Soil Erosion from Forest Haul Roads at Stream Crossings as Influenced by Road Attributes

Albert Joseph Lang

Dissertation submitted to the faculty of the
Virginia Polytechnic Institute and State University
in partial fulfillment of the requirements for the degree of

Doctor of Philosophy
In
Forest Resources and Environmental Conservation

W. Michael Aust (Co-Chair)
M. Chad Bolding (Co-Chair)
Kevin J. McGuire
Erik B. Schilling

May 4, 2016
Blacksburg, VA

Keywords: Forestry best management practices, potential sediment delivery, ditched forest haul roads, forest operations, stream crossing approaches, soil erosion modeling, model performance

Soil Erosion from Forest Haul Roads at Stream Crossings as Influenced by Road Attributes

Albert Joseph Lang

ABSTRACT

Forest roads and stream crossings can be important sources of sediment in forested watersheds. The purpose of this research was to compare trapped sediment and forestry best management practice (BMP) effectiveness from haul road stream crossing approaches and ditches. The three studies in this dissertation provide a quantitative assessment of sediment production and potential sediment delivery from forest haul roads in the Virginia Piedmont and Ridge and Valley regions. Sediment production rates were measured and modeled to evaluate and compare road and ditch segments near stream crossings with various ranges of road attributes, BMPs, and management objectives.

Sediment mass delivered to traps from 37 haul road stream crossing approaches ranged from <0.1 to 2.7 Mg for the one year collection. Collectively, five approaches accounted for 82% of the total sediment mass trapped. Approaches were categorized into Low, Standard, and High road quality rankings according to road attributes. Seventy-one percent (5 of 7) of Low ranked approaches delivered sediment to traps at rates greater than 11.2 Mg ha^{-1} yr^{-1}. Nearly 90% of Standard or High road quality approaches generated less than 0.1 Mg of sediment over one year. Among approaches with less than 0.1 Mg of trapped sediment, road gradients ranged from 1% to 13%, bare soil ranged from 2% to 94%, and distances to nearest water control structures ranged from 8.2 to 427.0 m. Such a wide spectrum of road attributes with relatively low levels of trapped

sediment indicate that contemporary BMPs can mitigate problematic road attributes and reduce erosion and sediment delivery.

Three erosion models, USLE-forest, RUSLE2, and WEPP were compared to trapped sediment data from the 37 forest haul road stream crossing approaches in the first study. The second study assessed model performance from five variations of the three erosion models that have been used in previous forest operations research, USLE-roadway, USLE-soil survey, RUSLE2, WEPP-default, and WEPP-modified. The results suggest that these soil erosion models could estimate erosion and sediment delivery within 5 Mg ha^{-1} yr^{-1} for most approaches with erosion rates less than 11.2 Mg ha^{-1} yr^{-1}, while model estimates varied widely for approaches that eroded above 11.2 Mg ha^{-1} yr^{-1}. Based on the results from the 12 evaluations of model performance, the modified version of WEPP consistently performed better compared to all other model variations tested. However, results from the study suggest that additional field evaluations and improvement of soil erosion models are needed for stream crossings. The soil erosion models evaluated are not an adequate surrogate for informing policy decisions.

The third study evaluated sediment control effectiveness of four commonly recommended ditch BMPs on forest haul road stream crossing approaches. Sixty ditch segments near stream crossings were reconstructed and four ditch BMP treatments were tested. Ditch treatments were bare (Bare), grass seed with lime fertilizer (Seed), grass seed with lime fertilizer and erosion control mat (Mat), rock check dams (Dam), and completely rocked (Rock). Mat treatments had significantly lower erosion rates than Bare and Dam, while Rock and Seed produced intermediate levels. Findings of this study suggest Mat, Seed, and Rock ditch BMPs were effective at reducing erosion, but Mat was most effective directly following construction because Mat provided immediate soil protection measures. Any BMPs that reduce bare soil can provide reduction in erosion and even natural site condition, including litterfall and invasive vegetation can provide erosion control. However, ditch BMPs cannot mitigate inadequate water control structures.

