catchment scale (Flanagan and Nearing 1995). WEPP is capable of partitioning soil loss and sediment yields associated with roads into individual road features, such as the road surface, cutslope, fillslope, ditch, and lower hillslope (Fu et al, 2010). Previous studies have shown that WEPP is a useful tool for estimating soil erosion from surfaces with low infiltration rates, such as forest roads, where overland flow is the dominant hydrologic process (Elliot et al. 1999; Laflen et al. 2004; Grace 2005; Croke and Nethery 2006; Dun et al. 2009; Fu et al. 2010), but WEPP has not been evaluated in the field for a wide range of forest roads and rainfall conditions.

Measurement of sediment yield and sediment delivery is highly variable, even under well-controlled field experiments. For example, Fu et al. (2010) describe variation as a result of differences in methodology. Much uncertainty exists in measured sediment yields when sediment trap data is used to estimate total sediment yields. This is because roadside sediment traps effectively sample coarse sediment yields, but may miss the finer sediment fractions of total sediment yield. In addition, road erosion rates display a wide range of variability across different areas owing to differences in rainfall timing, frequency, and intensity, as well as topography, slope, frequency of traffic and maintenance, and surface type. For example, Brooks et al. (2006) state that because measurements of soil erosion are so highly variable, predicted erosion rates should not be assumed to be more accurate than ± 50%. Laflen et al. (2004) reviewed published studies related to WEPP goodness-of-fit and suggest that without calibration, WEPP performs as well as USLE-Forest. This suggests that when controlled field experiments or databases of site-specific characteristics are used to parameterize the model, WEPP has the potential to be an effective tool for watershed managers.

Because WEPP mainly considers overland flow, it is currently best suited to predict runoff and soil erosion on surfaces where overland flow dominates hydrological processes. Substantial variability exists in model predictions of runoff and sediment yield in disturbed forest settings that are dominated by subsurface flow processes (Wu et al. 2000). However, it is arguable that for modeling sediment yield and delivery, which are governed by overland flow, WEPP performs best where it is needed the most (i.e. forest roads and skid trails). Therefore, WEPP can be very useful for small, high-risk road segments, such as road approaches to stream crossings. However, in order to compare relative sediment delivery ratios between roads and other forest practices not dominated by overland flow, model performance must be improved.

Further modifications to the processes that control subsurface flow within the WEPP model are necessary to better estimate runoff and erosion at the catchment scale. In addition, consideration of subsurface flow interception by forest roads should be considered in future modifications to the model. Controlled field experiments on forest roads and other disturbed forest areas (harvested, burned, site prepared) across a broad range of landscapes are necessary to test WEPP representation of hydrologic and soil erosion processes and evaluate uncertainty in model predictions.

2.5 Summary

This review concentrated on soil erosion and sediment delivery associated with reduced-impact forest operations in the Piedmont region. General results indicate that soil erosion per unit area is greatest for roads and skid trails, while comparatively less for harvested and site prepared areas. BMP implementation is most effective in water quality protection when prescriptions are made on a case by case basis and guided by characteristics such as percentage of bare soil, slope steepness, topographical features such as gullies and swales, and rainfall timing and magnitude. Much more is known about soil erosion rates, as opposed to sediment delivery ratios for various forest practices. Future research should utilize sediment tracing methods to identify sediment source areas that are often a small percentage of total forest operations and non-uniformly distributed. Coupling of well-controlled hydrologic/soil erosion field studies and soil erosion modeling is beneficial because it provides much-needed measurements of soil erosion and sediment delivery with which to calibrate and evaluate soil erosion model performance, as well as BMP performance under changing land use scenarios.

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3.0 SEDIMENT DELIVERY FROM BARE AND GRAVELED FOREST ROAD STREAM CROSSING APPROACHES IN THE VIRGINIA PIEDMONT

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3.1 Abstract

Forest road stream crossing approaches, or the section of road immediately adjacent to the stream crossing, represent primary sources and nearly direct pathways for sediment delivery to stream channels. This research quantified sediment delivery rates associated with reopening abandoned legacy road stream crossing approaches and evaluated the effectiveness of gravel surfacing of the entire running surface in reducing sediment delivery at stream crossings in the Virginia Piedmont. Sediment delivery rates from five regraded (bare) legacy road approaches were compared to those from four completely graveled road approaches. Repeated measurements of road derived sediment trapped by silt fences were used to quantify sediment delivery rates from the road approaches for one year (Aug. 5, 2011 to Aug. 5, 2012). Annual sediment delivery rates from the bare approaches were 7.5 times higher than those of the gravel approaches. Sediment delivery rates ranged from 34 to 287 Mg ha^{-1} year^{-1} for the bare approaches and from 10 to 16 Mg ha^{-1} year^{-1} for the graveled approaches. The highest sediment delivery rates were associated inadequate road surface cover and insufficient water control structures. These findings show that reopened legacy roads and associated stream crossing approaches can deliver significant quantities of sediment if roads are not adequately closed or maintained and that

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corrective best management practices (BMPs), such as gravel and appropriate spacing of water control structures, can reduce sediment delivery to streams.

3.2 Introduction

Soil erosion and subsequent sediment delivery to streams associated with low-standard roads is consistently cited as a primary source of pollution within forested land use (Anderson and Lockaby, 2011; Croke and Hairsine, 2006; Grace, 2005; Luce, 2002). Roads are an essential component for many forest management activities and provide access for timber harvest operations, site preparation activities, fire management, insect and disease control, and recreational opportunities. Road surfaces are compact and largely impervious by design to provide access for pre-planned traffic volumes, vehicle types, and loads, which are a function of land use objectives and tract area to be served by the road. Compact surfaces, including permanent and temporary roads, skid trails, and log decks, represent the primary instances in forested environments where infiltration-excess overland flow is possible even for low-intensity rain events (Ziegler et al., 2007). Subsurface hillslope flow interception from insloped or through-cut roads with cutslopes and ditches can dominate road surface runoff during rain events because the drainage area of the hillslope is often greater than that of the road surface (MacDonald et al., 2001; Wemple and Jones, 2003). This condition of enhanced infiltration-excess overland flow can increase road surface erosion.

Poorly designed or maintained forest road networks can increase hydrologic connectivity (drainage density) to streams by routing stormwater runoff through roadside ditches that connect directly to streams at road stream crossings, as well as further away from stream channels when gullies form below surface runoff relief culverts (Wemple et al., 1996). This increased hydrologic connectivity may impact the timing and magnitude of streamflow response to rain events and increase the frequency and magnitude of flood flows (Beschta et al., 2000; Eisenbies et al., 2007; La Marche and Lettenmaier, 2001). These direct hydrologic connections can adversely impact water quality through increased sedimentation from road erosion sources, while increased stormwater runoff may induce stream geomorphological changes, re-mobilize existing sediment stored within the stream channel, and result in the degradation of aquatic habitat (Goode et al., 2012).

Chronic fine sediment contributions associated with hydrologic connectivity between road networks and stream channels pose issues for water quality and aquatic habitat degradation (Goode et al., 2012; Robinson et al., 2010), which was underscored by the 2012 U.S. Supreme Court consideration of the Ninth Circuit Court ruling that was initiated by *Northwest Environmental Defense Center v. Brown.* The Ninth Circuit ruling stated that roadside ditches are point sources, requiring a National Pollution Discharge Elimination System (NPDES) permit, if they collect and deposit stormwater into the surface waters of the U.S. (Boston, 2012). The U.S. Supreme Court decision retained the nonpoint source pollution (NPSP) status of forest roads and silvicultural exemptions by reversing the Ninth Circuit ruling in March of 2013, but further litigation is possible until the NPSP status of forest roads is clarified. Nevertheless, it is clear that improved cost-effectiveness and implementation of forest road BMPs are critical for water quality protection.

Reduced-impact forestry utilizes BMPs to reduce the impacts of forest roads on water quality. BMPs for road design include adequate consideration of appropriate road standards and planning the layout of road networks to reduce the number of stream crossings and control road gradient (Walbridge, 1997). Water control structures, such as ditches with relief culverts, broad based dips, water bars, and turnouts, are used to drain insloped road surfaces and minimize the travel length of overland flow (Keller and Sherar, 2003). Vehicles with low-pressure tires may be implemented to minimize the impacts of traffic on road surface erosion (Foltz and Elliot, 1997). Road surfacing techniques, such as the use of gravel, are used to enhance trafficability and minimize soil erosion on active roads (Clinton and Vose, 2003; Kochenderfer and Helvey, 1987; Swift, 1984). During road closure, techniques to control erosion and sediment delivery include traffic restriction and natural vegetation reestablishment for temporary closure. Road decommissioning may also include soil ripping to alleviate compaction, recontouring and culvert removal to restore natural drainage patterns, replanting with native vegetation, and stream crossing removal and stabilization for permanent abandonment.

Research has shown that properly implemented BMPs reduce soil erosion and protect water quality (Anderson and Lockaby, 2011; Aust and Blinn, 2004; Stuart and Edwards, 2006). However, most BMP effectiveness studies have focused on quantifying soil erosion and not sediment delivery to stream channels from specific forest management operations (i.e., timber harvesting, site preparation, roads, skid trails, log decks) (Croke and Hairsine, 2006; Grace, 2005). The difficulty in quantifying sediment delivery ratios is that only a portion of upslope soil erosion reaches the stream due to factors such as the distance between the road and the stream, as well as watershed topographic characteristics (e.g., breaks in grade or depressions) and surface roughness features that act to trap and store sediment transported by surface runoff.

Research that has focused on quantifying sediment delivery potential from forest roads suggests that the degree of hydrologic connectivity of a road can be highly variable and site specific (i.e., dependent on catchment characteristics such as slope, road location, spacing of water control structures, and road and drainage density) (Takken et al., 2008). Sun and McNulty (1998) developed a sediment routing system by coupling a Geographical Information System

(GIS) and the Universal Soil Loss Equation modified for forest land (USLE-forest) (Dissmeyer and Foster, 1984) and calculated sediment delivery ratios for well-managed (0.15:1) and poorly managed (0.36:1) logging roads in the Blue Ridge Mountains of southwestern North Carolina. Previous studies in the Piedmont region of the U.S. have estimated sediment delivery ratios associated with harvesting, prescribed burning, and replanting (Lakel et al., 2010) and harvesting, mechanical and chemical site preparation, and planting (Ward and Jackson, 2004). In both studies, upslope erosion rates were modeled with either the Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1991) or USLE-forest (Dissmeyer and Foster, 1984) and compared with field measurements of trapped sediment at or within streamside management zone (SMZ) boundaries. Lakel et al. (2010) estimated sediment delivery ratios for forest operations (including roads and skid trails) to be between 0.03:1 and 0.14:1, while Ward and Jackson (2004) estimated a sediment delivery ratio of 0.25:1 for areas having mechanical site preparation. Both studies recognized the importance of adequate SMZs for trapping sediment before it entered streams and Lakel et al. (2010) emphasized that stream crossings were a major mechanism for increasing sedimentation by penetrating the SMZ.

Few BMP effectiveness studies have measured both soil erosion and sediment delivery rates associated with specific locations and management activities within forest operational areas. However, it is well established that BMP failures (i.e., problem areas for sediment delivery) are often non-uniformly distributed, represent small proportions of the total forest operational area, and contribute disproportionate amounts of sediment to the stream (Rivenbark and Jackson, 2004). Most often, these areas are associated with inadequate BMP implementation for forest roads and trails. These sediment delivery hotspots often exhibit one or more of the following characteristics: large contributing areas (mean = 0.4 ha (Rivenbark and Jackson, 2004)), areas of convergence (e.g., gullies and swales), compacted soils, minimal surface cover and roughness, and steep slopes (Lakel et al., 2010; Litschert and MacDonald, 2009; Rivenbark and Jackson, 2004; Swift, 1986).

Progress toward the prediction of high-risk areas for water quality degradation in light of current and future forest management operations include the development and field testing of soil erosion and hydrologic models and sediment tracing methods (Anderson and Lockaby, 2011; Fu et al., 2010). Concurrent field studies should focus on minimizing erosion where sediment has the highest probability of being delivered to the stream, while gaining valuable field data with which to test model performance. Hydrologic and soil erosion models have been developed to assist land managers in identifying high-risk areas for sediment delivery and implementing appropriate BMPs for water quality protection. Several state and federal organizations, including the Virginia Department of Forestry (VDOF), the State Foresters Council for the Southeast U.S., and the United States Department of Agriculture (USDA) Forest Service are interested in implementing the Water Erosion Prediction Project (WEPP) model to predict forest road soil erosion and sediment delivery. WEPP is a physics-based soil erosion and hydrologic model developed by the USDA Natural Resource Conservation Service and Forest Service that estimates soil loss and sediment yields from hillslope erosion at the small catchment scale (Flanagan and Nearing, 1995). WEPP is capable of partitioning road erosion and sediment delivery into individual road features, such as the road surface, cutslope, fillslope, ditch, and lower hillslope (Fu et al., 2010). Previous studies have shown that WEPP is potentially useful for estimating soil erosion from forest roads, where overland flow is the dominant runoff process (Dun et al., 2009; Elliot et al., 1999; Grace, 2005; Laflen et al., 2004). Empirical models, such as the Universal Soil Loss Equation adapted for forest land (USLE-forest) (Dissmeyer and Foster,

1984) and RUSLE2 (Foster, 2003) have been used comparatively with WEPP to predict erosion associated with various road closure BMPs for bladed and overland skid trails in the Virginia Piedmont (Sawyers et al., 2012; Wade et al., 2012). However, model performance has not been evaluated at the road-stream interface for a wide range of approach characteristics, BMP implementation, and rainfall conditions. Such approaches are critically important because the U.S. EPA is currently requesting that state forestry organizations further quantify the relationship between BMP compliance and sediment delivery, particularly from sediment producing areas such as roads. Use of calibrated models is often the only practical methodology allowing for wide scale examination of these linkages.

In terms of the likelihood of sediment delivery, forest road stream crossings represent one of the most direct pathways for overland flow and sediment transport to stream channels (Lane and Sheridan, 2002). Aust et al. (2011) evaluated upstream and downstream water quality, including suspended sediment concentration, for 23 operational stream crossings in the Virginia Piedmont. Overall, this study found that water quality was most negatively impacted during the installation and harvest operational phases, and that portable bridges were the most effective crossing type for water quality protection. The authors concluded that in addition to crossing type, the degree of water quality protection is heavily influenced by road standards and the characteristics (i.e., bare soil percentage, length, and slope) of road approaches to stream crossings. Wear et al. (2013) measured upstream and downstream concentrations of total suspended solids to quantify the effectiveness of skid trail stream crossing closure BMP implementation to reduce stream sedimentation at five harvest sites in the Virginia Piedmont. Surface cover treatments on the temporary skid trail approaches to stream crossings included (1) slash, (2) mulch and grass seed, (3) and mulch, grass seed, and silt fencing. BMP efficiencies, as adapted from Edwards and Williard (2010), were quantified for each of the treatments under different daily rainfall conditions following Clinton and Vose (2003). Slash reduced TSS by 97.2, 67.7, and 62.7% for the medium, high, and maximum rainfall conditions, respectively, and this treatment represented the most cost-effective option for the reduction of sediment delivery if slash application could be incorporated into the logging operation.

Time since disturbance also has been shown to be an important factor regarding potential water quality impacts of road erosion and sedimentation, with the majority of soil erosion occurring shortly after disturbance. Road decommissioning techniques such as recontouring the road bed to approximate the hillslope shape prior to road construction (Foltz, 2012; Madej, 2001), as well as road reopening (Foltz et al., 2009), and road and skid trail construction (Sawyers et al., 2012; Wade et al., 2012) result in freshly disturbed, unconsolidated soil that needs immediate protection. Reopening of legacy roads is a common practice in forest management and it is done to minimize the cost of constructing new roads. Foltz et al. (2009) noted that incentives for reopening legacy roads are increasing and examples include increased access for timber harvesting, hazardous fuel reduction efforts, and woody biomass utilization for energy. However, legacy roads may have lower road standards and BMP implementation than newly constructed forest roads. Therefore, legacy roads may require significant improvements to road gradient, placement of water control structures, and road surfacing materials, such as gravel or mulch, to minimize surface erosion and protect water quality.

In terms of soil erosion potential, there is arguably no better place to study the effectiveness of forest road BMPs in protecting water quality than the Piedmont region of the United States. Industrial forest operations are ubiquitous throughout the region and future increases in the demand for southeastern wood products may result in intensified forest road management activities and use of legacy roads (Wear and Greis, 2002). Future scenarios for forest management throughout the southeastern U.S. may include increased stand entry, shorter rotations, and higher overall production over fewer forested areas (Wear and Greis, 2002). In comparison to Mountain and Coastal Plain sites of the southeastern United States, suspended sediment production from Piedmont forestry operations is high because of the interaction between site preparation intensity and topographic relief (Riekerk et al., 1989), as well as clay rich soils. In addition, the Piedmont region has a history of past agricultural practices that have left a legacy of highly eroded landscapes and sediment-laden valley bottoms and stream channels (Jackson et al., 2005; Trimble, 1985).

The following research questions provided the impetus for this study: 1) What is the annual rate of sediment delivery from forest road stream crossing approaches due to road reopening in the Virginia Piedmont? 2) How do sediment delivery rates of reopened bare approaches compare with existing graveled approaches? 3) What are the major road approach characteristics above the stream crossing that govern rates of sediment delivery?

The main objective of this study was to quantify annual rates of sediment delivery from forest road stream crossing approaches with minimum surface cover (newly reopened bare legacy roads) and maximum surface cover (existing gravel roads) using sediment traps and repeated measures. Sediment delivery rates were compared by surface type (bare, graveled) to evaluate the sediment reduction efficacy of graveling the entire running surface of forest road approaches to stream crossings. An additional objective was to evaluate the extent to which observed sediment delivery rates were governed by road approach bare soil percentage, length (m), slope (%), rainfall amount (cm), rainfall intensity (cm hr^{-1}), and the number of days since road reopening or construction (time since disturbance). Road approach length, slope, and surface coverage represent management factors that can be controlled during pre-harvest planning, construction, and implementation phases. This study investigates their effects on soil erosion and sediment delivery and evaluates the effectiveness of related BMPs in minimizing sediment delivery to the stream. A secondary objective was to evaluate USLE-forest and WEPP model predictions of sediment delivery at forest road stream crossing approaches.

3.3 Methods

3.3.1 Study Area

Nine road segments in the southwestern Virginia Piedmont were selected for study of annual sediment delivery at the Reynolds Homestead Forest Resources Research Center (RHFRRC), located in Critz, Virginia (Patrick County) (Figure 3.1). As is representative of the Piedmont, topography is characterized by rolling hills, with side slopes generally ranging from 8 to 25% and a mean elevation of approximately 335 m above mean sea level (NRCS, 2011). Mean annual rainfall is 1250 mm, with a mean snow contribution of 270 mm to the total precipitation. Mean air temperature ranges from a low of -1.8°C in January to a high of 29.7°C in July (Sawyers et al., 2012). Hourly rainfall data for this study were obtained from a Soil Climate Analysis Network (SCAN) weather station located at RHFRRC (NRCS NWCC, 2010). The predominant soil series is Fairview sandy clay loam (fine, kaolinitic, mesic Typic Kanhapludults). Soil parent material is residuum from mica schist and mica gneiss. Soils are characterized as being moderately eroded and well drained, with the most limiting layer having saturated hydraulic conductivities ranging from 1.4 to 5.0 cm hr^{-1} (NRCS, 2011). Kadak (2012, unpublished data) used a double-ring infiltrometer to measure infiltration rates (cm hr^{-1}) from six forest road stream crossing approaches that were reopened by bulldozer blading at RHFRRC.

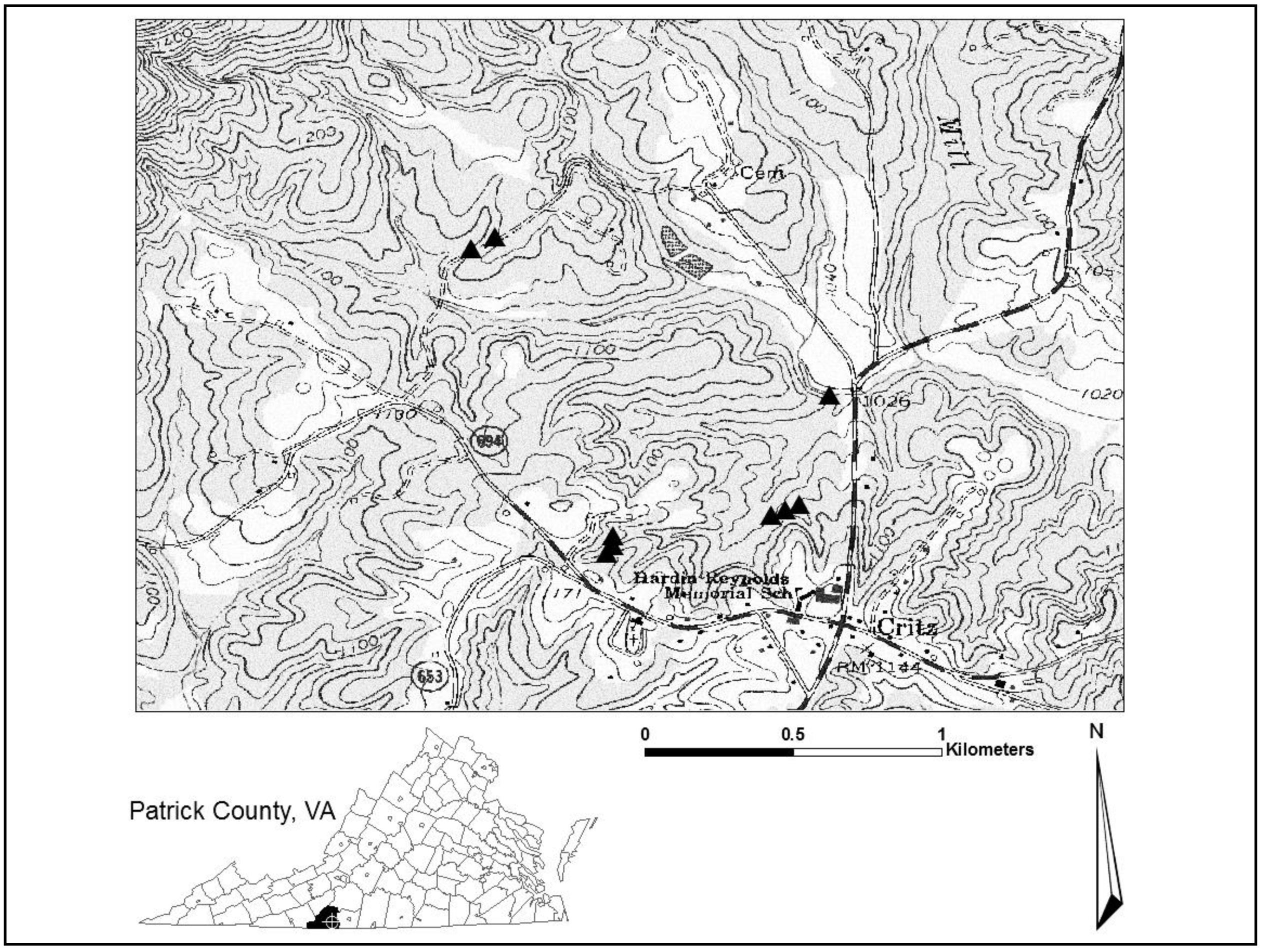

Figure 3.1. Location map of the Reynolds Homestead Forest Resources Research Center in Patrick County, Virginia, USA and a local topographic map depicting the proximity of the road approach study plots (black triangles) to one another.