Overall, forest roads and stream crossings have the potential to be major contributors of sediment in forested watersheds when roads are not designed well or when BMPs are not properly implemented. Forestry BMPs reduce stormwater runoff velocity and volume from forest roads, but can have varying levels of effectiveness due to site-specific conditions. Operational field studies provide valuable information regarding erosion and sediment delivery rates, which helps guide BMP recommendations and subsequently enhances water quality protection.

ACKNOWLEDGEMENTS

I would like to thank my co-advisors, Drs. W. Michael Aust and M. Chad Bolding for the opportunity to study and conduct research in the field of forest operations at Virginia Tech. I am grateful for their patience and provision of mentorship, encouragement, and logical advice every step of the way. I would also like to thank my other committee members, Drs. Kevin J. McGuire and Erik B. Schilling for their insights into the project development and excellent manuscript reviews. The National Council for Air and Stream Improvement provided financial support that made the research projects and my graduate education possible. Additional thanks are due to Greg Scheerer, Dwayne Stilwell, Mark Miller, and Jesse Overcash for their assistance locating suitable project sites and allowing us to conduct field studies on property they manage.

I have been fortunate to work with numerous people throughout my time at Virginia Tech. I am very grateful for the field, lab, and/or analysis assistance provided by Tal Roberts, Clay Sawyers, Dave Mitchem, Kris Brown, David Passauer, Brian Morris, Richie Cristan, Drew Cockram, Andy Neal, Brian Parkhurst, Mike Durbiano, Robert Crowther, Lindsey Nolan, Andrew Vinson, and Victoria Nystorm. Additionally, I am thankful for Kathie Hollandsworth, Sue Snow, and Tracey Sherman whom provided administrative support.

I am appreciative of my parents, Albert and Laura Lang, and my in-laws, Kenneth and Mary Kobylinski, for their love, support, and encouragement. Finally, I could have never finished without my wife, Abby, whom patiently waited, supported, encouraged me when I needed help most.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

SOIL EROSION FROM FOREST HAUL ROADS AT STREAM CROSSINGS AS INFLUENCED BY ROAD ATTRIBUTES

1.0 Introduction

Forest roads and stream crossings have been identified as potentially important sources of sediment in forested watersheds (Yoho 1980, Jackson et al. 2004, Aust et al. 2015). Excessive sediment affects water quality by altering the chemical, biological, and physical components of streams. Sediment can adsorb contaminates, alter stream temperature, oxygen level, and pH, decrease photosynthetic rates of aquatic plants, impair aquatic biota, and change stream geomorphology (Wood and Armitage 1997, Henley et al. 2000, Jackson et al. 2005, Jones et al. 2011). Extensive sediment deposition in streams can take years to export, which can affect the recovery process of stream habitats (Jackson et al. 2005). The Federal Water Pollution Control Act and its amendments (Clean Water Act [CWA]) mandate that waters of the United States be maintained or restored to acceptable chemical, physical, and biological criteria whenever attainable. Section 208(b1A2F) of this Act specifically mandates ("to a feasible extent") the control of non-point source pollutants from silvicultural operations.

Potential water quality issues associated with forest operations have been recognized and addressed for decades (Kraebel 1936; Bailey 1948; Trimble and Sartz 1957). Following the CWA, state agencies officially developed forestry best management practice (BMP) guidelines to address non-point source pollution from silvicultural operations. BMPs minimize erosion by controlling the volume and velocity of stormwater runoff and can greatly reduce sediment delivery to streams. Water quality and economic efficiency are foci of protection efforts during pre-harvest planning and operations

(Anderson and Lockaby 2011a). State agencies continue to improve and modify BMP recommendations for forest operations based on applicable research findings. Several extensive reviews of forestry BMP literature report minimal impacts to water quality when BMP guidelines are properly implemented (Aust and Blinn 2004, Ice 2004, Shepard 2006, NCASI 2012, Cristan et al. 2016). However, researchers have recognized that the greatest potential for increased sediment delivery is often associated with poorly designed or maintained roads and stream crossings (Edwards et al. 2015).