Experiments were replicated based on road approach position (top, middle, and toe of the slope) for a total of 18 experimental units. Infiltration rates were analyzed with ANOVA by slope position and soil texture. No significant differences were detected ($p > 0.05$), but the experiments provided important data regarding infiltration rates from reopened bare forest road stream crossing approaches, which ranged from 0.06 to 0.72 cm hr^{-1}. As is typical of the Piedmont region, most of the forested watersheds have experienced prior conversions to agriculture during the 1800s and eroded landscapes and relic agricultural gullies are common (Trimble, 1974, 1985).

3.3.2 Site Survey

In July 2011, a total station (Sokkia total station model SET-520, Tokyo, Japan) was used to measure the length of the road approaches to stream crossings (Table 3.1). Length was defined as the distance between the nearest water control structure (i.e., rolling or broad-based dip) and the stream. Road approach slope and mean width of the road surface were also quantified during the total station survey. In addition, the Universal Soil Loss Equation modified for forest land (USLE-forest) (Dissmeyer and Foster, 1984) was used to estimate annual sediment delivery from the road approaches under pre-treatment conditions and hand texturing of the surface soil was used to estimate soil texture (NRCS, 2013). A previous study on nearby sites on the RHFRRC established that the USLE-forest provided acceptable estimates of measured erosion from bladed skid trails (Wade et al., 2012).

3.3.3 Treatment Installation

Five abandoned forest road segments were reopened by bulldozer blading in late-July 2011, creating initial conditions of 100% bare soil on the running surface, to simulate sediment delivery from reopening legacy roads (bare sites 1-5, Table 3.1). Two of the road segments represented road approaches to a 1970s era abandoned skidder crossing resembling an earthen dam (bare sites 1 and 2, Table 3.1). The remaining three road segments represented sections that were not immediately connected to the stream channel (bare sites 3-5, Table 3.1), but could become hydrologically connected to the stream during intense rain events through channel

Table 3.1. Road characteristics for the four graveled and five bare road approach study plots. Multiple segments at a site indicate a break in road gradient. Therefore, the total approach length is the sum of the multiple segments. Road surface bare soil percentages and USLE-forest erosion predictions (Mg ha^{-1} $year^{-1}$) correspond to site conditions immediately before and after bulldozer blading on July 31, 2011. Bare soil percentages and erosion predictions are the same before and after treatment for gravel sites 1-3 because no bulldozer work was performed at these sites.

Site, segment ID	**Crossing type**	**Length (m)**	**Running surface width (m)**	**Running surface area (m^2)**	**Slope (%)**	**Vertical shape**	**Soil texture**	**Bare soil (%), USLE (Mg ha^{-1} $year^{-1}$) before treatment**	**Bare soil (%), USLE (Mg ha^{-1} $year^{-1}$) immediately after treatment**
Gravel 1,2	Culvert	29.0	3.2	92.8	10.0	Linear	Sandy Clay Loam	5, 0.2	5, 0.2
Gravel 1,1	Culvert	11.0	3.0	33.0	12.0	Linear	Sandy Clay Loam	5, 0.2	5, 0.2
Gravel 2	Culvert	24.4	3.4	83.0	4.0	Linear	Sandy Clay Loam	5, 0.1	5, 0.1
Gravel 3,1	Culvert	12.5	3.4	42.5	2.0	Linear	Sandy Clay Loam	5, 0.0	5, 0.0
Gravel 3,2	Culvert	39.3	2.7	106.1	6.7	S-shape	Sandy Clay Loam	5, 0.1	5, 0.1
Gravel 4,1	Culvert	9.8	4.5	44.1	14.3	Linear	Silt Loam	10, 0.1	15, 1.4
Gravel 4,2	Culvert	31.7	5.2	164.8	19.0	S-shape	Silt Loam	20, 0.9	15, 4.6
Bare 1	Earth Dam	21.0	2.4	50.4	21.0	Convex	Clay Loam	65, 18.3	100, 130.5
Bare 2	Earth Dam	10.0	2.1	21.0	19.0	Convex	Silty Clay Loam	50, 7.8	100, 81.8
Bare 3,2	-	19.5	2.7	52.7	14.3	Convex	Silty Clay Loam	15, 0.3	100, 74.4
Bare 3,1	-	22.6	2.1	47.5	13.5	Linear	Silty Clay Loam	15, 0.3	100, 70.7
Bare 4	-	129.5	3.0	388.5	3.9	Linear	Loamy Sand	95, 23.9	100, 42.5
Bare 5	-	75.1	2.4	180.2	4.0	Linear	Silty Clay Loam	75, 2.5	100, 33.6

network extension, as road runoff becomes a tributary to the channelized runoff of legacy agricultural gullies. Where cut and fill road profiles existed, in-sloping (i.e. a slight cross-slope toward the cutslope) and outboard edge berm installation were used to redirect all runoff and sediment downhill to the bottom of the plot, where silt fence sediment traps were installed to trap all sediment from the road prism, similar to the method suggested by Robichaud and Brown (2002) (Figure 3.2).

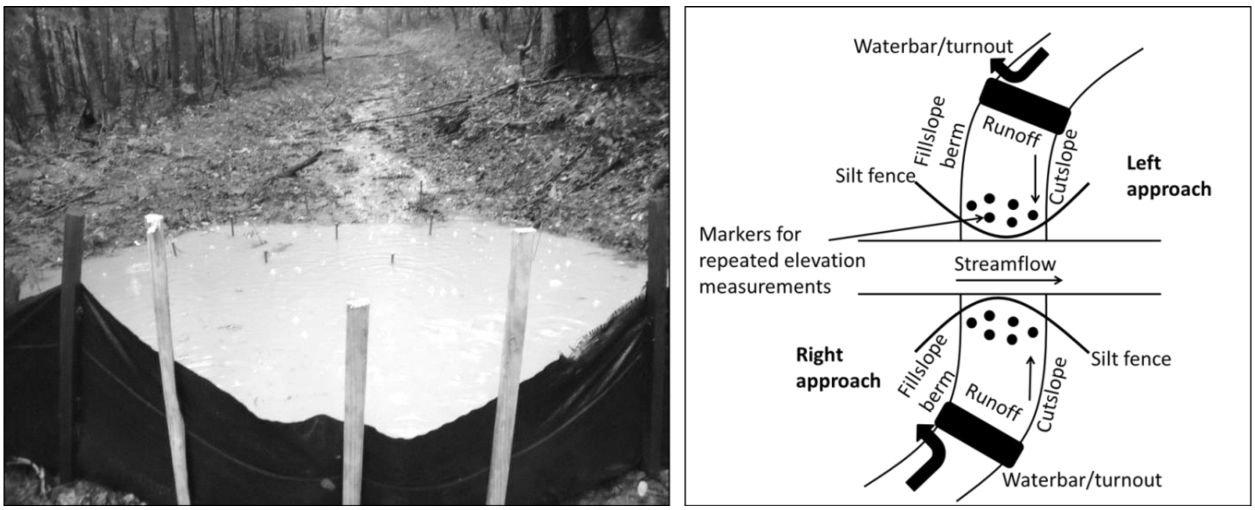

Figure 3.2. Left: Photo of bare site 3 during a rainfall event on June 22, 2012 from the vantage point of the silt fence. Landscape nails mark the locations of repeated elevation measurements and they can be seen above the pooled water surface at the uppermost part of the pool. Right: Schematic of two idealized road approaches to a stream crossing. A waterbar and turnout define the uppermost plot boundary and the silt fences at the bottom of the plot define the lower plot boundary.

Silt fence sediment traps were installed at the bottom of three culvert crossing approaches on a graveled road that was completed in November 2010 (gravel sites 1-3, Table 3.1). Finally, silt fence sediment traps were installed at the bottom of a legacy gravel road approach to a culvert crossing that was reshaped and re-graveled. Reshaping of the road surface in this case included in-sloping to the cutbank on the upper and middle portions of the hillslope and crowning near the silt fence at the bottom of the hillslope (gravel site 4, Table 3.1). Treatment installation resulted in five bare and four graveled study plots. Immediately following treatment installation, USLE-forest was also used to estimate sediment delivery from the study plots (Table 3.1). Vehicle traffic was not analyzed in relation to sediment delivery rates because traffic was restricted by sediment traps that were installed across the entire width of the road prism.

The road approach study plots differed with respect to canopy cover. For example, in July 2012, the hardwood forest canopy coverage over bare soil on the running surface for bare sites 1-3 ranged from 90 to 100% and the mean height of canopy drip, which approximates the midpoint of canopy height (Dissmeyer and Foster, 1984), was four meters. Bare site 4 was located in a four-year-old loblolly pine stand, which provided no canopy cover over the running surface. Adjacent forest cover for bare site 5 consisted of a four-year-old loblolly pine stand on one side of the road and a mix of hardwoods and pines on the other side. Canopy cover over bare soil at bare site 5 was approximately 10% and the mean height of canopy drip was eight meters. Gravel sites 1-3 had adjacent hardwood forest cover, but road daylighting resulted in zero to 40% canopy coverage over bare soil. Mean height of canopy drip was approximately eight meters. Gravel site 4 had adjacent hardwood forest cover, where canopy coverage over bare soil ranged between 50 and 85%, with a mean canopy drip height of 16 meters.

3.3.4 Field Measurements of Sediment Delivery

A network of 20.3-cm-tall landscape nails marked the locations for repeated elevation measurements of the sediment trapped by the silt fences. Pin flags were used to mark the locations for elevation measurements in cases where sediment deposition began to bury the landscape nails. Elevation measurements were made at approximately monthly intervals at each site by way of differential leveling of sediment elevations with a total station to approximate sediment deposition at the silt fences between Aug. 5, 2011 and Aug. 5, 2012. Twelve measurements of sediment delivery were made between Aug. 5, 2011 and Aug. 5, 2012 at each of the nine study sites. Sediment volumes (m^3) for each repeated measurement were calculated by multiplying road surface depositional area (m^2) by elevation gain (m). Sediment volumes were converted to sediment mass (Mg) by multiplying by bulk density (Mg m^{-3}) of the trapped sediment. Bulk densities were obtained from the trapped sediment via the soil extraction method (SSSA, 1986). Sediment masses corresponding to each repeated measurement were expressed as mass on a per unit area basis (Mg ha^{-1}) by dividing by road surface area (ha), and later summed over the course of the year to express sediment delivery on an annual basis (Mg ha^{-1} $year^{-1}$). Hourly rainfall data obtained from a NRCS SCAN weather station (NRCS NWCC, 2010) located approximately 0.8 km from the study sites were summed to calculate total rainfall in between repeated measurements of sediment delivery. Similarly, hourly rainfall data were used to calculate mean rainfall intensity (cm hr^{-1}) for rainfall events that occurred between repeated measurements of sediment delivery. Bare soil percentages associated with each measurement of sediment delivery were quantified by walking in a zigzag pattern on the running surface for the entire length of the road approach study plots and counting the number of steps where the toe of the boot fell upon bare soil (i.e., "bare" steps/total steps = bare soil percent).

Measurements of sediment delivery were not associated with individual rain events. Each repeated measurement of sediment delivery corresponded to the additional quantity of sediment trapped upslope of the silt fences as a result of the total rainfall amount (sum of individual rainfall events) since the previous measurement of sediment delivery. The effect of rainfall amount on sediment delivery varies in magnitude as a result of rainfall timing and intensity. In addition, road surface sediment sources may be limited immediately following a high-intensity rainfall event due to the wash-off of loose surface soil, with the result that subsequent rainfall events generate less sediment than would normally be expected. Other factors that cause temporal variability in the effect of rainfall on sediment delivery include variation in road surface cover (e.g., litterfall on the road surface) and the number of days since the most recent road construction or maintenance activity. Therefore, we used a repeated measures design to evaluate the effects of road surface type (bare, gravel), the number of days since disturbance, and rainfall amount per measurement interval on sediment delivery rates from forest road stream crossing approaches.

3.3.5 Erosion Model Estimates

USLE-forest uses the following site-specific data to predict long-term average soil losses resulting from sheet and rill erosion: long-term rainfall averages, soil erodibility values determined by the NRCS for a given soil series, and factors for slope length and steepness, soil cover, and management practices. The USLE equation is described below:

$$A = RKLSCP \qquad \text{Equation 3.1}$$

where A is soil loss per unit area, R is the rainfall and runoff factor, K is the soil erodibility factor, LS is the slope-length factor, C is the cover and management factor (including bare soil, residual binding, soil reconsolidation, mean canopy height and cover, stepped topography, onsite storage, vegetation), and P is the support practice factor.

An R-value of 175 was selected from the rainfall index map in the USLE-forest manual (Dissmeyer and Foster, 1984). A K-value of 0.24 for the appropriate horizon of the Fairview sandy clay loam soil series was selected from the Web Soil Survey (NRCS, 2011). LS factors were determined by using the slope effect chart in the USLE-forest manual (Dissmeyer and Foster, 1984) and the road approach lengths and slopes that were measured during the July 2011 total station survey. Multiple *LS* factors for a given site were determined for road approach study plots that had a significant change in road gradient. The *LS* factors for each site remained constant throughout the 1-year study period. The *C* values were determined using subfactors for disturbed soils, and each study plot was evaluated using procedures from the USLE-forest manual on Aug. 6, 2011, Mar. 25, 2012, and Jul. 31, 2012. Bare soil percentages were quantified for each sub-segment within the study plot. In the case of multiple road segments within a study plot, erosion was estimated for each sub-segment and a weighted average was used to determine a single soil erosion estimate for the entire study plot. In summary, USLE-forest was used three times during this 1-year study period (following bulldozer re-grading) to estimate soil erosion for each road approach study plot, and the erosion estimates for each site were averaged, resulting in nine soil erosion estimates (Mg ha^{-1} $year^{-1}$).

The WEPP model (ver. 2012.8) was used to build hillslope profiles that were representative of site conditions at the road approach study plots. Files related to slope, climate, soils, and vegetation management were created to build hillslope profiles for each of the nine road approach study plots. Data regarding road approach length, slope, road surface width, road vertical shape (concave, convex, linear, or S-shaped), and aspect were obtained during the July 2011 total station survey. Daily rainfall, minimum air temperature, and maximum air temperature data were downloaded from the NRCS SCAN weather station (NRCS NWCC, 2010) at RHFRRC for the period of Jan. 1, 2010 to Dec. 31, 2012. Within WEPP, a climate file was built by using the actual daily rainfall and temperature data, coupled with WEPP-simulated daily weather data, which was obtained through an interpolation of long-term climatic data for weather stations surrounding RHFRRC. Model estimates of soil erosion (Mg ha^{-1} $year^{-1}$) represented the total erosion resulting from simulated rain events from Aug. 5, 2011 to Aug. 5,

2012. A clay loam soil texture (from the Disturbed WEPP, “skid-clay loam” soil file) was used for all sites with the following parameter values: 2.6 mm hr^{-1} for effective hydraulic conductivity, 2 Pa for critical shear, 0.002 s m^{-1} for rill erodibility, and 4.0 x 10^6 kg s m^{-4} for interrill erodibility. The parameter value for effective hydraulic conductivity was selected based on infiltration measurements at RHFRRC (Kadak, 2012 unpublished data).

Parameter values for critical shear, rill, and interrill erodibility, as well as vegetation management procedures were based on communication with Elliot and Foltz (USDA Forest Service, pers. comm., July 2013). For the bare roads with no adjacent forest cover (bare sites 4 and 5), “Road-insloped bare ditch” was used for the vegetation management file (surface cover near 0%). For the bare roads with adjacent forest cover (bare sites 1-3), observed litterfall cover on the road surface from Fall 2011 through Spring 2012 (80 to 50% cover, respectively) was modeled by modifying the “Tall grass prairie” vegetation management file for local conditions, which included parameter changes for Julian day of senescence, biomass energy ratio (kg MJ^{-1}), and decomposition rate (unitless).

In addition, the “Forest Road (Blade)” tillage operation was used to represent road reopening by bulldozer blading in late-July 2011. The combined effect of the “Tall grass prairie” vegetation management file and the “Forest Road (Blade)” tillage operation was that rill and interrill cover started at zero percent (immediately after road reopening in late-July 2011), increased to approximately 80% by Nov. 2012, and decreased to approximately 50% by Jun. 2012, similar to the observed temporal variation in road surface cover. Surface cover management for the gravel roads (gravel sites 1-4) also involved modeling the combined effect of gravel and litterfall on road surface cover (85 to 95%) by adjusting parameter values for the day of senescence, biomass energy ratio, and decomposition rate within the vegetation management file (Elliot and Foltz, pers. comm., July 2013).

3.3.6 Erosion Model Evaluation

USLE-forest and WEPP model predictions of annual sediment delivery (Mg ha^{-1} $year^{-1}$) for the bare and gravel road approaches were compared to measured annual sediment delivery rates from the sediment trap study. Model performance was evaluated in terms of percent deviation from measured sediment delivery at each site:

Percent deviation = ((Predicted-Measured)/Measured)*100 Equation 3.2

In addition, models were evaluated for their ability to rank the sites in accordance with measured sediment delivery rates. In other words, can USLE-forest and WEPP rank the sites from "worst" to "best" (highest to lowest in terms of sediment delivery)? Finally, the models were evaluated for their ability to predict differences in sediment delivery by road surface type (bare, gravel).

3.3.7 Statistical Analyses

Measured rates of annual sediment delivery (Mg ha^{-1} $year^{-1}$) for the bare and gravel approaches were natural-log-transformed and a Shapiro-Wilk test of normality (Shapiro-Wilk W = 0.9, p = 0.24) was performed with JMP 10 software (SAS Institute Inc., 2012). Differences in the natural-log-transformed annual sediment delivery rates by road surface type (bare, gravel) were analyzed with a two-sample t-test in JMP 10 (N = 9, alpha = 0.10). We hypothesized that gravel roads would produce less sediment than bare roads (Gravel-Bare < 0). Therefore, the null hypothesis was that Gravel-Bare ≥ 0.

Repeated measurements of sediment delivery (Mg ha^{-1}) at the bare and gravel approaches from Aug. 5, 2011 to Aug. 5, 2012 were natural-log-transformed and analyzed using the SAS procedure PROC MIXED for repeated measures ANOVA (SAS Institute, Inc., 2012). Components of the repeated measures model included the main effect of road surface type (bare, gravel), time (Measurements 1-12), and the interaction between road surface type and time. Twelve measurements were made at each of the nine study sites over the course of the one-year study period for a total of 108 observations. Akaike's Information Criterion (AIC) was used to select the appropriate model covariance structure (Akaike, 1974; Moser, 2004). The autoregressive heterogeneous covariance structure had the lowest AIC value (280.4), and represented the best fit for the data. The repeated measures model evaluated the main effect of road surface type on sediment delivery over the course of the study, sediment delivery rate changes over time for the bare and gravel approaches, and differences in the sediment delivery rate changes over time between the bare and gravel approaches (i.e., the interaction between road surface type and time). Analysis of model residuals indicated that the assumptions of homogeneity of variance and normality (Shapiro-Wilk W = 0.98, p = 0.24) were met.

3.4 Results and Discussion

3.4.1 Annual Sediment Delivery Rates of Road Stream Crossing Approaches

The mean annual sediment delivery rate associated with the bare treatment plots was 98 Mg ha^{-1} year^{-1} and ranged from 34 to 287 Mg ha^{-1} year^{-1} (Figure 3.3). Sediment delivery rates were highest for the sites with the longest approaches and least surface cover. Road segment length for bare sites 4 and 5 was 129.5 and 75.1 meters, respectively, and sediment delivery rates were 287 and 85 Mg ha^{-1} year^{-1}. Approach length for bare sites 1-3 ranged from 10.0 to 42.1 meters and sediment delivery rates ranged from 34 to 41 Mg ha^{-1} year^{-1}. Despite having 4% slopes, the mean sediment delivery rate at bare sites 4 and 5 (186 Mg ha^{-1} year^{-1}) was 4.8 times higher than that of bare sites 1-3 (39 Mg ha^{-1} $year^{-1}$), which had 14 to 21% slopes. The high sediment delivery rates at bare sites 4 and 5 exemplify the importance of using water control structures at appropriate intervals and minimizing bare soil, even for low-gradient road segments with no traffic.

Annual sediment delivery rates of the reopened road approaches are within the range of soil erosion rates measured in previous studies at RHFRRC. Wade et al. (2012) measured annual erosion from bladed skid trails to evaluate the effectiveness of various levels of post-harvest skid trail closure BMP implementation in minimizing soil erosion. The skid trail plots were 15.2 meters in length and had 10 to 20% slopes.

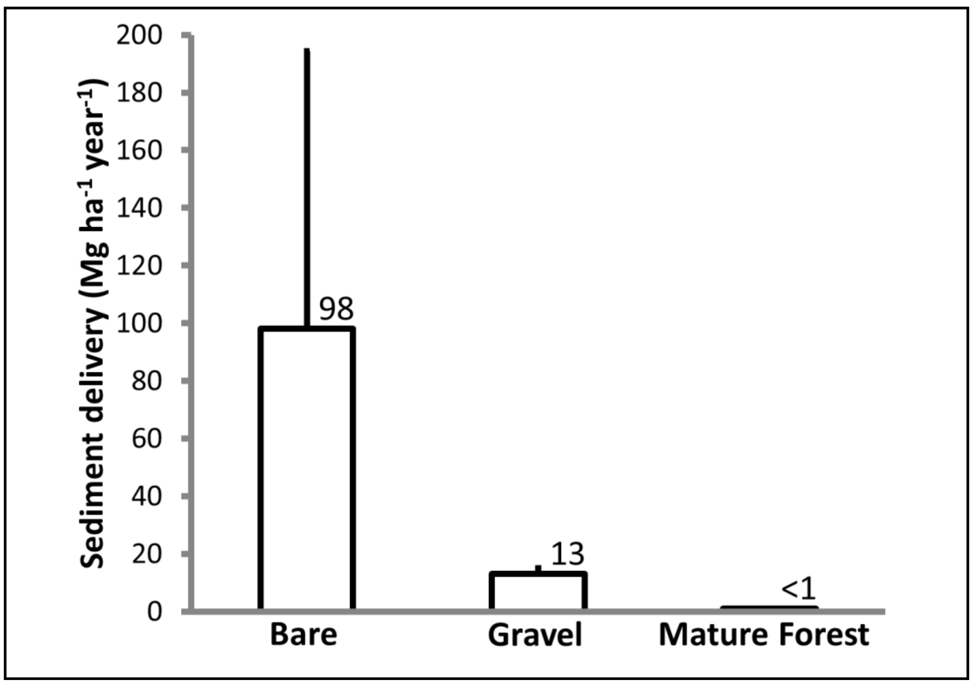

Figure 3.3. Mean annual sediment delivery rates (Mg ha^{-1} $year^{-1}$) at the bare and graveled approaches as compared with mature forest (from Patric, 1976). Error bars show standard deviation calculated from the population of measured sediment delivery rates at five bare and four gravel approaches.

For the bare treatment (water bar only), mean annual soil erosion was 125 Mg ha^{-1} year^{-1}. In a companion study, Sawyers et al. (2012) evaluated the efficacy of post-harvest overland skid trail closure BMP implementation. The skid trail plots were 15.2 meters in length and had 14 to 26% slopes. Control treatment (water bar only) skid trail plots generated a mean erosion rate of 24 Mg ha^{-1} year^{-1}. Erosion rates from the Sawyers et al. (2012) study were lower because overland skid trail construction resulted in less bare soil than bulldozer blading.

We found that the mean annual sediment delivery rate from the four graveled forest road approaches to stream crossings was 13 Mg ha^{-1} year^{-1} and ranged from 10 to 16 Mg ha^{-1} year^{-1} (Figure 3.3). For gravel sites 1-4, approach lengths ranged from 24.4 to 51.8 meters and slopes ranged from 4 to 17%. The mean annual sediment delivery rate from the bare approaches was 7.5 times higher than that of the gravel approaches. Bare site 4 was excluded from statistical analysis due to sediment transport being dominated by channelized surface runoff, as opposed to sheetflow, at this site (discussed further in 3.4.2.). Natural-log-transformed annual sediment delivery rates of the gravel roads were significantly less than those of the bare roads (t-test difference = -1.4, df = 4.2, p = 0.001). However, erosion rates from the gravel roads were still much higher than those of well-managed forest land, which typically range between 0.11 and 0.22 Mg ha^{-1} year^{-1} (Patric, 1976) (Figure 3.3).

3.4.2 Temporal Variability of Sediment Delivery Rates

Total rainfall during the one-year study period was 149.5 cm. Over half of the total rainfall (75.8 cm) occurred during the last quarter of the study (Apr. 6 to Aug. 5, 2012). Rainfall per repeated measurement ranged from a low of 3.9 cm from Jan. 16 to Feb. 6, 2012 (corresponding to measurement 7) to a high of 23.1 cm from Aug. 20 to Oct. 2, 2011 (corresponding to measurement 2). Median rainfall intensity (cm hr^{-1}) per repeated measurement was 0.23 and ranged from 0.08 during storms from Jan. 16 to Feb. 6, 2012 to a high of 0.39 during storms from Jun. 19 to Jul. 9, 2012. Median sediment delivery rate (Mg ha^{-1}) per repeated measurement was 2.0 for the bare plots and 0.3 for the gravel plots. The highest sediment delivery rates were measured on Oct. 2, 2011, which represented the greatest precipitation amount (23.1 cm) between measurements during this study (Figure 3.4). For the bare plots, this measurement interval also contained the first series of substantial rain events following bulldozer blading in late-July 2011. This finding suggests that extra care is necessary to control erosion and sediment delivery during any forest management activity that results in large quantities of bare soil (i.e., road construction, road reopening by blading, soil ripping, and road recontouring).

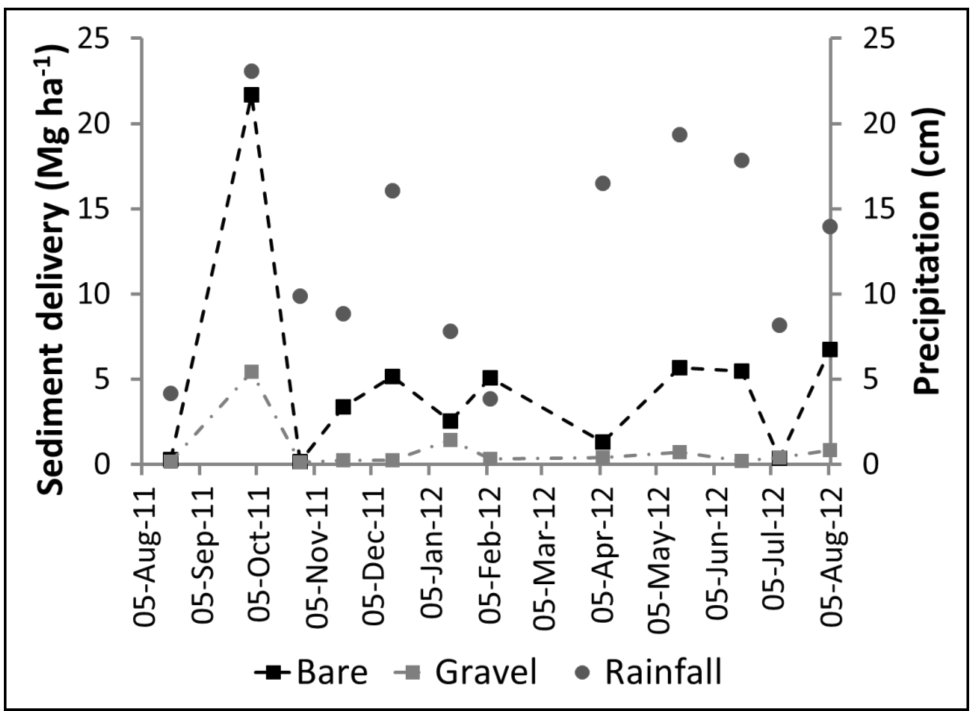

Figure 3.4. Median sediment delivery rate (Mg ha^{-1}) per repeated measurement by road surface type. The secondary y-axis depicts precipitation totals corresponding to the repeated measurements.

Sediment delivery rates were also impacted by the presence or absence of forest cover. For example, from Oct. 2011 through May 2012, leaf litter from adjacent hardwood forests provided substantial ground surface cover on bare plots 1-3 (Figure 3.5), helping to reduce soil erosion and sediment delivery. Leaf litter from hardwood forests also provided additional surface cover for gravel plots 1-4, but bare soil percentages were already low as a result of gravel surfacing. Bare soil percentages remained at 90-100% throughout the year for bare plots 4 and 5 due to a lack of any significant canopy or leaf litter coverage, which led to continued erosion and sediment delivery throughout the year (Figure 3.6).

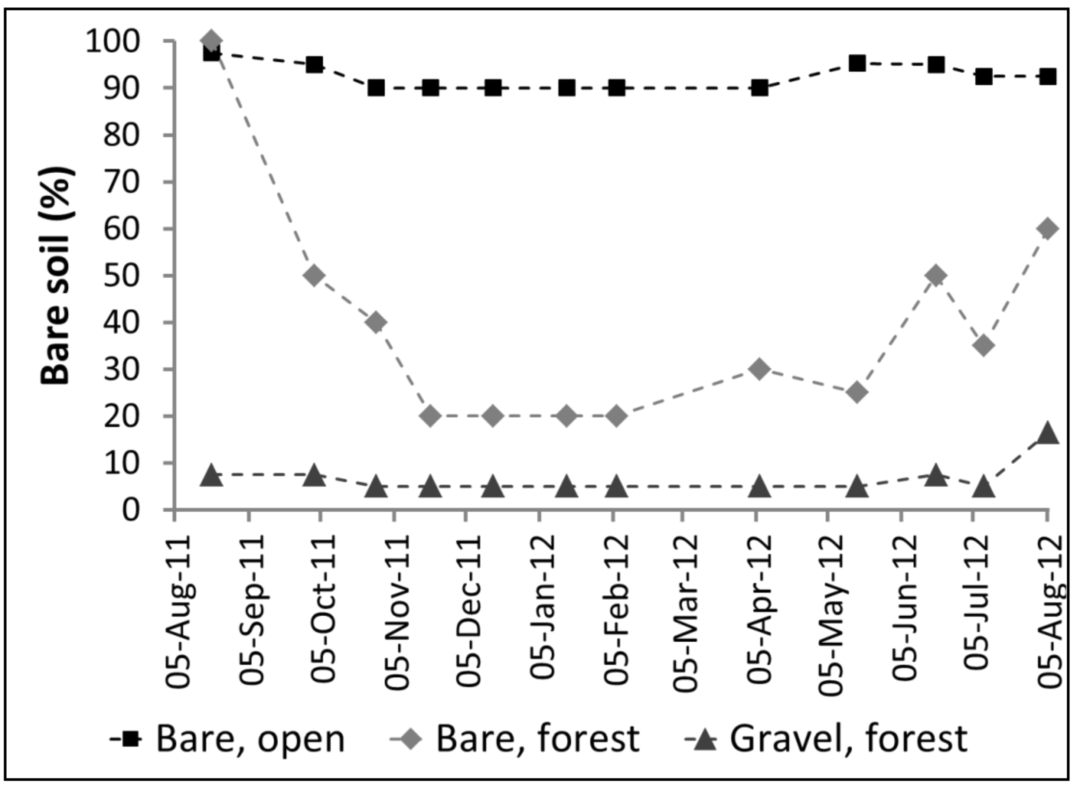

Figure 3.5. Bare soil percentages on the road surfaces throughout the study. “Bare, open” sites included bare plots 4 and 5. “Bare, forest” sites included bare plots 1-3 and “Gravel, forest” sites included gravel plots 1-4. Leaf litter provided ground cover for the forested sites during the fall, winter, and spring seasons.

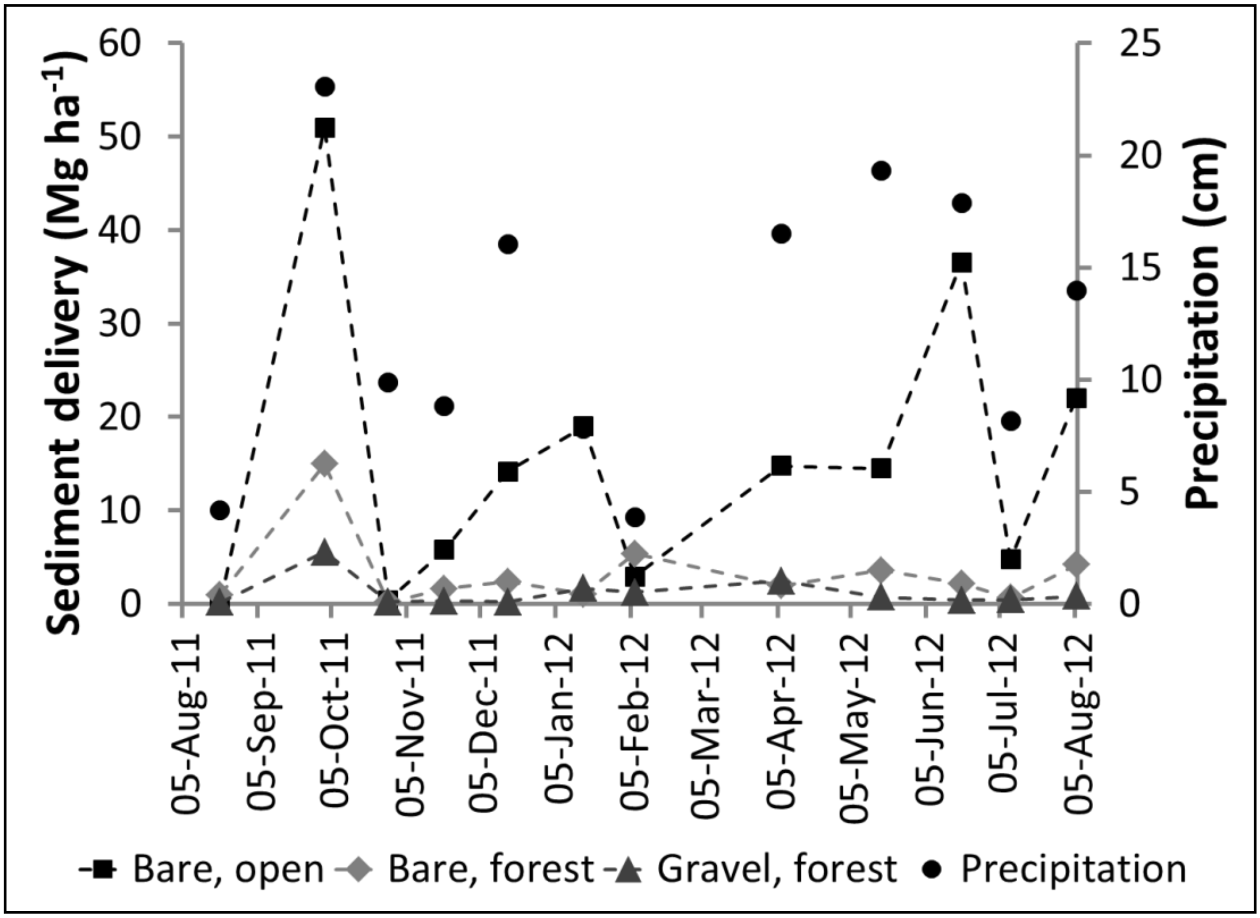

Figure 3.6. Sediment delivery rate (Mg ha^{-1}) and rainfall amount (cm) per repeated measurement by road surface and forest cover group.

Bare site 4 was not included in the repeated measures analysis because the severe erosion processes associated with such a large drainage area (388.5 m^2, Table 3.1) and concomitant minimal surface cover (90-100% bare soil) were not seen at any of the other road approach study plots. Surface runoff occurred largely as sheetflow at the other road approach study plots, whereas channelized runoff dominated sediment transport at bare site 4 (Figure 3.7).

Figure 3.7. Runoff conditions at bare site 4 following a thunderstorm on Jun. 22, 2012. Notice the channelized runoff.

Analysis of the natural log of repeated measurements of sediment delivery by road surface type revealed that the main effect of gravel was significant ($p = 0.0008$) in reducing sediment delivery (Table 3.2). Sediment delivery rates at the bare and gravel approaches changed over time ($p < 0.0001$), with both groups experiencing the greatest sediment delivery rates for the second repeated measurement (Oct. 2, 2011), followed by a leveling off of sediment delivery for the rest of the study period (Table 3.2). Sediment delivery rate changes over time at the bare and gravel roads were not significantly different ($p = 0.50$), meaning that there was not a significant interaction between road surface type and time (Table 3.2).

Table 3.2 Results of the SAS procedure PROC MIXED for repeated measures ANOVA used to analyze the effect of road surface type, time, and their interaction on road approach sediment delivery rates.

Effect	**Numerator df**	**Denominator df**	**F value**	**Pr > F**
Road surface	1	6	38.32	0.0008
Time	11	66	13.60	<0.0001
Surface*Time	11	66	0.95	0.4979

3.4.3 Comparison of Measured and Predicted Soil Erosion Rates

Neither erosion model (USLE-forest, WEPP) performed well in predicting annual sediment delivery rates from the forest road stream crossing approaches. Percent deviations from measured sediment delivery rates ranged from -98 to 61 for USLE-forest and from -100 to 46 for WEPP (Table 3.3). Percent deviations for both models were similar to the ranges reported by Sawyers et al. (2012) (-78 to 57 for USLE-forest and -38 to 19 for WEPP). The wide disparities between predicted and observed sediment delivery rates are not uncommon. In reference to the large-scale prediction of sediment delivery from forest road networks, Brooks et al. (2006) stated that predicted erosion rates should not be assumed to be any more accurate than ±50%, owing to the inherent natural variability in soil properties, annual climatic variability, and a lack of field data to estimate model parameters.

However, USLE-forest and WEPP both performed well in identifying problem road approaches. Both models predicted substantial annual sediment delivery rates for sites with inadequate surface cover and minimal sediment delivery rates for the gravel approaches. WEPP performed better than USLE-forest in ranking the problem road approaches, which is a critical model capability in prioritizing BMP implementation to reduce sediment delivery from road approaches.

Table 3.3. Mean annual sediment delivery rates for the five bare and four gravel road approach study plots based on measured sediment delivery rates from the sediment trap experiment and predicted sediment delivery rates from USLE-forest, WEPP, and their associated percent deviations.

Site	**Measured erosion ($Mg\ ha^{-1}\ year^{-1}$)**	**Modeled erosion**			
		USLE-forest		**WEPP**	
		Erosion estimate ($Mg\ ha^{-1}\ year^{-1}$)	**Percent deviation (modeled v. measured)**	**Erosion estimate ($Mg\ ha^{-1}\ year^{-1}$)**	**Percent deviation (modeled v. measured)**
Bare 4	287	44.8	-84	245.4	-14
Bare 5	85	29.1	-66	124.4	46
Bare 3	41	51.2	25	6.1	-85
Bare 1	41	66.2	61	3.1	-92
Bare 2	34	43.1	27	1.6	-95
Gravel 2	16	0.3	-98	0.0	-100
Gravel 1	13	0.7	-95	0.0	-100
Gravel 4	12	2.5	-80	0.0	-100
Gravel 3	10	0.3	-97	0.0	-100

3.4.4 Potential for Reducing Sediment Delivery by Using Contemporary Virginia Forest Road BMPs

Virginia revised its technical manual for forestry BMPs in 2011 (VDOF, 2011). The manual suggests that road grades be kept between 2 and 10% whenever possible. For permanent roads or roads that are currently in use, broad-based or rolling dips that are slightly out-sloped should be used at appropriate intervals to redistribute runoff from the road surface. For 2 to 12% road gradients, broad-based dips should be located at corresponding intervals of 91.4 to 41.1 m. For road gradients ranging from 2-5% to over 16%, rolling dips should be spaced at corresponding intervals of 54.9 to 36.6 m. In addition, gravel, mulch, or other suitable material should be used to stabilize road approaches to stream crossings for a minimum of 15.2 m on either side of the crossing, or to the top of the grade that is contributing sediment to the stream crossing. Water turnouts should be located at least 7.6 m before stream crossings.

In this study, road approach length and surface cover outweighed the effects of slope on sediment delivery. Sites bare 1 and 2 had slopes of 21% and 19%, respectively, but leaf litter coverage from Fall 2011 through Spring 2012 helped to reduce sediment delivery. Gravel sites 1 and 4 were also steep (10 to 19% slopes), but gravel surface coverage reduced sediment delivery. In addition, the effects of road approach slope on sediment delivery were dampened by the highest sediment delivery rates being measured on gently-sloping (4%) road segments (i.e., bare sites 4 and 5). However, these sites were characterized by 130 and 75 m in between water control structures and nearly 100% bare soil conditions throughout the year. Previous studies have found slope to be an important factor governing sediment delivery (Lakel et al., 2010; Litschert and MacDonald, 2009; Rivenbark and Jackson, 2004; Swift, 1986; Trimble and Sartz, 1957).

All of the bare treatment plots in this study were located on reopened legacy roads that were designed and constructed prior to the advancements of contemporary forest road BMPs. Utilization of legacy roads is common during forest management. However, legacy roads may require significant re-grading, water control structure installation, and gravel application in order to protect water quality. In this study, road segment length was an important variable governing sediment delivery rates. The high sediment delivery rates at bare sites 4 and 5 (287 and 85 Mg ha^{-1} $year^{-1}$) resulted from 90 to 100% bare soil conditions throughout the year, as well as inadequate spacing of water control structures (130 and 75 m, respectively), despite having gentle road gradient (4%). With contemporary BMPs, these road segments should have rolling dips at 54.9-m intervals or broad-based dips at 61.0-m intervals. Bare site 3 (41 Mg ha^{-1} $year^{-1}$) had adjacent forest cover, was 42.1-m in length, and had a mean slope of 14%, which is adequate in terms of the spacing of water control structures by the standards of contemporary BMPs. Bare sites 1 and 2 (34 and 41 Mg ha^{-1} $year^{-1}$) were short (21.0 and 10.0-m in length), steep (20% slope) approaches to an earthen fill crossing with adjacent forest cover. The profiles of the road approaches are both through-cuts, which prevents the installation of water turnouts. However, complete graveling of the approaches would reduce soil erosion rates.

Gravel site 4 represented a graveled legacy road approach to a culvert crossing. The site had adjacent forest cover, a length of 41.5 m, a mean slope of 18%, and a mean annual sediment delivery rate of 12 Mg ha^{-1} $year^{-1}$. Although this site has a change in grade from 19.0 to 14.3% at a distance of 9.8 m from the stream crossing, contemporary BMPs would suggest a water turnout at least 7.6 m before the stream crossing. Gravel sites 1-3 (mean = 13 Mg ha^{-1} $year^{-1}$) represent a road designed and constructed with contemporary BMPs. All of the approaches are graveled to the top of the approach that is contributing sediment to the stream and the maximum grade of any of the road segments is 12%, which acted to reduce sediment delivery.

3.4.5 Research Questions Revisited

3.4.5.1 What is the Annual Rate of Sediment Delivery from Forest Road Stream Crossing Approaches due to Road Reopening in the Virginia Piedmont?

Bare sites 1 and 2 had sediment delivery rates of 34 and 41 Mg ha^{-1} year^{-1}, respectively. Sediment delivery rates measured at bare sites 3, 4, and 5 were 41, 287, and 85 Mg ha^{-1} year^{-1}, respectively. The highest sediment delivery rates were associated with inadequate spacing between water control structures, 90-100% bare soil conditions throughout the year, and a lack of forest cover.

3.4.5.2 How do Sediment Delivery Rates of Reopened Bare Approaches Compare with Existing Graveled Approaches?

The mean annual sediment delivery rate from the four graveled forest road approaches to stream crossings was 13 Mg ha^{-1} year^{-1} and ranged from 10 to 16 Mg ha^{-1} year^{-1}. The mean annual sediment delivery rate from the bare approaches was 7.5 times higher than that of the gravel approaches. These study findings support the contemporary BMP recommendation to gravel road segments to the top of the approach that is contributing sediment to the stream and to redistribute stormwater runoff from the road surface at least 7.6 m before the stream crossing.

3.4.5.3 What are the Major Road Approach Characteristics above the Stream Crossing that Govern Rates of Sediment Delivery?

Road approach length and bare soil percentage were the most important factors controlling sediment delivery. Fortunately, road approach length and bare soil percentage are both factors that can be controlled. Therefore, study findings support contemporary BMP recommendations for the spacing of water control structures at appropriate intervals and to stabilize road approaches near stream crossings with gravel, mulch, or other suitable material. In addition, the presence of adjacent forest cover helped to reduce bare soil percentages on the road approach study plots during the fall, winter, and spring months by covering the running surface with leaf litter. This effect was most pronounced at bare sites 1-3, which had the greatest canopy coverage. Currently, there are no BMPs that provide specifications for suggested forest cover alongside forest roads to control soil erosion and sediment delivery. Conversely, daylighting is a technique used to allow sunlight drying of active roads and minimization of erosion resulting from vehicle traffic on a wet road surface. Residual trees alongside closed forest roads could add litterfall coverage to the road surface, but it is likely that existing BMPs, such as spreading slash on bare soil surfaces, would be more advantageous for erosion control because slash coverage is immediate, controllable, and takes much longer to decompose in comparison to litterfall. Slash can also act as a deterrent to ATV traffic on closed roads.

3.5 Conclusions

Forest operations often involve the development of new forest roads or the use of existing legacy roads. With new construction it is easier to design and construct roads having acceptable locations, grades, water control, and surfacing. Legacy roads are used to allow access from pre-existing access agreements, to fit difficult access conditions, and to minimize the cost of new construction, yet they may have lower road standards and require additional measures to minimize water quality problems, especially if stream crossing approaches are steep, bare, and have inadequate water control. Forest road approaches to stream crossings represent areas with relatively high potential for water quality impacts and extra precautions against direct sediment inputs must be made.

This study evaluated sediment delivery rates associated with reopening abandoned legacy road segments. Sediment delivery rates for existing gravel road approaches to stream crossings were also evaluated, as well as the road approach characteristics that are most strongly associated with measured sediment delivery rates. Study findings suggest that reopening legacy roads can generate significant quantities of sediment if roads are simply bladed and no improvements to the spacing of water control structures or road surfacing techniques are made upon reopening. Bare road segments generated 7.5 times more sediment than graveled road segments. Problem roads that delivered the most sediment were characterized by excessive lengths in between water control structures and inadequate surface cover. These research outcomes support the judicious implementation of BMPs for road segments that are high-risk areas for water quality impairment. These results emphasize that the use of appropriate BMPs can minimize sediment contributions from forest roads, even in situations where the original road design was not ideal.