Forest roads expose soil and alter hillslope hydrology, which affects the timing, quantity, and pathways of water through catchments (Dymond et al. 2014). Stream crossings and road approaches are of particular interest because they provide direct connections to streams (Croke et al. 2005). Erosion control at stream crossings can directly reduce water quality impacts from forest operations. However, the amount of sediment delivered to streams from crossings is temporally and spatially complex. Sediment delivery from forest roads is governed by factors involving the road attributes, soil composition, and infiltration characteristics (Luce and Black 1999, Brown et al. 2013). Previous stream crossing research has shown influences of various crossing types (Thompson et al. 1996, Aust et al. 2011), BMPs (Brown et al. 2013, 2015), installations (Morris et al. 2016), permanency (Taylor et al. 1999), and decommissioning (Madje 2001). Many researchers have demonstrated the reduction in erosion with use of forestry BMPs, but further research is necessary to quantify the effectiveness and costs of specific BMPs over a range of site characteristics (Anderson and Lockaby 2011b).

Although the successes of state forestry BMP programs for protection of water quality are recognized, individualized problems with roads and stream crossings persist and create impetus for considering regulation of forest management (Boston 2012, Loehle et al. 2014). Legal controversy over ditched forest roads at stream crossings initiated a series of litigation that have challenged the Environmental Protection Agency's (EPA) "silvicultural exemption" (Boston and Thompson 2009). Silvicultural stream crossings are typically exempt from obtaining point source permits. Legal controversy over these exemptions has led to a United States Supreme Court ruling that allowed the EPA to manage point source permitting as they deemed appropriate (McCurdy and Timmons, 2013). The EPA has since upheld the silvicultural exemption, although legal cases regarding this matter continue (EPA, 2015). Regardless of current and future legal outcomes, stream crossings remain an important area of research for foresters.

Field experiments can provide valuable information about erosion and sediment delivery rates. Focusing field experiments along operational stream crossings provides an opportunity to approximate sediment delivery impacts; assess the effectiveness current BMPs; and compare sediment delivery rates with erosion model estimates. This information can help forest managers decide when, where, and how to allocate limited budgets for BMP implementation. Improvements in the effectiveness of BMP to minimize erosion and sediment delivery to streams can improve water quality and protect aquatic habitats in managed forests. Erosion models can provide forest managers with a time-efficient and cost-effective tool to evaluate management choices. However, models need to be tested in order to address the utility of model predictions. Additionally, field experiment data may assist policymakers in their assessment of the need for additional policy or policy changes. Exploring the impacts of forest management through field research and continuing to develop and modify practices is needed to accommodate society's demand for forest products, further scientific knowledge, and maintain healthy forested watersheds. Thus, the intent of this dissertation was to quantify and model road and ditch sediment delivery along haul roads near stream crossings with various road standards and BMPs to evaluate site characteristics and erosion model predictions associated with a range of sediment delivery rates.

1.1 Objectives and Organization

This dissertation is organized into five chapters. The first chapter provides an outline for the dissertation and evaluates how forest road BMPs and stream crossing characteristics affect sediment delivery in forested watersheds. Chapters two through four were designed to be separate manuscripts that have been, or will be, submitted for peer-review publication. Chapter 5 summarizes the findings from this dissertation and discusses the importance, applicability, and implications of the dissertation results for forest management and policy.

The second chapter presents the results of an observational field experiment in the Piedmont and Ridge and Valley physiographic regions of Virginia. This study was conducted to evaluate factors affecting sediment deposition from 37 haul road stream crossing approaches with varying road/trafficability standards. Conveyor belt diverters and silt fence sediment traps were used to collect sediment laden runoff. Stream crossing approaches in the Piedmont region were a part of intensive, short-rotation, pine silvicultural management, while approaches in the Ridge and Valley region were part of extensive, long-rotation, hardwood silvicultural management. The objectives of this manuscript were to measure sediment delivery from approaches and examine the relationships between road attributes, Virginia's BMP audit assessment, and measured sediment. This study investigated operational stream crossings and evaluated the effectiveness of BMPs in minimizing sediment delivery to streams. Data from this study were presented at the 17th Biennial Southern Silviculture Research Conference and the Society of American Foresters National Convention. This manuscript was written by Albert Lang with contributions from Drs. W. Michael Aust, M. Chad Bolding, and Erik B. Schilling. This manuscript was submitted to a peer-reviewed journal for publication consideration.