3.6 Acknowledgements

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4.0 THE EFFECT OF INCREASING GRAVEL COVER ON FOREST ROADS FOR REDUCED SEDIMENT DELIVERY TO STREAM CROSSINGS

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4.1 Abstract

Direct sediment inputs from forest roads at stream crossings are a major concern for water quality and aquatic habitat. Legacy road-stream crossing approaches, or the section of road leading to the stream, may have poor water and grade control upon reopening, thus increasing the potential for negative impacts to water quality. Rainfall simulation experiments were conducted on the entire running surface area associated with six reopened stream crossing approaches in the southwestern Virginia Piedmont physiographic region, USA. Event-based surface runoff and associated total suspended solids (TSS) concentrations were compared among a succession of gravel surfacing treatments that represented increasing intensities of best management practice (BMP) implementation. The three treatments were No Gravel (10-19% cover), Low Gravel (34-60% cover), and High Gravel (50-99% cover). Increased field hydraulic conductivity was associated with maximized surface cover, and ranged from 7.2 to 41.6, 11.9 to 46.3, and 16.0 to 58.6 mm h^{-1}, respectively, for the No Gravel, Low Gravel, and High Gravel

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treatments. Median TSS concentration of surface runoff for the No Gravel treatment (2.84 g L^{-1}) was greater than Low Gravel (1.10 g L^{-1}) and High Gravel (0.82 g L^{-1}) by factors of 2.6 and 3.5, respectively. Stream crossing approaches with 90-99% surface cover had TSS concentrations below 1 g L^{-1}. Reducing the length of road segments that drain directly to the stream can reduce the costs associated with gravel surfacing. This research demonstrates that judicious and low-cost BMPs can ameliorate poor water control and soil erosion associated with reopening legacy roads.

4.2 Introduction

Forest roads are the dominant sediment sources associated with forested land use (Croke et al., 1999a). The potential for sediment delivery to waterways is greatest where the road meets the riparian area because forest road-stream crossings penetrate riparian vegetation and represent relatively unimpeded paths for overland flow to the stream channel. The direct nature of sediment delivery to streams at road-stream crossings has sparked a series of recent legislative debates in the United States about Federal Clean Water Act permits for forest roads (i.e., *NEDC v. Brown* (Boston, 2012)), as well as their nonpoint-source pollution status (i.e., *NEDC v. Decker* (U.S. Court of Appeals for the Ninth Circuit, 2013)).

Major sediment inputs to streams occur during stream crossing installation (Lane and Sheridan, 2002; Wang et al., 2013) and removal (Foltz et al., 2008b), but stream crossing approaches can also represent significant sediment sources (Lane and Sheridan, 2002; Aust et al., 2011; Brown et al., 2013). The potential for road-stream crossings to negatively impact water quality relates to crossing type, traffic type, vehicle weight, traffic frequency, surface cover, road approach length and slope, and the effectiveness of water-control structures (Reid and Dunne, 1984; Sheridan and Noske, 2007). In addition, reopening abandoned "legacy" roads that were originally constructed prior to the BMP era in the U.S. (circa 1977) raises concerns about their potential impacts to water quality. Common water quality problem areas associated with legacy roads include poor road location, as well as inadequate water and grade control. The effects of forest roads on water quality and aquatic habitat may be more pronounced for small streams (< 10 L s^{-1}), while the impacts of road-derived sediment may be diluted on larger streams (> 500 L s^{-1}) (Sheridan and Noske, 2007).

Few studies have quantified the efficacy of best management practices (BMPs) to reduce soil erosion and sediment delivery, as well as BMP implementation costs (Brown et al., 1993; Sawyers et al., 2012; Wear et al., 2013), but both are critical to justify BMP implementation. In particular, how effectively sediment is reduced by different intensities of BMP implementation or how effective BMPs are under the influence of individual rain events with different characteristics is not known. Rainfall simulation experiments at stream crossing approaches can be used to provide insights about both of these factors.

This study focuses on three research questions in the Piedmont physiographic region of southwestern Virginia, USA: (1) How does gravel cover affect road surface runoff?; (2) What are the total suspended solids (TSS) concentrations in road surface runoff during rainfall events for reopened stream crossing approaches?; (3) What sediment reduction can be achieved using different graveling intensities on stream crossing approaches and what is the cost of reducing sediment by graveling? Rainfall simulation experiments were conducted on the entire running surface area associated with six reopened stream crossing approaches. Event-based road surface runoff and associated TSS concentrations were compared among a succession of treatments that represented increasing intensities of BMP implementation. The three treatments were (1) No Gravel: existing conditions following reopening by bulldozer blading (10-19% cover); (2) Low

Gravel: gravel application beginning at the stream and continuing uphill for 9.8 m (34-60% cover); and (3) High Gravel: gravel application that doubled the length of the Low Gravel treatment (50-99% cover). The cost-effectiveness of the gravel treatments was evaluated by relating the costs associated with BMP implementation to reductions in sediment yield.

4.3 Materials and Methods

4.3.1 Study area

Six road approaches associated with three unimproved ford stream crossings on a legacy forest road at the Reynolds Homestead Forest Resources Research Center (RHFRRC), located in Critz, Virginia (Patrick County), USA, were selected for study of event-based surface runoff and associated TSS concentrations (Figure 4.1). Topography is characterized by rolling hills, with side slopes generally ranging from 8 to 25% and a mean elevation of approximately 335 m above mean sea level (NRCS, 2011). As is typical of the Piedmont region, old agricultural gullies are common because most of the contemporary forested watersheds were in agriculture during the 1800s (Trimble, 1974).

Mean annual rainfall is 1250 mm, with a mean snow contribution of 270 mm to the total precipitation. Mean air temperature ranges from a low of -1.8°C in January to a high of 29.7°C in July (Sawyers et al., 2012). The predominant soil series is Fairview sandy clay loam (fine, kaolinitic, mesic typic Kanhapludults). Soil parent material is residuum from mica schist and mica gneiss. The severe erosion hazard rating for forest roads and trails at RHFRRC (NRCS, 2011) underscores the importance of controlling road grade, water, and surface cover to reduce erosion and sediment delivery.

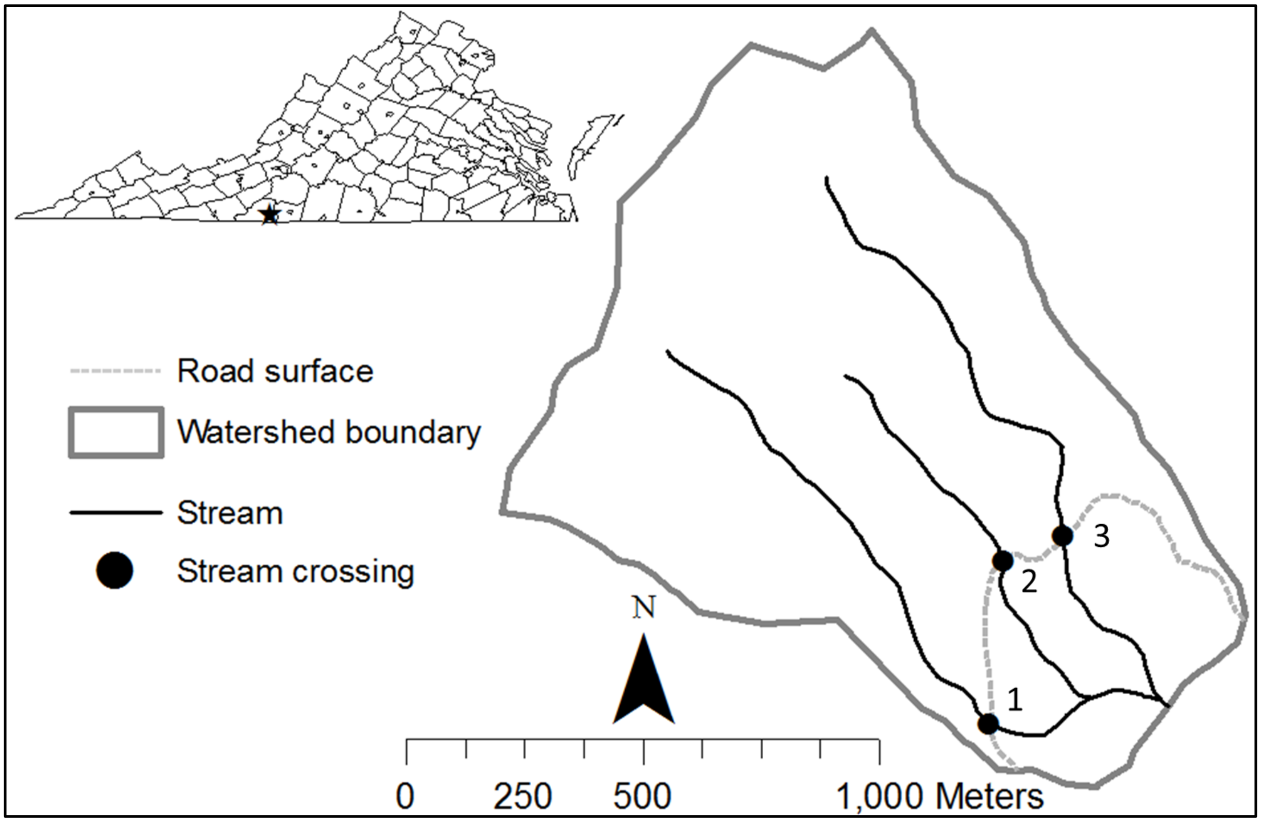

Figure 4.1. Location map of the Reynolds Homestead Forest Resources Research Center in Critz, Virginia (Patrick County), USA, and a schematic showing the road location within the 2nd-order watershed that contains the three unimproved ford stream crossings. This 135-ha watershed drains to the southeast from No Business Mountain and is part of the western headwaters of Mill Creek.

4.3.2 Field methods

Prior to road reopening, a Sokkia model SET-520 total station was used to measure the length of the stream crossing approach study plots, as well as approach slope and mean running surface width (Table 4.1). Length was defined as the distance between the nearest water-control structure (i.e., water bar/turnout) and the stream. In late-July 2011, six stream crossing approaches were reopened by bulldozer blading, creating initial conditions of approximately

100% bare soil on the approach running surfaces. The road profiles of each approach study plot were cut-and-fill.

Table 4.1. Physical characteristics of the running surface component of the road-stream crossing approach study sites at the Reynolds Homestead Forest Resources Research Center, Critz, Virginia (Patrick County), USA.

Site ID	Approach length (m)	Mean width (m)	Surface area (m^2)	Mean slope (%)	Bulk density (g cm^{-3})	Soil texture*	Road vertical curvature	Running surface profile
1	25.8	3.1	79.4	9.4	1.61	SCL	S-shaped	Insloped
2	41.3	2.5	102.6	4.7	1.55	SCL	Linear	Insloped
3	19.2	3.2	61.3	13.7	1.51	SCL	Convex	Insloped
4	35.3	2.5	86.8	15.4	1.46	SCL	S-shaped	Insloped
5	39.6	2.9	115.7	16.1	1.37	SCL	S-shaped	Insloped
6	23.5	2.6	61.3	6.4	1.47	CL	Linear	Insloped

*SCL and CL denote sandy clay loam and clay loam soil textures, respectively.

Bulldozer blading created a slight cross-slope in the running surface, which directed surface runoff and suspended sediment toward the cutslope bank en route to the bottom of the plot, which ended just before the stream (Figure 4.2). Outboard-edge berm installation was also used to prevent surface runoff from traveling away from the running surface and down the fillslope bank. Open-top box culverts (Trimble and Sartz, 1957) were installed at the lower plot boundary to redirect stormwater runoff away from the stream and toward a flume, where runoff volumes were measured. In October 2011, Kadak (2012, unpublished data) used double-ring infiltrometers to estimate infiltration rates of the reopened stream crossing approaches. Infiltration rates ranged from 0.6 to 7.2 mm hr^{-1}. Bulk density samples (n = 4 to 7 per site) were obtained from the running surface via the soil extraction method (SSSA, 1986).

Rainfall simulation experiments were conducted for a succession of gravel surfacing treatments that represented increasing intensities of BMP implementation on the reopened stream crossing approaches. Rainfall simulation experiments began in February 2012. The unsurfaced approaches were trafficked with a bulldozer immediately before the first series of experiments to mimic newly-disturbed conditions associated with road reopening.

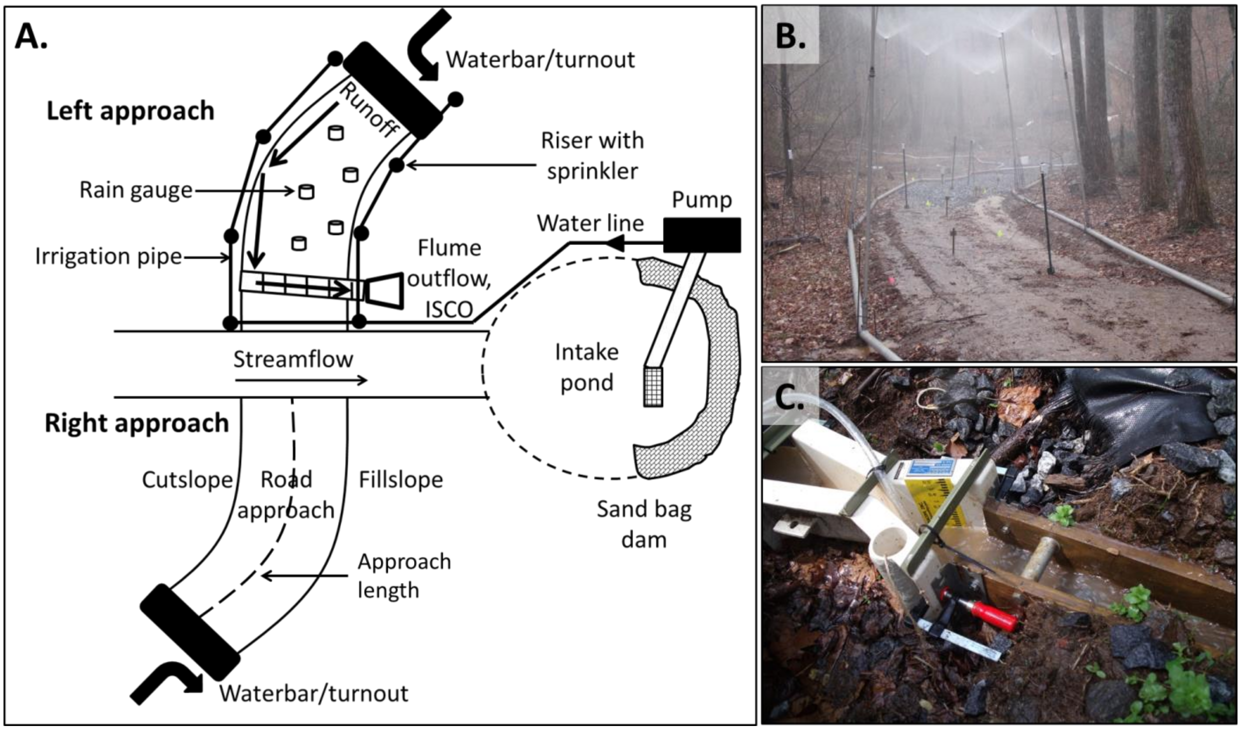

Figure 4.2. Plan view of two idealized stream crossing approaches with rainfall simulator equipment and monitoring instrumentation. Open-top box culverts were located at the bottom of the plot, immediately upslope of the stream (A). Photographs depicting a rainfall simulation experiment on March 12, 2012 (B) and the equipment used to measure surface runoff quantity and quality (C).

Following this treatment ("No Gravel"), the stream crossing approaches had 10-19% surface cover, which consisted of residual leaf litter and other debris. Following the No Gravel treatment rainfall simulations, a dump truck was used to tailgate spread a mixture of size 3, 5, and 7 (ranging from 5.1 to 1.9-cm diameter) granite gravel beginning at the lower plot boundary and continuing uphill for a distance of 9.8 m ("Low Gravel" treatment). Mean gravel depth was approximately 0.08 m and the width of gravel application extended across the width of the road between the outer edges of the running surfaces, which averaged 2.8 m. Gravel was not washed prior to application to the stream crossing approaches as is typical during forest road construction and graveling. The near-stream 9.8-m gravel section was chosen for the Low Gravel treatment because this length approximated half the distance of the shortest approach used in this study. The Low Gravel treatment resulted in different proportions of cover on the running surface area of the study plots primarily because approach lengths were all different, ranging from 19.2 to 41.3 m (Table 4.1). For example, the Low Gravel treatment resulted in 60% surface cover for the shortest approach and 40% surface cover for the longest approach used in this study. Following the Low Gravel rainfall simulations, no additional gravel was applied to the initial 9.8-m-long segment, but gravel was applied to the adjacent (uphill) 9.8-m section of the road approach ("High Gravel"). This treatment effectively doubled the length of the first gravel application and resulted in an overall range of 50 to 99% surface cover on the approach running surfaces. The succession of treatments at each site (No Gravel, Low Gravel, High Gravel) represented increasing intensities of BMP implementation and facilitated the evaluation of a wide range of surface cover on the stream crossing approaches (10-99%) in reducing sediment delivery to streams during simulated rainfall events.

4.3.3 Calculation of gravel cost

The volume of gravel (m^3) applied to each stream crossing approach was calculated as mean gravel depth (0.08 m) x mean running surface width (range = 2.5 to 3.2 m) x length of the graveled section (i.e., initially 9.8 m for the Low Gravel treatment and then 19.6 m for the High Gravel treatment). Gravel volume was converted to a mass (Mg) by assuming a gravel bulk density of 1.6 Mg m^{-3} (O'Neal et al., 2006). Local gravel cost for Critz, VA was \$27.56 Mg^{-1}. Mean gravel costs (i.e., the cost of the rock only) for the Low Gravel and High Gravel treatments were \$91.77 and \$183.54, respectively. Mean gravel cost per meter of approach length was

$9.41 for the Low Gravel and High Gravel treatments. Mean gravel cost per unit area on the running surface was $3.34 m^{-2}. Calculation of gravel costs allowed for an evaluation of the cost-effectiveness of the treatments to reduce sediment delivery to streams during simulated rain events.

4.3.4 Rainfall simulation

A high-pressure pump with a 1.34 x 10^4-W (18-hp) gas-driven engine and a maximum flow rate of 37.9 L s^{-1} was used to pump water for the rainfall simulation experiments from temporary impoundments located approximately 50 m downstream of each stream crossing. Irrigation equipment included a 10.2-cm diameter intake hose and strainer, a 7.6-cm diameter outflow component, and fire hose to connect to 7.6-cm diameter aluminum irrigation pipelines that ran upslope along both sides of the stream crossing approach (Figure 4.2). A 3.0-m-long irrigation pipe, and a pair of 90° angle couplings, joined the two parallel segments of pipe downslope of the lower plot boundary. The pipelines adjacent to the study plots consisted of 6.1-m-long irrigation pipes that were coupled together for the entire length of the approach, including the ungraveled lengths of the approaches. A water-control structure (water bar/turnout) and the open-top box culvert served as the upper and lower plot boundaries, respectively. Risers with sprinklers (Rain Jet 78C, Rain Jet Corp.) at a height of 3.4 m to attain near-terminal velocity of individual water drops (Ward, 1986) were located at 6.4-m intervals along the entire length of the study plots. This irrigation setup was used previously in rainfall simulation experiments to test the effectiveness of agricultural BMPs in reducing soil erosion during rain events (Dillaha et al., 1988) and has a designed rainfall application rate of 50.8 mm h^{-1}. At 50.8 mm h^{-1}, the Rain Jet 78C nozzle provides about 40% of the kinetic energy of natural rainfall (Renard, 1989).

A series of three applied rainfall events, ranging in duration from 10 to 50 min (median = 20 min), was performed for each treatment (No Gravel, Low Gravel, and High Gravel) at each of the six road approaches for a total of 3 x 3 x 6 = 54 experiments (Table 4.2). When stream discharge permitted sufficient water storage in the impoundments, rainfall was applied to mimic short, medium, and long events (usually 10, 20, and > 30 min, respectively) for a given site and treatment combination (e.g., Site 1: No Gravel 1, No Gravel 2, and No Gravel 3). When water availability was limited, (i.e., dry periods in the summer months), rainfall was applied until the impoundments were pumped dry. Applied rainfall events for a given site and treatment combination were typically completed on the same day, with approximately 30 min between each experiment. Rainfall simulation experiments began within 1 to 2 days following treatment application at each site (i.e., first bulldozer trafficking, then Low Gravel application, then High Gravel application). The time in between treatment applications at each site ranged from 2 to 4 weeks. Median intensity of applied rainfall was 49.8 mm h^{-1} and ranged from 20.4 to 73.0 mm h^{-1} (Table 4.2). Event rainfall totals ranged from 6.4 to 41.6 mm, with a median of 16.6 mm. Simulated rainfall events had recurrence intervals of < 1 to 5 years for Critz, Virginia, USA (NWS, 2011).

4.3.5 Sediment collection

Rainfall amount and intensity were measured with six wedge collection-type rain gauges and five automatic tipping-bucket rain gauges (ECRN-50 Low-Resolution Rain Gauge, Decagon Devices, Inc.) that were set to log data at 1-min intervals. Surface runoff discharge was measured with a 2.5-cm x 45.7-cm cutthroat flume (Tracom Fiberglass Products) that was fitted to the outflow end of the open-top box culverts, located at the bottom of the plot (Figure 4.2).

Table 4.2. Running surface cover on the stream crossing approaches and applied rainfall event characteristics for each site and treatment combination.

Site ID	**Surface cover (%)**			**Mean rainfall duration (min)**			**Mean rainfall intensity (mm h^{-1})**		
	No Gravel	Low Gravel	High Gravel	No Gravel	Low Gravel	High Gravel	No Gravel	Low Gravel	High Gravel
1	10	35	90	21	20	35	51	40	48
2	15	40	50	24	30	24	63	24	30
3	17	60	90	19	24	22	64	58	52
4	10	34	50	16	14	14	56	64	64
5	14	47	63	23	15	13	60	48	46
6	19	60	99	33	23	18	56	72	60

The flume stilling well was equipped with a pressure transducer (HOBO U20 Water Level Data Logger, Onset Computer Corporation) to measure water level at 1-min intervals. Manual water level observations were also made using a staff gage that was mounted to the wall of the flume at the inflow end. Water level was converted to discharge (L s^{-1}) using the stage-discharge equation for the specific dimensions of the cutthroat flume. An ISCO automatic stormwater sampler (ISCO 3700 Series, Teledyne ISCO) was programmed to collect 500-mL stormwater runoff samples at 2 to 5 min intervals. Stormwater runoff samples were analyzed in the laboratory for TSS concentration by way of vacuum filtration of a known amount of sample volume and mass of oven-dried sediment trapped by the filters (Eaton et al., 2005). During rainfall simulation, cheesecloth was stretched over the intake strainer to filter suspended solids from the source water to limit carryover; however, applied rainfall samples were collected during each experiment for laboratory analysis of TSS concentration and this was subtracted from the TSS concentration of road surface runoff. Event-based suspended-sediment loads were calculated by multiplying TSS concentration by surface runoff volume in 1-min intervals and then summing the 1-min sediment loads for the entire event duration.

Event hydrographs were used to identify the time intervals corresponding to nearly constant runoff conditions (i.e., soil was partially saturated) and during these times, field hydraulic conductivity of the road surface, K_s, was estimated as rainfall rate minus runoff rate (Reid and Dunne, 1984; Figure 4.3). The median K_s value was used to approximate road surface K_s.

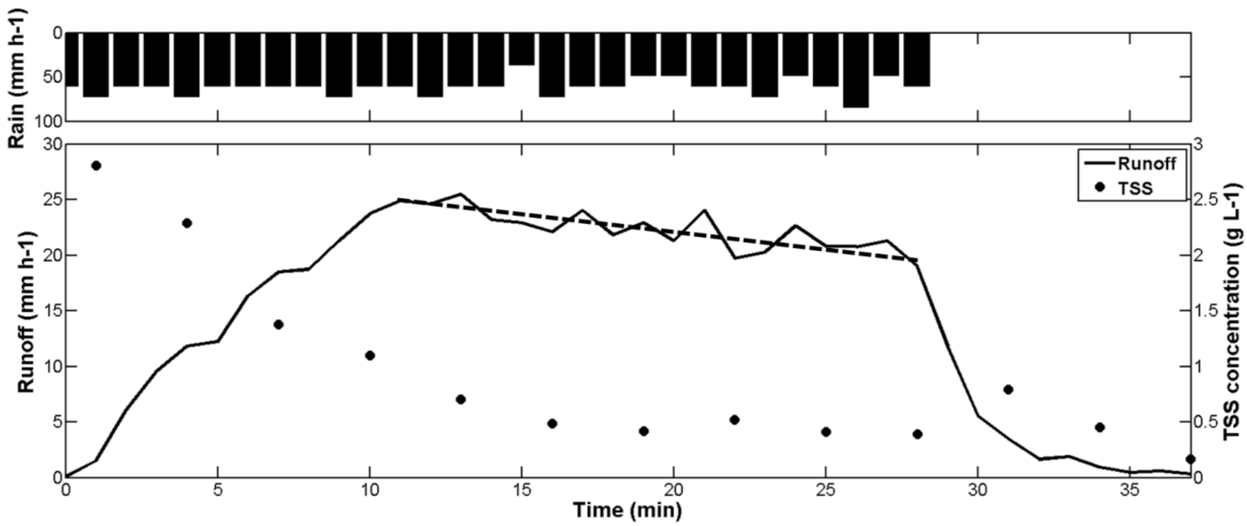

Figure 4.3. Event hydrograph for Site 3, depicting the first of three rainfall simulation experiments for the High Gravel treatment. Field hydraulic conductivity, Ks (mm hr^{-1}), was estimated as the median value of rainfall rate minus runoff rate during conditions approximating steady-state runoff (i.e., from 11 to 28 min in Figure 3, as indicated by the dashed line). These time periods represented partially saturated soil conditions. Notice that runoff decreased due to decreasing rainfall intensity from the middle to the end of the experiment.

Event sediment yield was quantified by dividing event TSS load by the running surface area of the stream crossing approach, which accounted for differences in approach length (i.e., the approach running surface widths did not differ substantially (Table 4.1)). Also, to account for variability in applied rainfall amounts and intensities (Table 4.2), event sediment yield was standardized per unit rainfall (mg m^{-2} mm^{-1}) according to the methods of Sheridan and Noske (2007) and Sheridan et al. (2008):

$TSS_{yield\ (standardized)} = TSS_{yield} / Rain_{total}$ Equation 4.1

where TSS_{yield} is event TSS transport per unit area (mg m^{-2}) and $Rain_{total}$ is total event rainfall (mm).

4.3.6 Statistical analysis

The dependent variables included TSS concentration of road surface runoff, sediment yield per unit rainfall, and K_s. TSS concentration and sediment yield per unit rainfall were natural-log transformed. Each dependent variable was analyzed with a full-factorial design using JMP 10 software (SAS Institute Inc., 2012), which included all possible interactions among the following fixed effects: Treatment (No Gravel, Low Gravel, High Gravel), Rainfall Simulation Group (1, 2, 3), and Slope Group (< 10% slope, > 10% slope); as well as the random effect of Site ID (road approaches 1-6) nested within Slope Group. Treatment was included to account for the effect of gravel surface cover on road surface runoff and suspended-sediment delivery. Rainfall Simulation Group was included to account for decreases in sediment due to the chronological order in which experiments were completed at each stream crossing approach (i.e., three for the No Gravel treatment, followed by three for Low Gravel, and lastly three for High Gravel). Rainfall Simulation Group 1 included the first in a series of three rainfall simulations for each treatment, while Rainfall Simulation Groups 2 and 3 included the second and third rainfall simulations within a given treatment, respectively. The six stream crossing approaches were categorized by slope, with Sites 1, 2, and 6 (slope = 9.4, 4.7, and 6.4%, respectively) in the < 10% slope group and Sites 3, 4, and 5 (slope = 13.7, 15.4, 16.1%, respectively) in the > 10% slope group. Virginia's technical manual for forestry BMPs suggests that road grades be kept between 2 and 10%, whenever possible (VDOF, 2011).

The Site ID variable takes into account that observations within one study plot will be correlated with each other as opposed to observations from different study plots. It is used as a random effect because the study plots represent a subset of all possible forest road-stream crossing approaches. Model fitting involved the retention in the final model of all significant fixed effects (alpha = 0.05), as well as the random effect of Site ID nested within Slope Group (denoted as "Site ID[Slope Group]"). Separation of least squares means was performed with Tukey's HSD in JMP 10 software (SAS Institute Inc., 2012). Model residuals were analyzed for normality and homogeneity of variance with Quantile-Quantile plots in JMP 10.

4.4 Results and Discussion

4.4.1 TSS concentration

Overall, gravel application reduced TSS concentrations of road surface runoff (Table 4.3). TSS concentration for the No Gravel treatment experiments was 2.6 and 3.5 times greater than that of the Low Gravel and High Gravel treatments, respectively (Figure 4.4). The effect of Treatment on TSS concentration was significant ($p < 0.0001$; Table 4.4) and separation of the least squares means indicated that all groups were significantly different (No Gravel > Low Gravel > High Gravel). TSS concentration was greatest during the first rainfall simulation experiment within the No Gravel treatment, while subsequent No Gravel treatment experiments were similar (Figure 4.5). This finding indicates that the supply of loose sediment was approaching depletion after the first No Gravel treatment rainfall simulation experiment. The treatment effect of gravel application (i.e., increased road surface cover) is evidenced by further declines in TSS concentrations of road surface runoff with successive gravel treatments (Figure 4.5).

Renewed sediment sources associated with the application of the gravel treatments included truck traffic on the stream crossing approaches and dust associated with the unwashed gravel. Consequently, TSS concentrations in road surface runoff were also greatest for the first rainfall simulation experiments within the Low Gravel and High Gravel treatments, while subsequent experiments within each treatment were similar (Figure 4.5). The effect of Simulation Group on TSS concentration was significant ($p < 0.0001$; Table 4.4) and separation of the least squares means indicated that Simulation Group 1 > Simulation Groups 2 and 3 (2 and 3 are not different). Least squares means separation for Site ID nested within Slope Group indicated that, in general, higher sediment concentrations were associated with steeper slopes. For example, the ranking of sites from greatest to least in terms of TSS concentration was as follows: Site 4 (15.4% slope), Site 1 (9.4% slope), Site 5 (16.1% slope), Site 3 (13.7% slope), Site 2 (4.7% slope), Site 6 (6.4% slope). Overall, TSS concentration for Site 4 was greater than that of Sites 2 and 6, but not significantly different than Sites 1, 3, and 5.

Table 4.3. Overall ranges of running surface cover on the stream crossing approaches and median TSS concentrations of road surface runoff by treatment (N = 18 rainfall simulation experiments per treatment).

Treatment	**Surface cover (%)**	**Grand median TSS conc. ($g\ L^{-1}$)**	**Range of median TSS conc. per event ($g\ L^{-1}$)**
No Gravel	10 - 19	2.84	0.57 - 7.42
Low Gravel	34 - 60	1.10	0.39 - 3.16
High Gravel	50 - 99	0.82	0.15 - 1.24

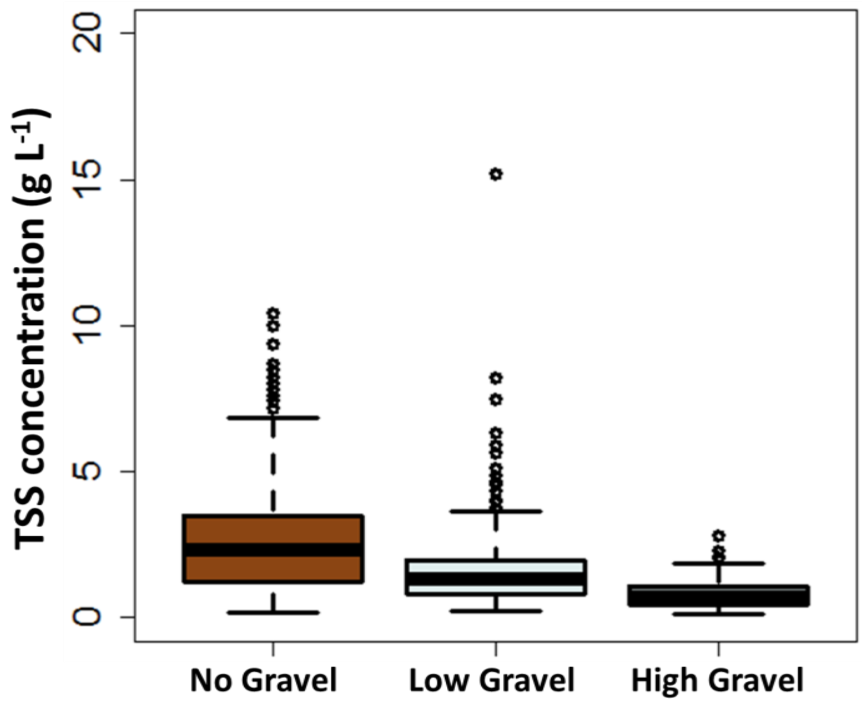

Figure 4.4. Box and whisker plots showing the 5th, 25th, 50th, 75th, and 95th percentiles of all samples of TSS concentrations of road surface runoff (g L^{-1}), aggregated by treatment type (No Gravel, Low Gravel, and High Gravel). N = 228, 222, and 231 for the No Gravel, Low Gravel, and High Gravel treatments, respectively.

Table 4.4. Fixed effect tests for the analysis of TSS concentration of road surface runoff, hydraulic conductivity (K_s), and sediment yield per unit rainfall.

Dependent variable	**Fixed effects**	**DF**	**DF Denominator**	**F Ratio**	**Prob > F**	**Model adjusted R-squared**
TSS	Treatment	2	44.0	57.2	< 0.0001	0.80
	Simulation Group	2	44.0	14.0	< 0.0001	
K_s	Slope Group	1	3.7	30.0	0.01	0.34
	Treatment	2	45.9	4.5	0.02	
Sediment yield	Treatment	2	48.1	33.0	< 0.0001	0.59
	Simulation Group	2	48.1	3.9	0.03	

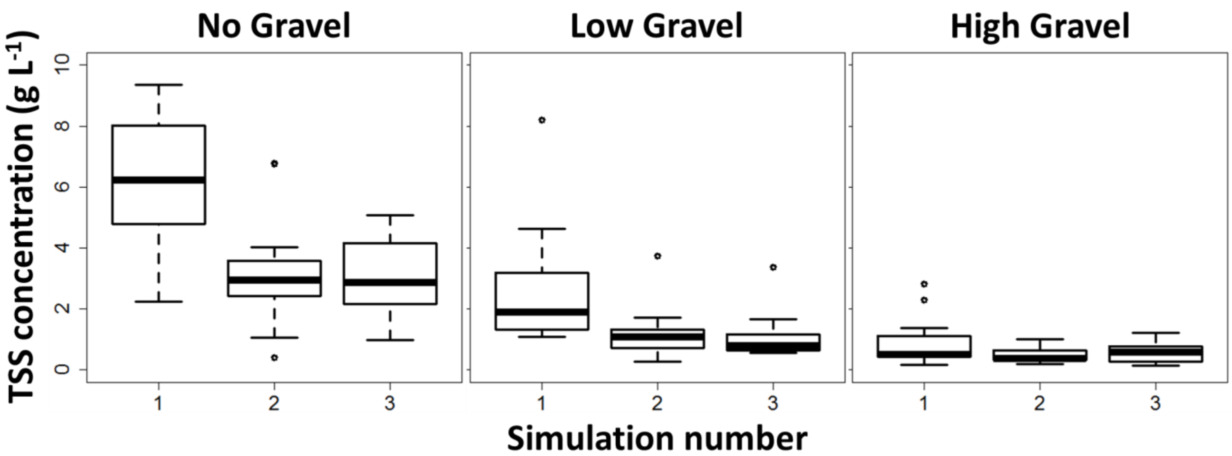

Figure 4.5. Box and whisker plots showing the 5th, 25th, 50th, 75th, and 95th percentiles of TSS concentrations of road surface runoff (g L-1) for successive rainfall simulation experiments within a given treatment. Data are shown for Site 3, where the Low Gravel and High Gravel treatments resulted in 60% and 90% cover, respectively, on the road approach running surface.

Suspended-sediment concentrations of road surface runoff for unsurfaced roads that have an abundant supply of loose sediment can reach 100 g L^{-1} at the onset of high-intensity rainfall (Ziegler et al., 2000a). In our study, maximum TSS concentrations were commonly greater than 8 g L^{-1} and were observed at the onset of applied rainfall for the experiments that were conducted immediately following bulldozer trafficking over the stream crossing approaches (i.e., the first No Gravel treatment rainfall simulation in a series of three). Such high concentrations of TSS in road surface runoff at the road-stream interface pose important issues for water quality protection (McCaffery et al., 2007; Pepino et al., 2012).

However, suspended sediment has been shown to decline rapidly over the course of a runoff-producing event due to a diminishing supply of loose sediment available for downslope transport (Croke et al., 1999a; Ziegler et al., 2000a, 2000b, and 2001; Foltz et al. 2009). Sheridan et al. (2008) reviewed the literature regarding forest road rainfall simulation experiments and found that net sediment concentration of surface runoff ranged from 1.0-18.9 g

L^{-1} for graveled roads (from Selkirk and Riley, 1996; Costantini et al., 1999; Croke et al., 1999b; Riley et al., 1999) and from 7.5-18.0 g L^{-1} for unsurfaced roads (from Croke et al., 1999a; Ziegler et al., 2001). In our study, median TSS concentration of road surface runoff ranged from 0.57 to 7.42 g L^{-1} and from 0.15 to 1.24 g L^{-1}, respectively, for the No Gravel and High Gravel treatment rainfall simulations (Table 4.3). The higher values are associated with the first rainfall simulation experiment following a disturbance event (i.e., bulldozer trafficking or tailgate spreading of gravel from a dump truck), while the lower values are associated with the final (third) rainfall simulation experiment within a given treatment.

4.4.2 Field hydraulic conductivity, Ks

Runoff coefficients by treatment, calculated as event runoff (mm) / event rainfall (mm), ranged from 0.11 to 0.84 (No Gravel), 0.02 to 0.58 (Low Gravel), and 0.07 to 0.46 (High Gravel). The median runoff coefficients of the No Gravel and Low Gravel treatment rainfall simulation experiments (0.36 and 0.35, respectively) were greater than that of the High Gravel treatment rainfall simulation experiments (0.24) by a factor of 1.5. Increases in hydraulic conductivity (mm h^{-1}) were associated with increasing gravel surface cover and ranged from 7.2 to 41.6 (No Gravel), 11.9 to 46.3 (Low Gravel), and 16.0 to 58.6 mm h^{-1} (High Gravel).
Luce (1994) estimated hydraulic conductivities from plots on freshly-bladed, native-surfaced forest roads comprising six different soils in Colorado, Idaho, and Montana. Estimated hydraulic conductivities ranged from 5×10^{-5} to 8.82 mm h^{-1}. Very low (near zero) rates of hydraulic conductivity are more representative of active haul roads with frequent traffic. Much greater road surface hydraulic conductivities have been estimated for roads that were abandoned for approximately 30 years (Foltz et al. 2009) and also for larger road segment-scale (> 10-m-long) rainfall simulation experiments (e.g., Ziegler et al., 2000a). Foltz et al. (2009) measured runoff

and suspended-sediment concentrations during rainfall simulation experiments on the tire track and non-tire track components of abandoned and reopened forest roads in northern Idaho. Saturated hydraulic conductivity ranged from 7-28 mm h^{-1} for the abandoned ("brushed-in") road and from 13-21 mm h^{-1} for the recently reopened road. Overall, infiltration was greatest in the non-track portions of the abandoned road that had been left to regrow.

Study findings from Luce (1994), Ziegler et al. (2000a), and Foltz et al. (2009) indicate that infiltration rates of unpaved forest roads can be highly variable and dependent on factors such as traffic (i.e., type, weight, and frequency), time since disturbance, soil type, and climate, as well as the scale of rainfall simulation experiments. Frequent traffic leads to highly-compacted road surfaces, and consequently, lower infiltration rates. Following road abandonment and traffic cessation, cracks and other macropores can form over time as a result of weathering, vegetation re-establishment and burrowing by organisms, resulting in increased infiltration rates. Also, the greater surface areas of large-scale rainfall simulation experimental plots can result in a higher probability of overall infiltration rates being influenced by one or more macropores (i.e. areas of high infiltration). Therefore, it is arguable that large-scale rainfall simulation experiments offer a more representative estimate of hydraulic conductivity values at the road segment scale, while being well-aligned with the spatial scale of forest road management and erosion modeling.

We found relatively high values for hydraulic conductivity, ranging from 7.2-41.6 mm h^{-1}, for abandoned forest road-stream crossing approaches that were recently reopened by bulldozer blading. Earlier work at RHFRRC that estimated road surface infiltration rates with double-ring infiltrometers found that rates ranged between 0.6 and 7.2 mm hr^{-1} (Kadak et al., 2000, unpublished data). It is hypothesized that the relatively large scale of our rainfall simulation experiments (61-116 m^2) had a greater likelihood of interaction with macropores at the road segment scale, which resulted in the greater conductivity rates. In addition, our estimates of hydraulic conductivity were representative partially saturated conditions, as opposed to completely saturated conditions. Our estimates of K_s would likely be lower had each applied rainfall experiment reached steady-state runoff conditions and soil was sufficiently saturated. This is because infiltration rates decrease over time as wetting front potential in the soil decreases. It is also important to note that the stream crossing approaches were abandoned prior to this study and that following reopening by bulldozer blading, traffic was limited to light-vehicle use to perform the rainfall simulation experiments (1 to 2 passes per week), as well as two passes by a dump truck to spread gravel on the approaches.

We found that median K_s for the High Gravel treatment (36.7 mm h^{-1}) was greater than that of the Low Gravel (33.9 mm h^{-1}) and No Gravel (33.1 mm h^{-1}) treatments. The Slope Group effect was significant ($p = 0.01$), with higher hydraulic conductivities occurring on steeper slopes, suggesting a potential interaction with soil texture (Table 4.1). The Treatment effect was significant ($p = 0.02$) and separation of the least squares means indicated that hydraulic conductivity was greatest for the High Gravel rainfall simulation experiments (Table 4.4). This finding suggests that infiltration rates tended to be higher with the highest gravel surface cover. Increased surface roughness likely acted to slow runoff velocities and allowed greater infiltration. Luce (2002) suggests that we may not be able to ameliorate road surface infiltration for active roads because they are impervious by design. However, we found that the most gravel surface cover was associated with increased infiltration.

More importantly, gravel surface cover helps to reduce TSS concentration of road surface runoff at stream crossings, and thus sediment delivery to streams.

4.4.3 Standardized sediment yield

Median TSS load per rainfall simulation experiment was 1.73, 0.38, and 0.17 kg, respectively, for the No Gravel, Low Gravel, and High Gravel treatment rainfall simulation experiments (Table 4.5). Sediment yield per unit rainfall (mg m^{-2} mm^{-1}) ranged from 185 to 6470 (No Gravel treatment), 30 to 1130 (Low Gravel treatment), and 58 to 268 (High Gravel treatment).

Table 4.5. Summaries of event TSS load (kg), runoff amount (mm), rainfall amount (mm), and runoff coefficient (runoff / rainfall) by treatment.

Treatment	**TSS load (kg)**			**Runoff (mm)**			**Rainfall (mm)**			**Runoff coefficient**		
	Min	*Mdn**	*Max*	*Min*	*Mdn*	*Max*	*Min*	*Mdn*	*Max*	*Min*	*Mdn*	*Max*
No Gravel	0.29	1.73	6.41	0.7	7.4	16.6	5.6	20.2	42.0	0.11	0.36	0.84
Low Gravel	0.04	0.38	1.86	0.3	5.5	21.5	8.8	14.8	36.8	0.02	0.35	0.58
High Gravel	0.04	0.17	0.57	0.9	3.5	9.6	4.2	15.0	29.6	0.07	0.24	0.46

*Mdn = median

Sheridan and Noske (2007) quantified sediment yield per unit rainfall for different forest road types in the State Forest in the Central Highlands area of the Great Dividing Range, Victoria, Australia and estimated a range of 216 to 5373 mg m^{-2} mm^{-1}. The lower value represented a minimum-traffic, high-quality gravel road, while the higher value represented an unsurfaced road, with erodible soil and moderate light-vehicle traffic.

In our study, following road reopening by bulldozer blading, traffic included light-vehicle use to complete the rainfall simulation experiments (i.e., 1 to 2 passes per week), as well as two passes by a dump truck to spread gravel on the approaches.

Sheridan et al. (2008) used rainfall simulation to quantify interrill erodibility index values of unsealed forest roads for comparison with observed forest road erodibility index values that were quantified from 1 yr of detailed in-situ erosion monitoring. Sediment yield per unit rainfall (mg m^{-2} mm^{-1}) estimated from 1 yr of monitoring (natural rainfall) ranged from 839 to 8671 for the bare roads and from 312 to 2687 for the gravel roads. Sediment yield per unit rainfall (mg m^{-2} mm^{-1}) estimated from the rainfall simulation experiments ranged from 3650 to 7024 for the gravel roads and from 3462 to 28,426 for the bare roads (Sheridan et al., 2008). Our estimates of road-surface erodibility index values are within the range of values estimated by Sheridan and Noske (2007) and Sheridan et al. (2008) for natural rainfall conditions, but considerably less than those estimated from applied rainfall experiments by Sheridan et al. (2008). Our rainfall experiments were more representative of natural rainfall conditions in that rainfall intensity was variable, and on average, rainfall intensity was half that of experiments by Sheridan et al. (2008), where applied rainfall intensity was constant (100 mm hr^{-1}) for 30 minutes. Dynamic erodibility of road surface soil (Ziegler et al., 2000b; Foltz et al., 2008a) is a function of rainfall intensity and soil type, as well as the cumulative rainfall amount since a disturbance event, such as road blading. Therefore, it may be difficult or impossible to assign a single soil erodibility parameter value with confidence to a specific road segment, which is common practice in parameterizing soil erosion models, even if the parameter value is based on well-controlled field experiments (Brazier et al., 2000).

We found that median sediment yield per unit rainfall (mg m^{-2} mm^{-1}) by treatment was 788 (No Gravel), 389 (Low Gravel), and 161 (High Gravel). Median sediment yield per unit rainfall for the No Gravel treatment rainfall simulation experiments was greater than that of the Low Gravel and High Gravel treatments by factors of 2.0 and 4.9, respectively.
Results for standardized sediment yield were similar to TSS concentration in that Treatment ($p < 0.0001$) and Simulation Group ($p = 0.03$) were both significant model effects (Table 4.4). Separation of the least squares means by treatment indicated that No Gravel > Low Gravel > High Gravel. Simulation Group 1 was greater than Simulation Group 3, but not different than Simulation Group 2. There were no differences in standardized sediment yield among slope groups.

4.4.4 Cost-effectiveness of gravel treatments

The mean cost of the Low Gravel treatment was $91.77 or $3.34 m^{-2}. When applied to stream approaches with different lengths (19.2 to 41.3 m), the Low Gravel treatment (i.e., gravel application beginning near the stream and continuing uphill for a length of 9.8 m) resulted in 34-60% cover on the road approach running surfaces. Increasing surface cover led to decreases in sediment yield (Figure 4.6). Overall, the Low Gravel treatment reduced sediment yield by a factor of 2.0 over the No Gravel treatment. The High Gravel treatment doubled costs ($183.54), and resulted in 50-99% coverage of the road approach running surfaces. Overall, the High Gravel treatment reduced sediment yield by a factor of 4.9 over the No Gravel treatment. However, increasing gravel surface cover did not always result in sediment yield reductions, indicating that BMP effectiveness is site specific. For example, rutting from the Low Gravel treatment application at Site 2 resulted in greater sediment yields than the No Gravel Treatment

(Table 4.6). In four of the six study sites, however, the succession of gravel treatments led to successive decreases in sediment yield (Table 4.6).