The third chapter presents the sediment delivery data from the 37 forest haul road stream crossing approaches in the second chapter and compares the sediment delivery data to model predictions of sediment delivery. Erosion models included the Universal Soil Loss Equation (USLE), Revised Universal Soil Loss Equation version 2 (RUSLE2), and Water Erosion Prediction Project (WEPP). The study assessed five variations of the three erosion models (USLE-roadway, USLE-soil survey, RUSLE2, WEPP-default, and WEPP-modified) using summary statistics, nonparametric analyses, linear relationships, percent agreement within assigned erosion categories, and contingency table metrics. The objectives of this manuscript were to evaluate model performance and assess model utility for identifying stream crossing approaches that may require additional BMPs. Data from this study were presented at the 37th Council on Forest Engineering Annual Meeting. This manuscript was written by Albert Lang, with contributions from Drs. W. Michael Aust, M. Chad Bolding, Kevin J. McGuire, and Erik B. Schilling. This manuscript was submitted to a peer-reviewed journal for publication consideration.

The fourth chapter presents the results from a field experiment that evaluated ditch BMPs for haul roads near stream crossings in the Ridge and Valley region of Virginia. Sixty 50-foot ditch segments were reconstructed using a bulldozer and farm tractor and five ditch BMP treatments were applied within a completely randomized design (11-13 replications per treatment). Ditch BMP treatments were (1) bare ditch, (2) grass seed with lime fertilizer, (3) grass seed with lime fertilizer and erosion control mat, (4) rock check dams, and (5) completely rocked. Silt fence sediment traps were used to collect sediment deposits for one year. The primary objective was to evaluate erosion control effectiveness due to ditch BMPs and secondarily to quantify ditch BMP implementation cost. Data from this study were presented at the 18th Biennial Southern Silviculture Research Conference and the 38th Council on Forest Engineering Annual Meeting. This manuscript was written by Albert Lang, with contributions from Drs. W. Michael Aust, M. Chad Bolding, Kevin J. McGuire, and Erik B. Schilling. This manuscript was submitted to a peer-reviewed journal for publication consideration.

1.2 Literature cited

Anderson, C.J., and B.G. Lockaby. 2011a. The effectiveness of forestry best management practices for sediment control in the Southeastern United States: A literature Review. *Southern Journal of Applied Forestry* 35(4): 170-177.

Anderson, C.J., and B.G. Lockaby. 2011b. Research gaps related to forest management and stream sediment in the United States. *Environmental Management* 47(2): 303-313.

Aust, W. M., M.C. Bolding, and S.M. Barrett. 2015. Best management practices for low-volume forest roads in the Piedmont region: Summary and implications of research. *Transportation Research Record: Journal of Transportation Research Board* 2472: 51-55.

Aust, W.M., and C.R. Blinn. 2004. Forestry best management practices for timber harvesting and site preparation in the eastern United States: An overview of water quality and productivity research during the past 20 years (1982–2002). *Water, Air, and Soil Pollution: Focus* 4(1): 5-36.

Aust, W.M., M.B. Carroll, M.C. Bolding, and C.A. Dolloff. 2011. Operational forest stream crossings effects on water quality in the Virginia Piedmont. *Southern Journal of Applied Forestry* 35(3): 123-130.

Bailey, R.W. 1948. Reducing runoff and siltation through forest and range management. *Journal of Soil and Water Conservation* 3: 24-31.