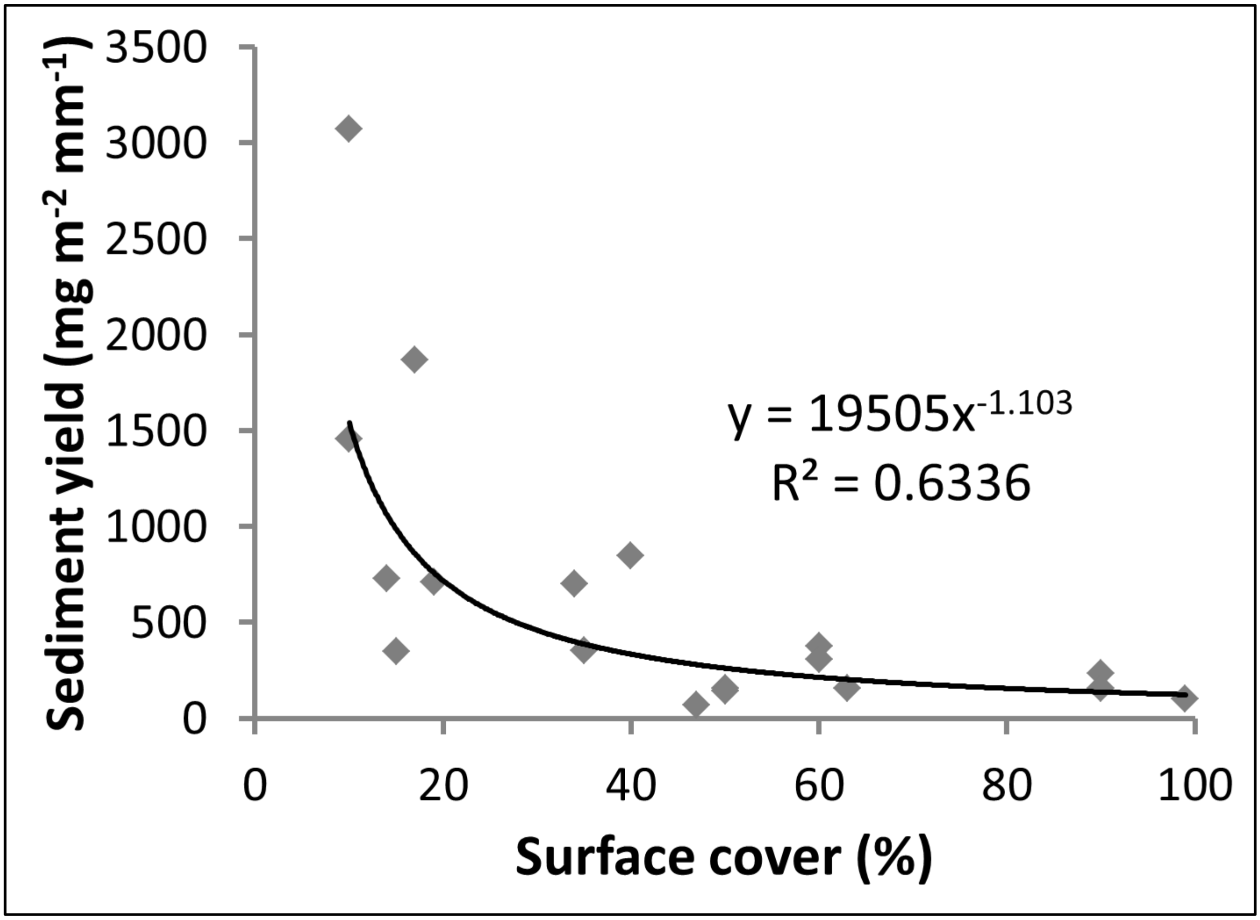

Figure 4.6. Mean sediment yield per unit rainfall (mg m^{-2} mm^{-1}) for each site and treatment combination (6 sites x 3 treatments = 18 observations) versus running surface cover.

The High Gravel treatment covered 90-99% of the running surface area for the shortest approaches (Sites 1, 3, and 6; Table 2) and sediment yields were reduced to less than 250 mg m^{-2} mm^{-1} (Figure 4.6). TSS concentrations were reduced to less than 1 g L^{-1} for sites with more than 90% cover. One option for reducing the costs associated with graveling stream crossing approaches is to use water-control structures (water bars/turnouts, cross drains, broad-based or rolling dips) to reduce the length of the road segment draining directly to the stream. Shorter approaches (e.g. 15.2 m) require less gravel and less money to surface.

Table 4.6. Costs associated with the gravel treatments and differences in sediment yield in comparison to the No Gravel treatment.

Site	Sediment yield ($mg\ m^{-2}\ mm^{-1}$)			Gravel cost ($)		Sediment yield difference from No Gravel (%)	
	No Gravel	Low Gravel	High Gravel	Low Gravel	High Gravel	Low Gravel	High Gravel
1	1458	351	233	100.96	201.92	-76	-84
2	350	850	158	81.42	162.84	143	-55
3	1867	374	158	104.22	208.44	-80	-92
4	3072	699	144	81.42	162.84	-77	-95
5	728	71	156	94.45	188.90	-90	-79
6	709	309	100	84.68	169.36	-56	-86

In addition, shorter approaches manage surface runoff in smaller quantities. In our case, the main cost associated with graveling the stream crossing approaches was that of the gravel itself. Gravel was procured and tailgate spread with a dump truck that is maintained and operated by RHFRRC staff. The costs associated with graveling as presented in this research would be increased by additional factors, such as whether or not a dump truck is owned or must be rented, operator costs, as well as the distance to the nearest gravel retailer. In this study, stream crossing approach lengths ranged from 19.2-41.3 m. The lengthy approaches are representative of legacy forest roads, indicating that upon reopening, additional measures to reduce the drainage length and maximize road approach surface cover may be necessary to protect water quality.

The Virginia Department of Forestry BMP manual (VDOF, 2011) recommends that the entire length and width of active stream crossing approaches be covered with gravel, mulch, or other suitable material, or for a minimum of 15.2 m on either side of the crossing. In many cases, low-cost water-control structures such as water bars, turnouts, or rolling dips could be installed on reopened stream crossing approaches to limit the drainage length to 15.2 m. The total cost of gravel to cover this section of road would then be 15.2 m x $9.41 per meter of road length = $143. Based on our study findings, completely-graveled stream crossing approaches with minimal traffic had TSS concentrations well below 1 g L^{-1} during rainfall events. This demonstrates that judicious BMP implementation can be used to reduce sediment delivery to streams in situations where original road conditions at the road-stream interface were less than ideal.

4.5 Conclusions

The direct contribution of surface runoff and sediment delivery to streams at forest road-stream crossings necessitates the careful management of stormwater runoff to reduce negative impacts to water quality and aquatic habitat. In the case of reopening legacy road-stream crossing approaches, perhaps even greater efforts are necessary to protect water quality, especially if the approaches have minimal surface cover, inadequate water control, or steep grade. Quantification of road segment-scale surface hydrologic processes and sediment delivery during storm events, as influenced by factors such as road type and traffic frequency, are crucial to facilitate the development of cost-effective BMPs to control sediment transport where the probability of water quality impacts are highest.

This research used rainfall simulation experiments to quantify road segment-scale surface runoff and suspended-sediment delivery for reopened stream crossing approaches with different levels of gravel surface cover. The cost-effectiveness of gravel application was also quantified in terms of reductions in suspended-sediment delivery. Study findings suggest that newly-reopened legacy road-stream crossing approaches can have much higher rates of hydraulic conductivity than highly-trafficked crossings, but that reopened (and unsurfaced) stream crossing approaches can represent significant sources of direct stormwater runoff and sediment delivery to streams. This research demonstrates that judicious and low-cost BMP implementation can be used to ameliorate common water quality problems associated with reopening legacy road-stream crossing approaches.

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5.0 CAN WEPP BE UTILIZED TO ESTIMATE BMP EFFECTIVENESS FROM FOREST ROADS?

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5.1 Abstract

Tools are needed to evaluate forestry best management practice (BMP) efficacy in reducing sediment delivery to streams, and to select appropriate BMPs for different road conditions. The Water Erosion Prediction Project (WEPP) model has been used extensively to evaluate BMP effectiveness and is currently recommended by many forest-management organizations. WEPP is physically based and was designed to incorporate field observations and site-level information for predicting sediment yield and runoff. In this study, we used WEPP to predict event-based sediment yield and runoff for rainfall simulation experiments on six reopened stream-crossing approaches that underwent different intensities of BMP implementation (i.e., different proportions of gravel on the road surface above the stream crossing). WEPP was calibrated for each of these stream-crossing approaches for three different BMP intensities using a Markov Chain Monte Carlo approach to explore parameter uncertainty/sensitivity, as well as prediction performance. The objectives of the study were to determine overall prediction performance of WEPP and its ability to distinguish between different BMP intensities. WEPP underpredicted event-based runoff. WEPP predictions of

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sediment yield showed clear differences among the different road surface treatments used in this study, but prediction intervals were wide, reflecting substantial parameter and prediction uncertainty. Posterior distribution analysis for rill erodibility, interrill erodibility, and critical shear indicated that we cannot recommend parameter ranges specific to different surface treatments. Study findings suggest that WEPP's utility in estimating BMP effectiveness is limited to predicting relative differences in sediment yield among vastly different surface treatments (e.g., native surfaced vs. completely graveled roads).

5.2 Introduction

Forest roads at stream crossings are a major source of sediment delivery to streams (Lane and Sheridan, 2002; Harris *et al.*, 2008; Anderson and Lockaby, 2011b). The management of channelized runoff from roads has been the focus of recent legislative debates in the U.S. regarding the protection of aquatic ecosystems in forests. For the past 40 years, forestry best management practices (BMPs) have been used to manage runoff and sediment delivery from roads, but environmental organizations such as the Northwest Environmental Defense Center have fought for legislation to achieve the goal of water-quality protection with National Pollution Discharge Elimination (NPDES) permits (Boston, 2012). Currently, stormwater runoff from forest roads is still managed as a nonpoint source in the U.S. However, the debates prompted the U.S. Environmental Protection Agency to request that state forestry organizations evaluate the effectiveness of existing BMPs to reduce sediment delivery from major sources (i.e. roads and stream crossings) and provide guidance for enhanced BMPs.

Field studies provide valuable information about the effectiveness of different intensities of BMP implementations to reduce erosion and sediment delivery (Appelboom *et al.*, 2002; Turton *et al.*, 2009; Anderson and Lockaby, 2011a). However, field experiments and monitoring of BMPs on roads is often an impractical option for managers due to cost, time, and heterogeneous road conditions (i.e., different climates, soils, slopes, road types, and traffic characteristics). This has led to an increased interest in using models for evaluating BMP effectiveness. In particular, the Water Erosion Prediction Project (WEPP) model has been used extensively as a tool for this purpose (Sawyers *et al.*, 2012; Wade *et al.*, 2012; Elliot, 2013) and is currently recommended by many forest management organizations.

WEPP is physically-based and was designed to incorporate field observations and site-level information for predicting sediment yield and runoff. Previous studies indicate that WEPP can be a useful tool for estimating soil erosion from forest roads, where overland flow is the dominant hydrologic process (Laflen *et al.*, 2004; Croke and Nethery, 2006; Fu *et al.*, 2010). However, model performance has not been evaluated for forests roads at stream crossings and for a wide range of approach characteristics, BMP implementations, and rainfall conditions.

In addition, methodologies for evaluating soil erosion model performance have typically included model calibration and validation procedures without explicitly accounting for sources of model prediction uncertainty, which can include model input parameter values and variability in observed data (Brazier *et al.*, 2000; Beven, 2008). Commonly, one or more objective functions (e.g., sum of squared errors, Nash-Sutcliffe efficiency) are used to identify the most acceptable model based on prediction performance for runoff and sediment yield. However, it is possible that multiple models, with unique combinations of model input parameter values, can generate equally acceptable model predictions. In the case of physically-based models with many model input parameters, it can be very difficult or impossible to identify the most acceptable model due to complex interactions among the model input parameters. Further, it is an uncommon occurrence for the modeler to know the appropriate values for each of the model input parameters. This is due to spatial heterogeneity of site physical characteristics, which translates to variability in measurements of runoff and sediment yield. Therefore, it appears prudent to embrace the concept of model equifinality, where a number of different models (i.e., unique sets of input parameter values) can produce satisfactory predictions, and identify distinct ranges of model input parameter values that are associated with acceptable model runs (Beven, 2008).

In light of the different types of uncertainty associated with soil erosion predictions (e.g., measurement error, model parameterization), model utility need not be defined solely by prediction accuracy, especially when the level of sediment reduction necessary to maintain or improve aquatic habitat is not often well defined. However, useful models should facilitate the identification of problem road segments for water-quality protection, or better, allow us to rank relative differences in erosion or sediment delivery among different treatments that represent increasing intensities of BMP implementation. In this study, WEPP was used to predict event-based sediment yield and runoff for rainfall simulation experiments on six reopened stream-crossing approaches that each underwent different intensities of BMP implementation (i.e., different proportions of gravel on the road surface above the stream crossing). WEPP was calibrated for each of these stream-crossing approaches for three different BMP intensities using a Markov Chain Monte Carlo (MCMC) approach to explore parameter identifiability and prediction accuracy and uncertainty. The objectives of the study were to determine overall prediction performance of WEPP and its ability to distinguish between different BMP intensities.

This study focused on three research questions in the Piedmont physiographic region of southwestern Virginia, USA: (1) How well does WEPP predict event-based runoff and erosion from forest roads at stream crossings?; (2) Can WEPP simulate changes in soil erodibility as a function of successive rainfall events, as well as the treatment effect of increasing gravel surface cover on the stream-crossing approaches?; and (3) Can distinct ranges of the model input parameters be identified in association with the different road surface treatments?

5.3 Materials and Methods

5.3.1 Study area

Six road approaches associated with three unimproved ford stream crossings on a legacy forest road at the Reynolds Homestead Forest Resources Research Center (RHFRRC), located in Critz, Virginia (Patrick County), USA, were selected for study of event-based surface runoff and sediment yield (Figure 5.1). Topography is characterized by rolling hills, with side slopes generally ranging from 8 to 25% and a mean elevation of approximately 335 m above mean sea level (NRCS, 2013). As is typical of the Piedmont region, old agricultural gullies are common because most of the contemporary forested watersheds were in agriculture during the 1800s (Trimble, 2008). Mean annual rainfall is 1250 mm, with a mean snow contribution of 270 mm to the total precipitation. Mean air temperature ranges from a low of -1.8°C in January to a high of 29.7°C in July (Sawyers *et al.*, 2012). The predominant soil series is Fairview sandy clay loam (fine, kaolinitic, mesic typic Kanhapludults). Soil parent material is residuum from mica schist and mica gneiss. The severe erosion hazard rating for forest roads and trails at RHFRRC (NRCS, 2013) underscores the importance of controlling road grade, water, and surface cover to reduce erosion and sediment delivery.

5.3.2 Field methods

Prior to road reopening, a Sokkia model SET-520 total station was used to measure the length of the stream-crossing approach study plots, as well as approach slope and mean running surface width (Table 5.1). Length was defined as the distance between the nearest water-control structure (i.e., water bar/turnout) and the stream.

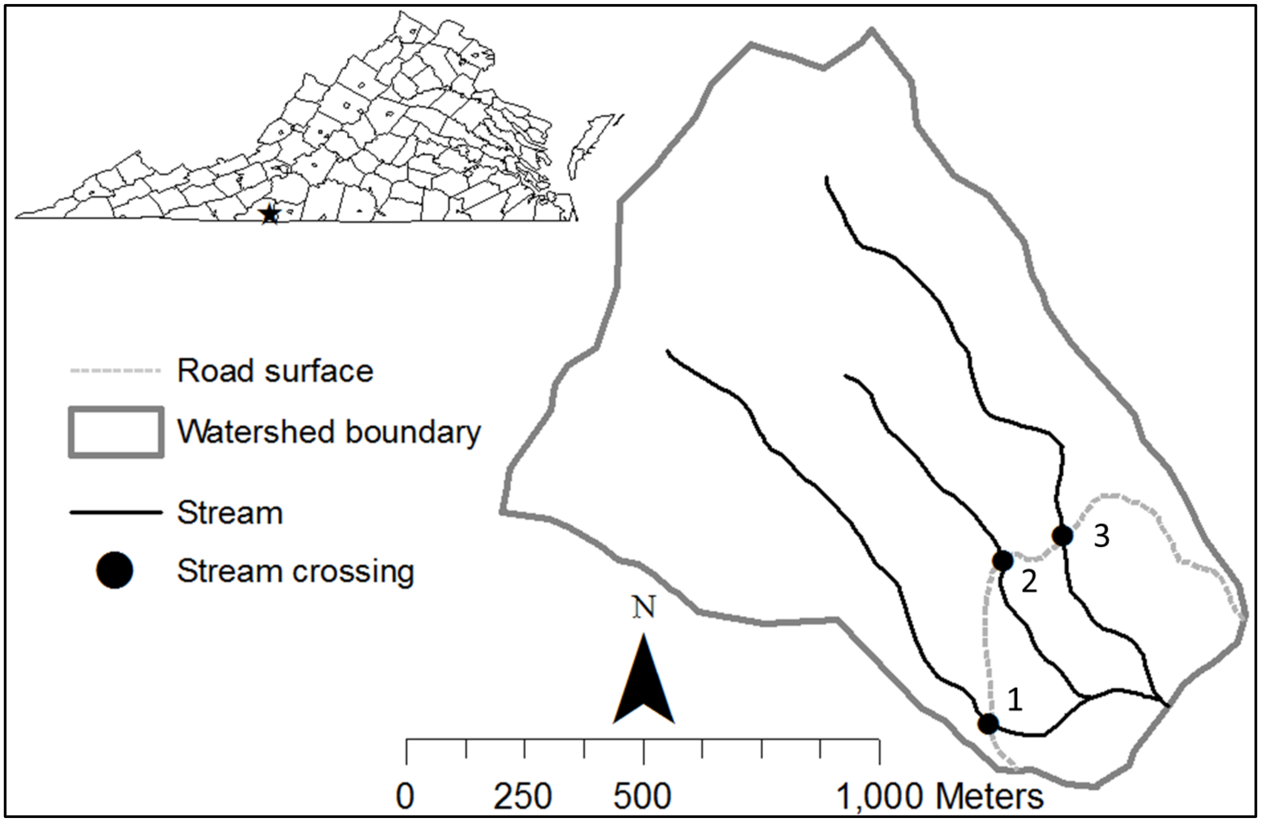

Figure 5.1. Location map of the Reynolds Homestead Forest Resources Research Center in Critz, Virginia (Patrick County), USA, and a schematic showing the road location within the 2nd-order watershed that contains the three unimproved ford stream crossings. This 135-ha watershed drains to the southeast from No Business Mountain and is part of the western headwaters of Mill Creek.

In late-July 2011, six stream-crossing approaches were reopened by bulldozer blading, creating initial conditions of approximately 100% bare soil on the approach running surfaces. The road profiles of each approach study plot were cut-and-fill. Bulldozer blading created a slight cross-slope in the running surface, which directed surface runoff and suspended sediment toward the cutslope bank en route to the bottom of the plot, which ended just before the stream (Figure 5.2). Outboard-edge berm installations were also used to prevent surface runoff from traveling away from the running surface and down the fillslope bank. Open-top box culverts (Trimble and Sartz, 1957) were installed at the lower plot boundary to redirect stormwater runoff away from the stream and toward a flume, where runoff volumes were measured. In October 2011, Kadak (2012, unpublished data) used double-ring infiltrometers to estimate infiltration rates of the reopened stream crossing approaches. Infiltration rates ranged from 0.6 to 7.2 mm hr^{-1}. Bulk density samples (n = 4 to 7 per site) were obtained from the running surface via the soil extraction method (SSSA, 1986).

Table 5.1. Physical characteristics of the running surface component of the road-stream crossing approach study sites at the Reynolds Homestead Forest Resources Research Center, Critz, Virginia (Patrick County), USA.

Site ID	**Approach length (m)**	**Mean width (m)**	**Surface area (m^2)**	**Mean slope (%)**	**Bulk density ($g\ cm^{-3}$)**	**Soil texture***	**Road vertical curvature**	**Running surface profile**
1	25.8	3.1	79.4	9.4	1.61	SCL	S-shaped	Insloped
2	41.3	2.5	102.6	4.7	1.55	SCL	Linear	Insloped
3	19.2	3.2	61.3	13.7	1.51	SCL	Convex	Insloped
4	35.3	2.5	86.8	15.4	1.46	SCL	S-shaped	Insloped
5	39.6	2.9	115.7	16.1	1.37	SCL	S-shaped	Insloped
6	23.5	2.6	61.3	6.4	1.47	CL	Linear	Insloped

*SCL and CL denote sandy clay loam and clay loam soil textures, respectively.

Rainfall simulation experiments were conducted for a succession of gravel surfacing treatments that represented increasing intensities of BMP implementation on the reopened stream-crossing approaches (Figure 5.3). See Brown et al. (2014) for additional details of the experimental design. Rainfall simulation experiments began in February 2012. The unsurfaced approaches were trafficked with a bulldozer immediately before the first series of experiments to mimic newly-disturbed conditions associated with road reopening. Following this treatment (“No Gravel”), the stream crossing approaches had 10-19% surface cover, which consisted of residual leaf litter and other debris. Following the No Gravel treatment rainfall simulations, a dump truck was used to tailgate spread a mixture of size 3, 5, and 7 (ranging from 5.1 to 1.9-cm diameter) granite gravel beginning at the lower plot boundary and continuing uphill for a distance of 9.8 m ("Low Gravel" treatment).

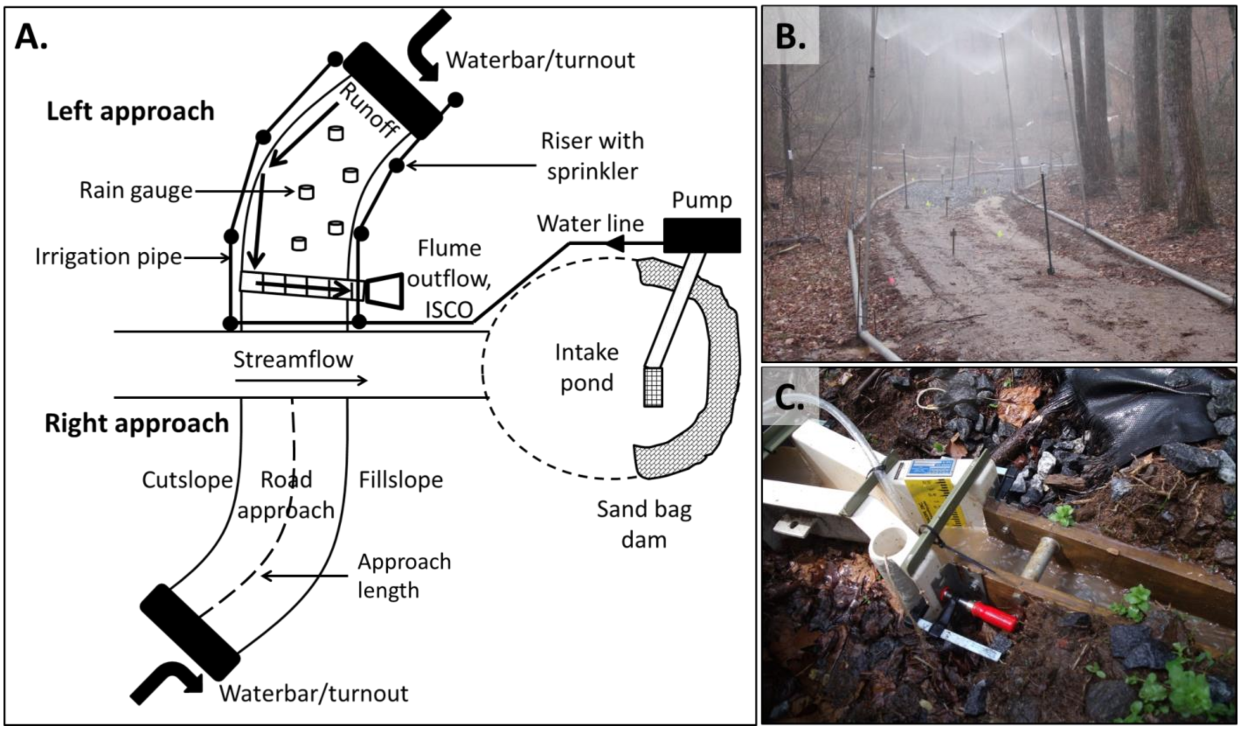

Figure 5.2. Plan view of two idealized stream crossing approaches with rainfall simulator equipment and monitoring instrumentation. Open-top box culverts were located at the bottom of the plot, immediately upslope of the stream (A). Photographs depicting a rainfall simulation experiment on March 12, 2012 (B) and the equipment used to measure surface runoff quantity and quality (C).

Mean gravel depth was approximately 0.08 m and the width of gravel application extended across the width of the road between the outer edges of the running surfaces, which averaged 2.8 m. Gravel was not washed prior to application to the stream crossing approaches as is typical during forest road construction and graveling.

Figure 5.3. Photographs depicting rainfall simulation experiments for the succession of gravel surfacing treatments at Site 5: A.) No Gravel; B.) Low Gravel; C.) High Gravel. Surface cover for the rainfall simulation experiments at Site 5 was 14, 47, and 63%, respectively, for the No Gravel, Low Gravel, and High Gravel treatments.

The near-stream 9.8-m gravel section was chosen for the Low Gravel treatment because this length approximated half the distance of the shortest approach used in this study. The Low Gravel treatment resulted in different proportions of cover on the running surface area of the study plots primarily because approach lengths were all different, ranging from 19.2 to 41.3 m (Table 5.1). For example, the Low Gravel treatment resulted in 60% surface cover for the shortest approach and 40% surface cover for the longest approach used in this study. Following the Low Gravel rainfall simulations, no additional gravel was applied to the initial 9.8-m-long segment, but gravel was applied to the adjacent (uphill) 9.8-m section of the road approach ("High Gravel"). This treatment effectively doubled the length of the first gravel application and resulted in an overall range of 50 to 99% surface cover on the approach running surfaces. The succession of treatments at each site (No Gravel, Low Gravel, High Gravel) represented increasing intensities of BMP implementation and facilitated the evaluation of a wide range of surface cover on the stream crossing approaches (10-99%) in reducing sediment delivery to streams during simulated rainfall events.

5.3.3 Rainfall simulation

A high-pressure pump with a 1.34 x 10^4-W (18-hp) gas-driven engine and a maximum flow rate of 37.9 L s^{-1} was used to pump water for the rainfall simulation experiments from temporary impoundments located approximately 50 m downstream of each stream crossing. Irrigation equipment included a 10.2-cm diameter intake hose and strainer, a 7.6-cm diameter outflow component, and fire hose to connect to 7.6-cm diameter aluminum irrigation pipelines that ran upslope along both sides of the stream crossing approach (Figure 5.2). A 3.0-m-long irrigation pipe, and a pair of 90° angle couplings, joined the two parallel segments of pipe downslope of the lower plot boundary. The pipelines adjacent to the study plots consisted of 6.1-

m-long irrigation pipes that were coupled together for the entire length of the approach, including the ungraveled lengths of the approaches. A water-control structure (water bar/turnout) and the open-top box culvert served as the upper and lower plot boundaries, respectively. Risers with sprinklers (Rain Jet 78C, Rain Jet Corp.) at a height of 3.4 m to attain near-terminal velocity of individual water drops (Ward, 1986) were located at 6.4-m intervals along the entire length of the study plots. This irrigation setup was used previously in rainfall simulation experiments to test the effectiveness of agricultural BMPs in reducing soil erosion during rain events (Dillaha *et al.*, 1988) and has a designed rainfall application rate of 50.8 mm h^{-1}. At 50.8 mm h^{-1}, the Rain Jet 78C nozzle provides about 40% of the kinetic energy of natural rainfall (Renard, 1989).

A series of three to four applied rainfall events, ranging in duration from 10 to 50 min (median = 20 min), was performed for each treatment (No Gravel, Low Gravel, and High Gravel) at each of the six road approaches for a total of 58 experiments (Table 5.2). When stream discharge permitted sufficient water storage in the impoundments, rainfall was applied to mimic short, medium, and long events (usually 10, 20, and > 30 min, respectively) for a given site and treatment combination (e.g., Site 1: No Gravel 1, No Gravel 2, and No Gravel 3). When water availability was limited, (i.e., dry periods in the summer months), rainfall was applied until the impoundments were pumped dry.

Applied rainfall events for a given site and treatment combination were typically completed on the same day, with approximately 30 min between each experiment. Rainfall simulation experiments began within 1 to 2 days following treatment application at each site (i.e., first bulldozer trafficking, then Low Gravel application, then High Gravel application). The time in between treatment applications at each site ranged from 2 to 4 weeks. Prior to each rainfall simulation experiment, volumetric soil water content (N = 4 to 21) was monitored with a soil-water sensor (Hydrosense II CS659, Campbell Scientific, Inc.) to a soil depth of 12 cm on the running surface component of the stream approaches. Median intensity of applied rainfall was 49.8 mm h^{-1} and ranged from 20.4 to 73.0 mm h^{-1} (Table 5.2). Event rainfall totals ranged from 6.4 to 41.6 mm, with a median of 16.6 mm. Simulated rainfall events had recurrence intervals of < 1 to 5 years for Critz, Virginia, USA (NWS, 2011).

Table 5.2. Running surface cover on the stream-crossing approaches and applied rainfall event characteristics for each site and treatment combination.

Site ID	**Surface cover (%)**			**Mean rainfall duration (min)**			**Mean rainfall intensity (mm h^{-1})**		
	No Gravel	Low Gravel	High Gravel	No Gravel	Low Gravel	High Gravel	No Gravel	Low Gravel	High Gravel
1	10	35	90	21	20	35	51	40	48
2	15	40	50	24	30	24	63	24	30
3	17	60	90	19	24	22	64	58	52
4	10	34	50	16	14	14	56	64	64
5	14	47	63	23	15	13	60	48	46
6	19	60	99	33	23	18	56	72	60

5.3.4 Sediment collection

Rainfall amount and intensity were measured with six wedge collection-type rain gauges and five automatic tipping-bucket rain gauges (ECRN-50 Low-Resolution Rain Gauge, Decagon Devices, Inc.) that were set to log data at 1-min intervals. Surface runoff discharge was measured with a 2.5-cm x 45.7-cm cutthroat flume (Tracom Fiberglass Products) that was fitted to the outflow end of the open-top box culverts, located at the bottom of the plot (Figure 5.2). The flume stilling well was equipped with a pressure transducer (HOBO U20 Water Level Data Logger, Onset Computer Corporation) to measure water level at 1-min intervals. Manual water level observations were also made using a staff gage that was mounted to the wall of the flume at the inflow end. Water level was converted to discharge (L s^{-1}) using the stage-discharge equation for the specific dimensions of the cutthroat flume. An ISCO automatic stormwater sampler (ISCO 3700 Series, Teledyne ISCO) was programmed to collect 500-mL stormwater runoff samples at 2 to 5 min intervals.

Stormwater runoff samples were analyzed in the laboratory for TSS concentration by way of vacuum filtration of a known amount of sample volume and mass of oven-dried sediment trapped by the filters (Eaton *et al.*, 2005). During rainfall simulation, cheesecloth was stretched over the intake strainer to filter suspended solids from the source water to limit carryover; however, applied rainfall samples were collected during each experiment for laboratory analysis of TSS concentration and this was subtracted from the TSS concentration of road surface runoff. Event-based suspended-sediment loads were calculated by multiplying TSS concentration by surface runoff volume in 1-min intervals and then summing the 1-min sediment loads for the entire event duration (Table 5.3).

Table 5.3. Summaries of event TSS load (kg), runoff amount (mm), rainfall amount (mm), and runoff coefficient by treatment.

Treatment	**TSS load (kg)**			**Runoff (mm)**			**Rainfall (mm)**			**Runoff coefficient**		
	Min	*Mdn**	*Max*	*Min*	*Mdn*	*Max*	*Min*	*Mdn*	*Max*	*Min*	*Mdn*	*Max*
No Gravel	0.29	1.73	6.41	0.7	7.4	16.6	5.6	20.2	42.0	0.11	0.36	0.84
Low Gravel	0.04	0.38	1.86	0.3	5.5	21.5	8.8	14.8	36.8	0.02	0.35	0.58
High Gravel	0.04	0.17	0.57	0.9	3.5	9.6	4.2	15.0	29.6	0.07	0.24	0.46

*Mdn = median

Event sediment yield was quantified by dividing event TSS load by the running surface area of the stream crossing approach, which accounted for differences in approach length (i.e., the approach running surface widths did not differ substantially (Table 5.1)). Also, to account for variability in applied rainfall amounts and intensities (Table 5.2), event sediment yield was standardized per unit rainfall (mg m^{-2} mm^{-1}) according to erodibility indices defined in previous studies (Sheridan and Noske, 2007; Sheridan *et al.*, 2008):

$$TSS_{yield\ (standardized)} = TSS_{yield} / Rain_{total} \qquad \text{Equation 5.1.}$$

where TSS_{yield} is event TSS transport per unit area (mg m^{-2}) and $Rain_{total}$ is total event rainfall (mm).

5.3.5 WEPP model setup

The WEPP model (ver. 2012.8) was used to build unique hillslope profiles for each rainfall simulation, which included site-specific details related to slope, soil type, vegetation management, and rainfall event characteristics (duration, intensity, and total). Data regarding stream-crossing approach length, slope, running surface width, road vertical shape (concave, convex, linear, or S-shaped), and aspect were obtained during the July 2011 total station survey. Six slope files were created corresponding to the six stream approaches used in this study. Unique breakpoint climate files were created for each rainfall simulation experiment so that WEPP could be run in single-storm mode. Significant changes in road grade (e.g., a stream approach that has 12% slope at the top of the approach and then changes to a gentler slope of 4% near the stream) were included in the slope profiles as breakpoints by using the slope profile editor. We selected the “skid-clay loam” soil file because the model input parameter values were representative of a low-volume forest road with a clay loam soil texture. We selected the “insloped road-unrutted bare” vegetation management file because it was representative of road surface conditions following road reopening by bulldozer blading. These files for soil and vegetation management were used at each of the study sites.

The initial plant file, which is part of the initial conditions database in the vegetation management file, was “insloped road-bare”. This file was used without alteration for each of the study sites. However, it was necessary to create unique vegetation management files for each rainfall simulation experiment to reflect changes in antecedent rainfall, as well as surface cover. Specifically, cumulative rainfall amounts since bulldozer trafficking were calculated for each rainfall simulation experiment and these values were input for the variable, “Cumulative rainfall since last tillage”. Percent cover was estimated prior to each rainfall simulation experiment by walking in a zigzag pattern on the running surface for the entire length of the road approach study plots and counting the number of steps where the toe of the boot fell upon covered ground (i.e., “covered” steps/total steps*100 = percent cover). These estimates of surface cover were used as input values for initial rill and interrill cover. Rill width type was set to “permanent” because tillage was not recurring. The mean of soil bulk density measurements for each site was used as a constant input in the vegetation management files. The variable, “Days since last harvest”, was set to 3650 (10 years), essentially making this a non-factor. Forest cover adjacent to the stream-crossing approaches consisted of mature hardwood forests. The variable, “Days since last tillage” was calculated as the cumulative number of days since the stream crossing approaches were trafficked with a bulldozer (i.e., prior to the first series of No Gravel treatment rainfall simulation experiments). The remaining model input parameters associated with the initial conditions section for “insloped road-unrutted bare” were not changed, with the exception of initial ridge roughness and initial rill width.

We defined ranges of values for model input parameters that are integral to WEPP predictions of runoff and sediment yield from forest roads. Specifically, we defined ranges of values for effective hydraulic conductivity, initial ridge roughness, initial rill width, rill erodibility, interrill erodibiliy, and critical shear. Field observations of antecedent soil water content expressed relative to saturation (i.e., determined from field conditions at each site) associated with the running surface component of the stream-crossing approaches were used as inputs for the variable, initial saturation, in the soil input files. The ranges for model input parameters were chosen to reflect site conditions of the stream-crossing approaches used in this study and were based upon our own field observations, when possible. In other cases, ranges for model input parameter values were based on field experiments by Foltz *et al.* (2008) or WEPP's technical documentation (NSERL, 1995).

A Markov Chain Monte Carlo (MCMC) algorithm, DREAM_(ZS) (ter Braak and Vrugt, 2008; Vrugt *et al.*, 2008; Laloy and Vrugt, 2012), was used to efficiently select parameter ranges for WEPP that minimize the discrepancy between model predictions and observations based on a simple least squares objective function. The use of ordinary Monte Carlo-based random sampling was not feasible because of the number of potential parameter sets that had to be generated to explore the complex parameter space. DREAM_(ZS) is adaptive and efficient for finding "acceptable" parameter sets in complex inverse modeling problems (Vrugt *et al.*, 2009). The resulting posterior parameter distributions from WEPP calibration using DREAM_(ZS) were used to explore uncertainty associated with model input parameters and model predictions, as well as parameter identifiability/sensitivity, and overall prediction performance.

The main advantage of the DREAM_(ZS) optimization algorithm over traditional Monte Carlo-based random sampling is that the sampling procedure learns from experience (i.e., model performance in predicting runoff and sediment yield) and provides denser sampling in the model input parameter ranges that are associated with acceptable model runs. In this way, fewer model runs (and less computer processing time) are necessary to adequately sample the model input parameter space. DREAM_(ZS) was used to generate 10,000 unique sets of model input parameter values. The number of model runs was selected based upon analysis of chain convergence for each of the model input parameters.

We chose to set the range of effective hydraulic conductivity based upon previous field estimates of hydraulic conductivity at the stream-crossing approaches (Brown *et al.*, 2014). During model calibration, the initial range for effective hydraulic conductivity was 0.1 to 10 mm h^{-1} (Table 5.4). WEPP model runs were performed with hydraulic conductivity held constant (i.e., WEPP did not internally adjust hydraulic conductivity during event simulations). The range for initial ridge roughness was 0 to 0.08 m, with the lower values representing road conditions immediately following bulldozer trafficking and the higher values associated with maximum gravel cover on the stream approaches.

No distinct rills were observed during the rainfall simulations. However, we used a range of 0 to 0.2 m for rill width to reflect the occurrence of both interrill and rill erosion processes during the rainfall simulations. The range for rill erodibility (0.0001 to 0.01 s m^{-1}) was set to reflect the wide range of values reported in the peer-reviewed literature (e.g., Foltz *et al.*, 2008). The range of values for interrill erodibility ($2x10^6$ to $11x10^6$ kg s m^{-4}) was based upon the range of values reported in WEPP's technical documentation (NSERL, 1995). The range of values for baseline critical shear was 0.4 to 2.6 N m^{-2} (Foltz *et al.*, 2008).

5.3.6 Evaluation of model predictions

Of the 10,000 initial model runs for each rainfall simulation, the last 25% were chosen for estimating the posterior distributions of parameters. Posterior distributions were estimated from the last samples in the Markov chains when convergence of the individual chains was consistently below the threshold of 1.2 for the Gelman and Rubin statistic (Gelman and Rubin, 1996). It has been suggested that the last 25% of the samples in each chain is an appropriate characterization of the posterior distribution (Vrugt et al., 2009). In all simulations for this study, convergence was reached within 1500 samples; thus, the last 25% was a conservative estimate. To account for variability in applied rainfall amounts and intensities, model performance was based upon the comparison of observed and predicted runoff coefficients (runoff depth / rainfall depth) and sediment yield per unit rainfall (mg m^{-2} mm^{-1}).

Table 5.4. Description of model input parameters and the ranges of values used in the generation of unique sets of parameters by way of Markov Chain Monte Carlo sampling of the model input parameter ranges.

Parameter	**Description**	**Units**	**Min**	**Max**	**WEPP file**
RRINIT	initial ridge roughness after last tillage	m	0	0.08	Management
WIDTH	initial rill width	m	0	0.2	Management
K_i	baseline interrill erodibility	kg s m^{-4}	$2x10^6$	$11x10^6$	Soil
K_r	baseline rill erodibility	s m^{-1}	0.0001	0.01	Soil
SHCRIT	baseline critical shear	N m^{-2}	0.4	2.6	Soil
AVKE	effective hydraulic conductivity	mm h^{-1}	0.1	10	Soil

95% confidence intervals of model predictions resulting from the posterior parameter distributions were used to evaluate model performance in comparison to event-based runoff and sediment yield for a succession of applied rainfall events, as well as a succession of gravel treatments that represented increasing intensities of BMP implementation. Posterior distributions of the model input parameter values were expressed as empirical cumulative distribution function (ECDF) plots to identify regions of the model input parameter space (i.e., specific ranges of values for each of the model input parameters) that were associated with acceptable model runs. For a continuous variable, the gradient of an ECDF plot is equal to the probability density at that point. This means that the steepest slopes on the ECDF plot indicate the highest relative frequencies on a histogram of the posterior distribution (Figure 5.4). DREAM_(ZS) provides denser sampling in the model input parameter ranges that are associated with acceptable model runs. Therefore, we can use ECDF plots to identify the best range of model input parameter values to be used for different road surface treatments (e.g., No Gravel, Low Gravel, High Gravel).

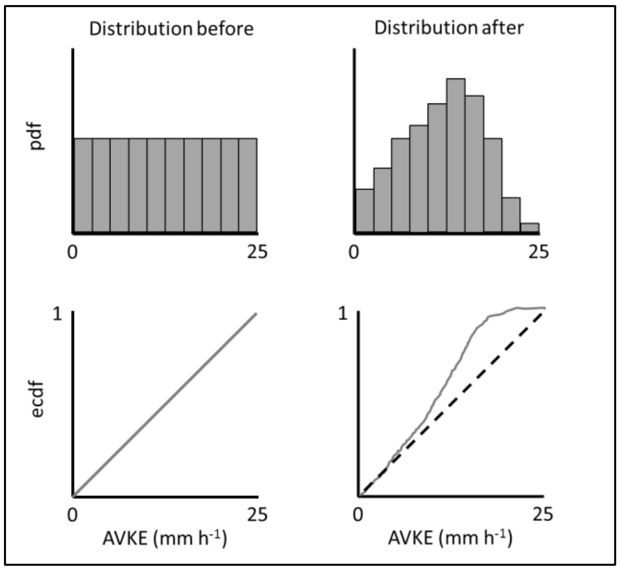

Figure 5.4. Idealized schematic depicting the relationship between probability density function (pdf) and cumulative distribution function (cdf) of the model input parameter values before and after Markov Chain Monte Carlo sampling of the parameter ranges.

5.4 Results and Discussion

5.4.1 Model performance in predicting event-based surface runoff

For many of the rainfall simulations, WEPP matched the observed runoff coefficient. A match is indicated by the star (observed runoff coefficient) being inside the box (95% confidence interval for the predicted runoff coefficients) in Figure 5.5. This means that for a given rainfall simulation, there was at least one parameter set that resulted in a prediction that matched observed runoff. WEPP also predicted higher runoff coefficients for the No Gravel treatment, which is similar to findings from the field rainfall simulations (Brown *et al.*, 2014). However, the range of predicted runoff coefficients were often very wide (Figure 5.5), reflecting substantial uncertainty associated with model input parameter values, as well as model predictions of runoff.

5.4.2 Model performance in predicting event-based sediment yield

WEPP performed well in predicting reductions in sediment yield for successive rainfall events within a given treatment, as well as reductions in sediment yield associated with increasing gravel surface cover on the stream crossing approaches (Figure 5.6). However, the prediction intervals were often wide, reflecting substantial uncertainty associated with model input parameters related to sediment yield. For example, observed sediment yield at Site 3 (Figure 5.6 C) decreased with successive rainfall simulation experiments for the No Gravel treatment (Simulation numbers 1, 2, and 3) as a result of decreased soil erodibility. WEPP predictions also reflect the effect of decreased soil erodibility as a function of successive rainfall events. In addition, observed sediment yield decreased due to increasing gravel surface cover on the approaches. WEPP predictions of sediment yield clearly show differences (i.e., sediment reductions) among the different road surface treatments that represent increasing intensities of BMP implementation. This capability is important for examining the linkages between BMP implementations and sediment reductions across broad spatial scales. However, such wide prediction intervals for runoff and sediment yield underscore the importance of explicitly accounting for uncertainty associated with model input parameters and predictions by utilizing a range of erosion predictions, as opposed to a single erosion prediction, to aid forest land managers in prescribing site-specific BMP implementations to reduce sediment delivery to water bodies.

Therefore, despite the ability of WEPP to predict relative differences in event-based sediment yield among different types of BMPs, such wide prediction intervals suggest a limited applicability for scenarios that demand a high level of prediction accuracy, such as total maximum daily load (TMDL) development. In such cases, it is recommended that prediction intervals be used to show the substantial variability in sediment yield predictions. Commonly, evaluations of WEPP model performance have compared an optimal model run to observations of runoff and sediment yield (Croke and Nethery, 2006; Sawyers *et al.*, 2012; Wade *et al.*, 2012; Brown *et al.*, 2013). Our study findings show that while a single optimal model run may be useful for comparing relative differences among treatments, it is less meaningful if a large subset of model runs (with unique combinations of values for model input parameter sets) can yield equally acceptable model predictions when defined by an objective function such as the least squares of the model residuals.

It is possible that the substantial uncertainty associated with WEPP predictions of event-based runoff and sediment yield in this research is a function of the relatively small quantities of runoff and sediment yield observed during the rainfall simulations. For example, in a few cases, we are attempting to predict runoff depth as low as 0.3 mm and sediment yield as low as 0.04 kg (Table 5.3). A runoff prediction of 1 mm in comparison to an observed runoff depth of 0.3 mm represents an overprediction by 233%. Prediction accuracies would likely improve in the case of much greater observations of runoff and sediment yield (i.e., for very large storm events or annual runoff amounts and rates of sediment delivery). For example, another study suggested that WEPP predictions of erosion could be assumed to be within ± 50% of observations for erosion predictions over longer timescales, such as in annual sediment budget analyses (Brooks *et al.*, 2006).