Boston, K. 2012. Impact of the Ninth Circuit Court ruling (Northwest Environmental Defense Center v. Brown) regarding forest roads and the Clean Water Act. *Journal of Forestry* 110(6): 344-346.

Boston, K., and M. Thompson. 2009. An argument for placing logging roads under the NPDES program. *Ecology Law Currents* 36: 169-176.

Brown, K.R., K.J. McGuire, W.M. Aust, W.C. Hession, and C.A. Dolloff. 2015. The effect of increasing gravel cover on forest roads for reduced sediment delivery to stream crossings. *Hydrological Processes* 29(6): 1129-1140.

Brown, K.R., W.M. Aust, and K.J. McGuire. 2013. Sediment delivery from bare and graveled forest road stream crossing approaches in the Virginia Piedmont. *Forest Ecology and Management* 310: 836-846.

Cristan, R., W.M. Aust, M.C. Bolding, S.M. Barrett, J.F. Munsell, and E.B. Schilling. 2016. Effectiveness of forestry best management practices in the United States: Literature review. *Forest Ecology and Management* 360: 133-151.

Croke, J., S. Mockler, P. Fogarty, and I. Takken. 2005. Sediment concentration changes in runoff pathways from a forest road network and the resultant spatial pattern of catchment connectivity. *Geomorphology* 68(3): 257-268.

Dymond, S.F., W.M. Aust, S.P. Prisley, M.H. Eisenbies, and J.M. Vose. 2014. Application of a distributed process-based hydrologic model to estimate the effects of forest road density on stormflows in the southern Appalachians. *Forest Science* 60(6): 1213-1223.

Edwards, P.J., J.E. Schoonover, and K.W.J.Willard. 2015. Guiding principles for management of forested. agricultural, and urban watersheds. *Journal of Contemporary Water Research and Education* 154(4): 60-84.

Environmental Protection Agency (EPA). 2015. Notice of opportunity to provide information on existing programs that protect water quality from forest road discharges. *Federal Register* 80(217): 69653-69660.

Henley, W.F., M.A. Patterson, R.J. Neves, and A.D. Lemly. 2000. Effects of sedimentation and turbidity on lotic food webs: A concise review for natural resource managers. *Reviews in Fisheries Science* 8(2): 125-139.

National Council for Air and Stream Improvement, Inc. (NCASI). 2012. *Assessing the effectiveness of contemporary forestry best management practices (BMPs): Focus on roads*. Special Report No. 12-01. Research Triangle Park, NC: National Council for Air and Stream Improvement, Inc.

Ice, G.G. 2004. History of innovative best management practice development and its role in addressing water quality limited waterbodies. *Journal of Environmental Engineering* 130(6): 684-689.

Jackson, C.R. G. Sun, D. Amatya, W.T. Swank, M. Riedel, J. Patric, T. Williams, J.M. Vose, C. Trettin, W.M. Aust, R.S. Beasley, H. Williston, and G.G. Ice. 2004. Fifty years of forest hydrology in the Southeast. In *A century of forest and wildland watershed lessons*, ed. G.G. Ice and J.D. Stednick, 33-112. Bethesda, MD: Society of American Foresters.

Jackson, C.R., J.K. Martin, D.S. Leigh, and L.T. West. 2005. A southeastern Piedmont watershed sediment budget: Evidence for a multi-millennial agricultural legacy. *Journal of Soil and Water Conservation* 60(6): 298-310.

Jones, J.I., J.F. Murphy, A.L. Collins, D.A. Sear, P.S. Naden, and P.D. Armitage. 2011. The impact of fine sediment on macro-invertebrates. *River Research and Applications* 28(8): 1055-1071.

Kraebel, C.J. 1936. Erosion control on mountain roads. US Department of Agriculture Circ. 380. [Old series]. 44 pp.

Loehle, C., T.B. Wigley Jr, A. Lucier Jr, E.B. Schilling, R.J. Danehy, and G.G. Ice. 2014. Toward improved water quality in forestry: opportunities and challenges in a changing regulatory environment. *Journal of Forestry* 112(1): 41-50.

Luce, C.H., and T.A. Black. 1999. Sediment production from forest roads in western Oregon. *Water Resources Research* 35: 2561-2570.

McCurdy, M.G., and D.L. Timmons. 2013. Questions remain for the timber industry after Supreme Court's decision in Decker V. Northwest Environmental Defense Center. *Environmental Law* 43(2013): 827-989.

Morris, B.C., M.C. Bolding, W.M. Aust, K.J. McGuire, E.B. Schilling, and J. Sullivan. 2016. Differing levels of forestry best management practices at stream crossing structures affect sediment delivery and installation costs. *Water*. 8(3): 92.

Shepard, J.P. 2006. Water quality protection in bioenergy production: the US system of forestry best management practices. *Biomass Bioenergy* 30(4): 378-384.

Taylor, S.E., R.B. Rummer, K.H. Yoo, R.A. Welch, and J.D. Thompson. 1999. What we know--and don't know--about water quality at stream crossings. *Journal of Forestry* 97(8): 12-17.

Thompson, J.D., S.E. Taylor, J.E. Glazin, R.B. Rummer, and R.A. Albright. 1996. Water quality impacts from low-water stream crossings. ASAE Paper No. 965015. St Joseph, Mich.: ASAE.

Trimble, G.R. and R.S. Sartz. 1957. How far from a stream should a logging road be located? *Journal of Forestry* 55(5): 339-341.

Wood, P.J., and P.D. Armitage. 1997. Biological effects of fine sediment in the lotic environment. *Environmental Management* 21(2): 203-217.

Yoho, N.S. 1980. Forest management and sediment production in the South: A review. *Southern Journal of Applied Forestry* 4(1): 27-36.

2.0 Forest Haul Road Attributes at Stream Crossings Influence Sediment Delivery

Albert J. Lang[a], W. Michael Aust[b], M. Chad Bolding[c], Kevin J. McGuire[d], and Erik B. Schilling[e]

2.1 Abstract

Forest road best management practices (BMPs) and road attributes influence sediment delivery from haul road stream crossing approaches. We estimated sediment delivery by trapping storm runoff for one year from 37 haul road stream crossing approaches within the Piedmont and Ridge and Valley (RV) regions of Virginia. Each approach was categorized into a road quality rank (Low, Standard, and High) according to slope, bare soil, distance to the nearest water control structure, traffic frequency, and level of surface armoring. Median trapped sediment mass was significantly different among road quality rankings ($p \leq 0.0011$). A post-hoc Steel-Dwass test showed that the median sediment mass for Low (451.5 kg) was significantly greater than Standard (19.3 kg) and High (1.4 kg). Additionally, Standard ranked approaches were significant greater than High. Piedmont approaches tended to erode more readily than approaches in the RV region ($p \leq 0.0252$) despite additional water control measures. Seventy-five percent of

[a] Graduate Research Assistant, Department of Forest Resources and Environmental Conservation (FREC), Virginia Tech, 305 Cheatham Hall, 310 West Campus Drive, Blacksburg, Virginia, 24061.
[b] Professor of Forestry, FREC, Virginia Tech, 228 Cheatham Hall 310 West Campus Drive, Blacksburg, Virginia, 24061.
[c] Associate Professor of Forest Engineering FREC, Virginia Tech, 228 Cheatham Hall 310 West Campus Drive, Blacksburg, Virginia, 24061.
[d] Associate Professor of Hydrology, FREC and Virginia Water Resource Research Center, Virginia Tech 210 Cheatham Hall, 310 West Campus Drive, Blacksburg, Virginia, 24061.
[e] Senior Research Scientist, National Council for Air and Stream Improvement Inc., Aubrey Texas 76227.

approaches monitored generated sediment masses less than 100 kg. Our findings indicate that most stream crossing BMPs were effective; however, some stream approaches required additional water turnout maintenance to ensure proper drainage and greater soil cover on approach surfaces.