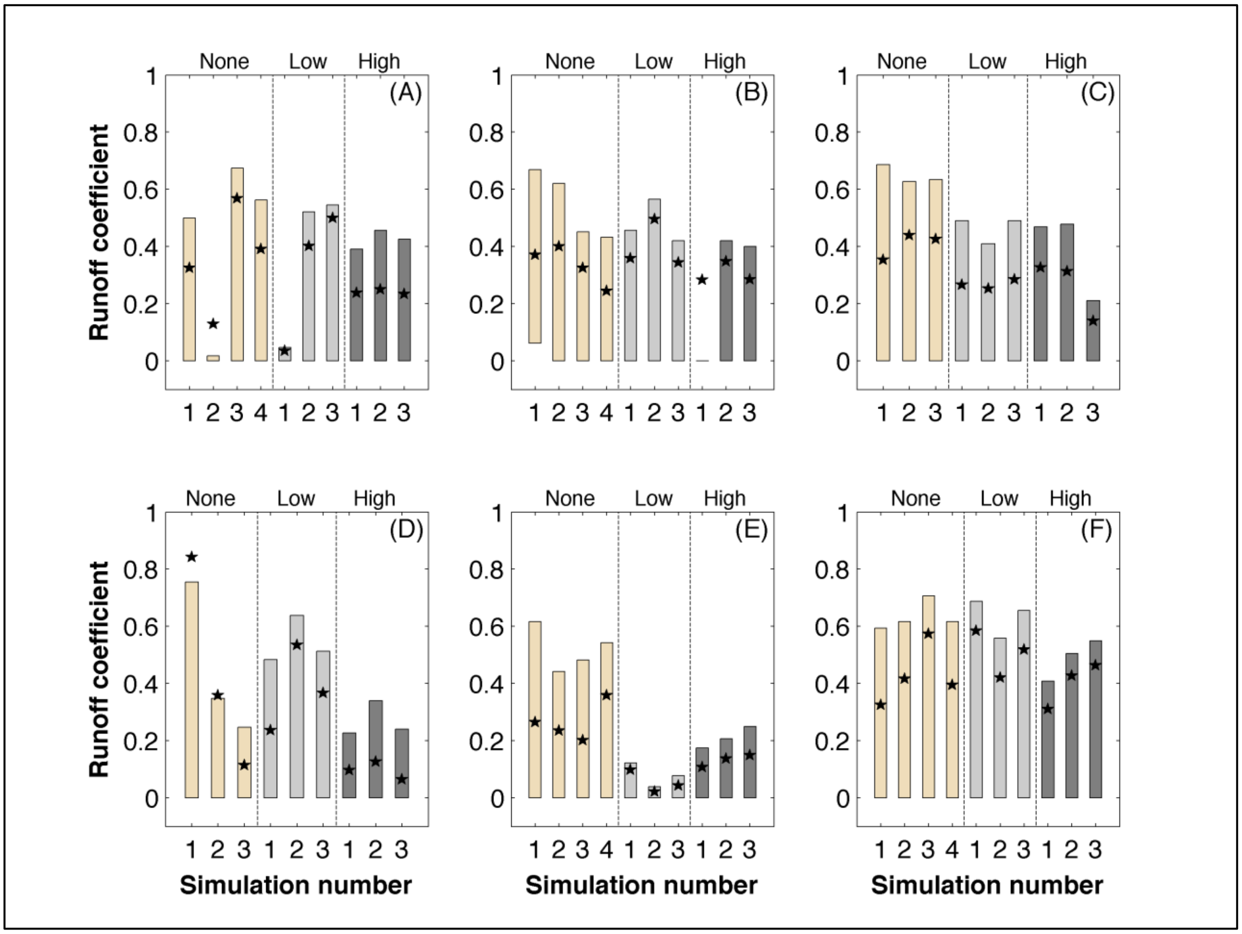

Figure 5.5. Predicted (boxes) versus observed (stars) runoff coefficients for the six stream-crossing approaches used in this study (sites 1-6 are shown as A-F) and by treatment type (None = No Gravel, Low Gravel, High Gravel). Boxes represent the 95% confidence intervals for the model predictions for each rainfall simulation. Simulation number specifies the order in which rainfall simulations were conducted within each treatment.

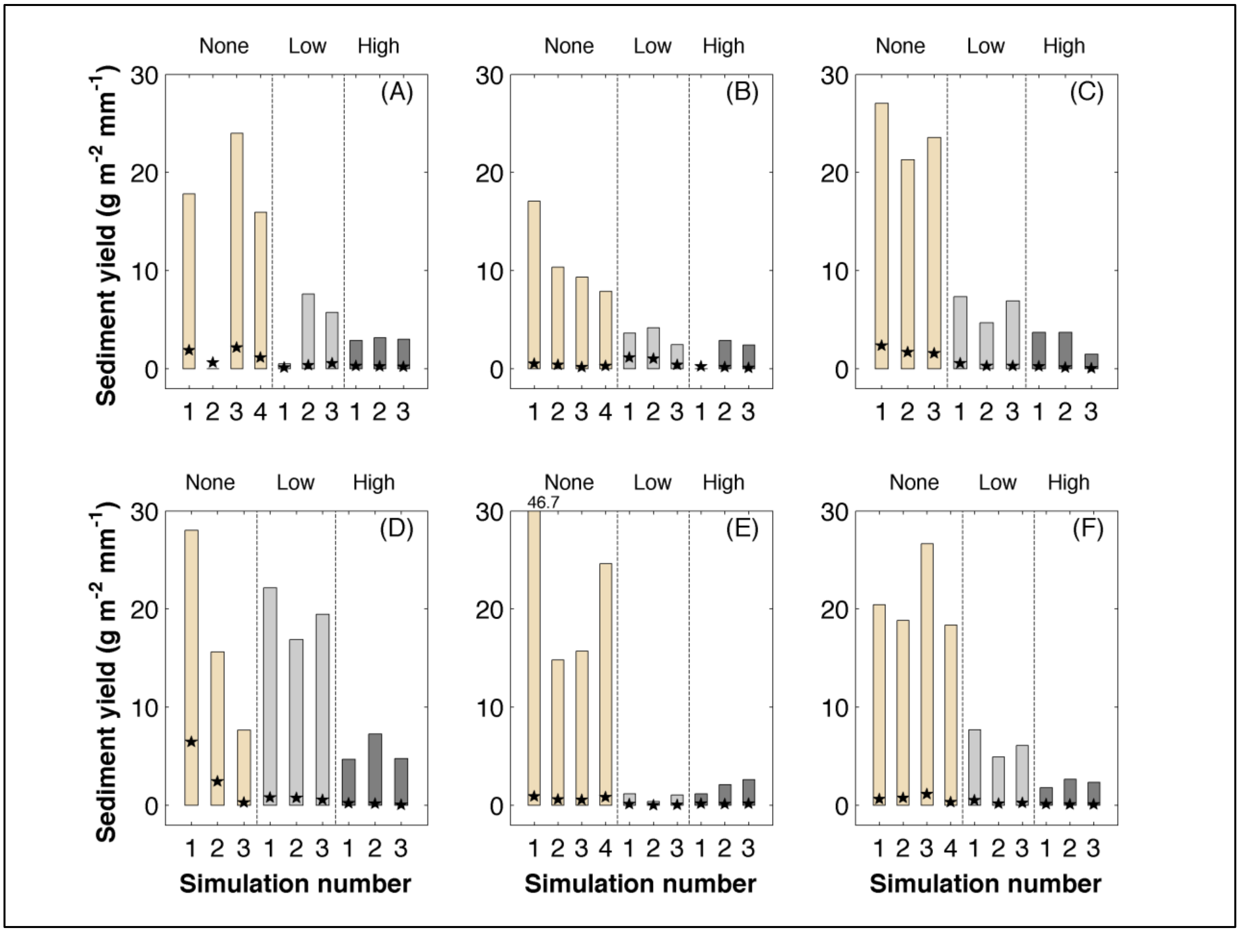

Figure 5.6. Predicted (boxes) versus observed (stars) sediment yield for the six stream-crossing approaches used in this study (sites 1-6 are shown as A-F) and by treatment type (None = No Gravel, Low Gravel, High Gravel). Boxes represent the 95% confidence intervals of predicted sediment yield for model predictions for each rainfall simulation. For instances where the prediction limit exceeded the y-axis limit (30 g m^{-2} mm^{-1}), the value is labeled at the top of the figures. Simulation number specifies the order in which rainfall simulation experiments were conducted within each treatment.

5.4.3 Model input parameter identifiability

The posterior parameter distributions for interrill erodibility (Ki), rill erodibility (Kr), and critical shear (SHCRIT) did not differ substantially from their *a priori* distribution (i.e., a uniform distribution) (Figure 5.7). This is indicated by the relatively constant slope steepness of the ECDF plots over the full range of model input parameter values for interrill erodibility, rill erodibility, and critical shear. Consequently, there are no discernable differences among the road surface treatments. Therefore, for these parameters, we cannot recommend parameter ranges that are specific to the road surface treatments used in this study. This finding indicates that these parameters are insensitive to changes in soil erodibility associated with successive rainfall events, as well as surface cover associated with the different gravel surface treatments. Yet, WEPP predictions showed decreases in sediment yield associated with successive rainfall events and increased gravel surface cover (Figure 5.6). It is postulated then, that WEPP predictions showed decreases in sediment yield because we manually changed the model input parameter values for cumulative rainfall amount since last disturbance, as well as initial rill and interrill cover, for each rainfall simulation.

For initial ridge roughness (RRINIT), better model runs for the High Gravel treatment were associated with lower values, while better model runs for the No Gravel treatment were associated with higher values (initial range = 0 to 0.08 m). For initial rill width (WIDTH), better model runs were associated with lower values (initial range = 0 to 0.2 m) for all treatments and this finding was most pronounced for the No Gravel treatment, followed by Low Gravel, and then High Gravel. For effective hydraulic conductivity (AVKE), better model runs for the Low Gravel and High Gravel treatment were associated with lower values (initial range = 0.1 to 10 mm h^{-1}).

Overall, despite using a MCMC optimization algorithm to search the model input parameter ranges, it was difficult to identify parameter ranges that were associated with acceptable model runs, especially for interrill erodibility, rill erodibility, and critical shear. Brazier et al. (2000) also found that parameter identifiability was difficult for interrill erodibility.

For physically-based models such as WEPP that have many model input parameters, there are complex interactions among model parameters that confound parameter identifiability (Beven, 2008). Therefore, in this case, it is possible for predictions to match observed runoff and sediment, but it is difficult to know if the model input parameters properly represent runoff and erosion processes that were measured in the field.

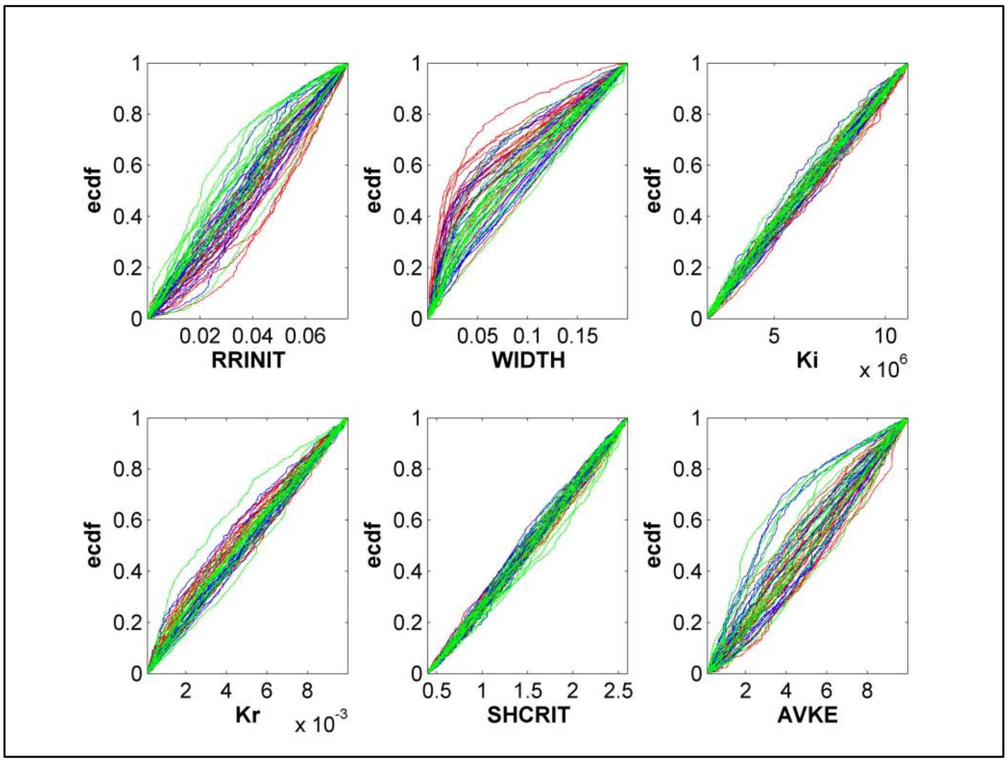

Figure 5.7. Empirical cumulative distribution function (ECDF) plots for model input parameters that were varied by way of Markov Chain Monte Carlo (MCMC) sampling. The ECDF plots shown here are for the best 2500 model runs for each rainfall simulation. No Gravel, Low Gravel, and High Gravel treatments, respectively, are associated with red, blue, and green lines. From left to right on the top row: Initial ridge roughness (m), initial rill width (m), and interrill erodibility (kg s m^{-4}). From left to right on the bottom row: Rill erodibility (s m^{-1}), critical shear (N m^{-2}), and effective hydraulic conductivity (mm h^{-1}).

5.5 Conclusions

Sediment delivery from forest roads at stream crossings is a major threat to water quality and aquatic habitat. Models are needed to evaluate the effectiveness of forestry BMPs to reduce sediment delivery over large spatial scales and to guide site-specific BMP implementations to protect water quality. In this study, WEPP model performance was evaluated for the prediction of event-based runoff and sediment yield at forest stream-crossing approaches and for different gravel surfacing treatments that represented increasing intensities of BMP implementation. WEPP was evaluated based on prediction accuracy for runoff and sediment yield, as well as its ability to distinguish between the different BMP treatments. The posterior parameter distributions that resulted from Markov Chain Monte Carlo sampling were evaluated to determine if we could recommend parameter ranges that are specific to the different road surface treatments used in this study.

WEPP was able to match observed runoff and sediment yield for many of the rainfall simulations. WEPP predicted reductions in sediment yield that were observed in the field resulting from decreased soil erodibility associated with successive rainfall events, as well as the treatment effect of increasing gravel cover on the stream-crossing approaches. However, 95% confidence intervals representing the range of predicted runoff and sediment yield for the best model runs were often very wide for each rainfall simulation. This reflects substantial uncertainty in model input parameter values, as well as model predictions. Based on analysis of the posterior distributions of model input parameters, we could not recommend ranges of values for interrill erodibility, rill erodibility, or critical shear that were specific to the different road surface treatments.

Overall, these results suggest that there is limited utility in estimating soil erosion or sediment delivery based on a single, optimized model run (i.e., one set of model parameters that result in an acceptable prediction for runoff and sediment yield). Rather, predictions should be made with a range of potential values for model input parameters related to runoff generation and sediment yield, to reflect uncertainty associated with model parameterization. In this way, a range of erosion predictions associated with different intensities of BMP implementations can be compared to aid in watershed management efforts to protect water quality, while explicitly accounting for uncertainty associated with model predictions. These results also suggest that watershed management decisions should not be based on WEPP predictions of sediment yield alone, but rather from a combined effort that includes field monitoring to determine BMP effectiveness in reducing sediment delivery, as well as the evaluation of models to estimate BMP efficacy.

5.6 Acknowledgements

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6.0 CONCLUSIONS

Forest roads and associated stream crossings are major sources of erosion and sediment delivery to streams. The effectiveness of BMPs to reduce sediment delivery from forest roads is site-specific (i.e., dependent on factors such as climate, soils, slope, road type, adequacy of surface cover and water control structures, traffic characteristics, and time since disturbance). As such, field studies are needed to evaluate the performance of various intensities of BMP implementations to reduce sediment delivery for a wide range of site conditions. Furthermore, soil erosion models are needed to evaluate BMP efficacy to reduce sediment delivery across broad spatial scales and to guide watershed management activities to protect water quality. However, soil erosion model performance must be tested for a wide range of site conditions to evaluate model utility in evaluating BMP effectiveness to reduce sediment delivery.

This research quantified annual and event-based sediment delivery rates for reopened legacy roads at forest stream crossings in the Virginia Piedmont physiographic region, USA. Sediment delivery rates were compared among stream-crossing approaches with diverse intensities of best management practice (BMP) implementations for surface cover and water control. USLE-forest and WEPP model predictions were evaluated for their ability to distinguish BMP effectiveness between road surface types (i.e., no gravel vs. completely graveled) at annual timescales. In addition, WEPP model predictions of event-based runoff and sediment yield were compared to field observations from rainfall simulation experiments to evaluate model performance in estimating BMP effectiveness for gravel surfacing treatments that represented increasing intensities of BMP implementations.

Annual sediment delivery rates from reopened (bare) legacy road approaches to stream crossings were 7.5 times higher than those of completely graveled approaches that were representative of contemporary road design, construction, and BMP implementations. In this study, the major characteristics of the problem stream-crossing approaches included inadequate water control and 90-100% bare soil conditions throughout the year. These study findings suggest that legacy roads may require additional BMP implementations upon reopening to protect water quality, especially if stream-crossing approaches are poorly located, steep, and have inadequate water control or surface cover. Currently, the VDOF BMP manual does not include specific recommendations for reopened legacy roads at stream crossings. However, our study showed that completely graveled stream-crossing approaches, as well as managing surface runoff in small amounts (i.e., with adequate water control structures) can be used to reduce sediment delivery to streams where existing conditions of the stream-crossing approaches are not ideal. Similar recommendations could be added to the BMP manual to help land managers utilize existing road networks while protecting water quality.

In terms of soil erosion model performance, neither model (USLE-forest nor WEPP) accurately predicted annual sediment delivery rates from the stream crossing approaches. This finding suggests a substantial gap between field monitoring and state-of-the-art soil erosion models in terms of their utility in evaluating BMP effectiveness. However, both models predicted substantial rates of sediment delivery for the bare roads and minimal rates of sediment delivery for the gravel roads, meaning that model predictions distinguished the different road surface types. WEPP ranked the problem road approaches better than USLE-forest.

Our findings indicate that USLE-forest may be better suited to forest land managers for several reasons. First, the data inputs to estimate annual soil erosion rates can easily be obtained through a combination of office planning (e.g., using the Web Soil Survey to obtain R and K factors) and field measurements to measure length and slope of the stream-crossing approaches, and to evaluate the conservation practice sub-factors. Conversely, WEPP is physically-based and requires the estimation of many more model input parameter values than USLE-forest. Furthermore, the input parameter values that are integral to WEPP predictions of runoff (e.g., effective hydraulic conductivity) and sediment yield (e.g., rill and interrill erodibility, critical shear) are not often known without having conducted well-controlled field experiments to estimate their values. The complexity of data requirements for the WEPP model makes it best suited for academicians and State and Federal agencies that have the resources to couple field monitoring with evaluations of erosion model performance. Secondly, we found that WEPP only marginally outperformed USLE-forest in that it ranked the problem road approaches in accordance with our measured values of annual sediment delivery rates.

Event-based surface runoff and associated total suspended solids (TSS) concentrations were compared among a succession of gravel surfacing treatments that represented increasing intensities of BMP implementations on reopened approaches. In this study, sediment yield was reduced by successive rainfall events on the stream-crossing approaches, as well as increasing gravel surface cover. Study findings indicated that reopened legacy roads and associated stream-crossing approaches can deliver significant quantities of sediment if roads are not adequately closed or maintained and that corrective best management practices (BMPs), such as gravel and appropriate spacing of water control structures, can reduce sediment delivery to streams.

Field data from the rainfall simulation experiments were used to evaluate WEPP model performance in estimating the effectiveness of increasing intensities of gravel surfacing BMPs to reduce event-based sediment yield from reopened stream-crossing approaches. The objectives of this study were to explore WEPP parameter identifiability/sensitivity and uncertainty, prediction uncertainty, and overall prediction performance with an inverse-modeling approach. Specific research questions were: (1) How well does WEPP predict event-based runoff and sediment yield from reopened forest roads at stream crossings?; (2) Can WEPP simulate changes in soil erodibility from successive rain events and the treatment effect of increasing gravel surface cover on the stream-crossing approaches? (3) Can distinct regions of the model input parameter space be identified in association with the different road surface treatments?

Sediment yield predictions showed clear differences among successive rainfall events within a given treatment (i.e. the effect of reduced soil erodibility on sediment yield), as well as differences among the different road surface treatments (i.e, the effect of increasing gravel surface cover on sediment yield). However, 95% confidence intervals representing the best model predictions of runoff and sediment yield were often very wide, reflecting substantial uncertainty in model input parameter values, as well as model predictions. Analysis of the posterior distributions of the model input parameters related to sediment yield indicated that for interrill erodibility, rill erodibility, and critical shear, we cannot recommend parameter ranges that are specific to the different surface treatments. Overall, research findings indicate that watershed management decisions that hinge upon WEPP predictions of sediment yield from forest roads must necessarily take into account a wide range of potential erosion rates for specific management scenarios. In this way, a range of erosion predictions associated with different intensities of BMP implementations can be compared to aid in watershed management efforts to protect water quality, while explicitly accounting for uncertainty associated with model predictions.

Overall, this research suggests that road planning is arguably the most crucial BMP to reduce sediment inputs to streams. For example, a poorly located road that results in a steep grade can be difficult and costly to improve. Therefore, future research should seek to improve road planning technology to reduce the total road area within a given watershed, minimize stream crossings, maintain relatively gentle road gradients, and avoid locations where it is difficult to shed water from the road surface. In this way, water quality protection is not overly-dependent on post-construction BMP implementations to correct road deficiencies that resulted from poor planning.

We found that increasing gravel surface cover decreased sediment yield from reopened stream-crossing approaches. However, we did not investigate the effects of traffic type and frequency on BMP effectiveness. It is hypothesized that the effect of completely graveled stream-crossing approaches on sediment yield reductions would be much more pronounced than that of partially-graveled approaches (e.g. 50% coverage) had the roads been subjected to truck traffic. In addition, our research focused on the running surface component of the stream-crossing approaches. However, cutslopes, ditches, as well as the stream crossing itself, can also be important sources of sediment delivery from roads at stream crossings. Sediment delivery is a problem for forest trails of all kinds. As such, quantitative studies of sediment delivery from recreational trails are needed to guide management of surface runoff from these sediment sources.

This research suggests a limited usefulness of existing soil erosion models to evaluate BMP effectiveness in reducing sediment delivery across broad spatial scales, which is critical for watershed management decisions to protect water quality. Our evaluations of USLE-forest and WEPP suggest that the models may be used to distinguish relative differences in sediment yield between vastly different road surface treatments (e.g., unsurfaced vs. completely graveled). However, subtler differences in road surface cover or adequacy of water control structures, for example, may be undetected due to substantial uncertainty associated with model input parameter values, as well as model predictions of sediment yield. Overall, this research underscores the importance of considering a range of model input parameter values, which will result in a range of predictions of soil erosion or sediment delivery for specific BMP scenarios. This approach accounts for the uncertainty associated with model parameterization and model predictions and this information can be utilized in watershed planning efforts to protect water quality.

Finally, future research initiatives should seek to standardize data collection procedures during field monitoring of BMP effectiveness to reduce sediment delivery from forest roads. It is often difficult to directly compare measurements of surface runoff and sediment yield due to differences in experiment duration (e.g., event-based vs. annual), experimental plot size, methods of quantifying rates of sediment delivery (e.g., sediment trapping vs. sediment loads calculated as runoff volume x TSS concentration), and rainfall inputs (natural vs. applied). In addition, the diversity of equipment used in rainfall simulation experiments leads to differences in rainfall intensity, rain drop size, and overall erosivity, further confounding the comparison of study findings. A more unified approach to data collection, as well as the quantification of erodibility indices, would allow for more direct comparisons of BMP effectiveness in reducing sediment delivery across broad spatial scales.