2.2 Introduction

Sediment is the most prevalent water pollutant from forest operations in the United States (Binkley and Brown 1993, Grace 2005, Anderson and Lockaby 2011). Sediment affects water chemistry, alters stream temperature, serves as a mechanism to transport contaminants, decreases water clarity and photosynthesis rates of aquatic plants, and impairs aquatic habitat and wildlife (Ryan 1991, Wood and Armitage 1997, Henley et al. 2000). Sedimentation from forest operations is often associated with the transportation network, including roads, skid trails, stream crossings, and log decks (Aust et al. 2015, Cristan et al. 2016). This infrastructure has been cited as primary sources of soil erosion within managed forests around the world (Fransen et al. 2001, Chappell et al. 2004, Sidle et al. 2004, Kreutzweiser et al. 2005, Croke et al. 2006, Jordán and Martínez-Zavala 2008, Anderson and Lockaby 2011).

In the United States, the Federal Water Pollution Control Act of 1972 and associated amendments (also known as the Clean Water Act [CWA]), was intended to restore and maintain the integrity of the Nation's waters (CWA, Section 101a). Under this act, states develop programs to address nonpoint source pollution with technical assistance from the EPA. Sediment pollution from most forest operations is classified as a nonpoint source activity under Section 208. State forestry agencies developed Best Management Practices (BMPs) to reduce erosion and sediment delivery (Ice 2004).

Forestry BMPs address a variety of nonpoint source pollutants, including nutrients, temperature, organics, and chemicals; however, their primary purpose is to reduce sediment delivery to streams (Shepard 2006).

Road networks can pose ecological challenges and require careful planning, management, and design to reduce environmental impacts and operational costs (Forman and Alexander 1998, Conrad et al. 2012). Forest access roads provide a level of utility and environmental protection at an acceptable cost (Kochenderfer et al. 1984). The desired level of utility dictates road construction standards, such as subgrade width, ditch width, cut and fill slope ratios, gradient, and curvature (Walbridge 1997). Forest roads have low standards with many of the following characteristics: unpaved, single lane, high clearance, constructed with native materials, minimum water control, and steep grades. These road characteristics are the lowest road standards in which low volume log truck traffic may access forestland (Kochenderfer and Helvey 1987). Low standard roads frequently contain exposed and compacted mineral soils, which may reduce infiltration and generate overland flow more readily than undisturbed soil (Kochenderfer and Helvey 1987, Grace 2005, Ziegler et al. 2007). Soil erosion is common on such low standard road surfaces, but BMPs are frequently used to prevent or minimize eroded material deposition into stream networks (Croke and Hairsine 2006, North Carolina Forest Service 2014). Sediment delivery at stream crossings is driven by precipitation (e.g., amount, form, intensity, and duration), while the magnitude of sediment delivery is influenced by site-specific characteristics (e.g., road dimensions, design, location, percent canopy cover, percent bare soil, and BMPs) (Grace and Zarnoch 2013), traffic (e.g., volume, weight, and type) (Luce and Black 2001), and stream crossing type (Megahan and

Ketcheson 1996, Aust et al. 2003, Aust et al. 2011). Subsequently, sediment delivery rates are dynamic with respect to spatial and temporal factors (Lane and Sheridan 2002, Croke et al. 2006, Brown et al. 2013).

All states with significant forestry operations have developed nonpoint source pollution control programs based on implementation of BMPs (Ice et al. 2010) with most states implementing compliance monitoring programs (Ellefson et al. 2001). Considerable research has shown that properly implemented forestry BMPs reduce nonpoint source pollution and protect water quality (Stuart and Edwards 2006, Edwards and Williard 2010, Cristan et al. 2016). Recent research projects have also estimated sediment delivery and BMP effectiveness on reopened legacy stream crossing approaches (Brown et al. 2013, 2015) and operational skid trail closures (Sawyers et al. 2012, Wade et al. 2012a,b, Wear et al. 2013). Research clearly indicate that stream crossings can be potential sources of sediment and BMPs can minimize sedimentation. Previous stream crossing research has shown the influence of crossings of various types (Thompson et al. 1996, Aust et al. 2011), installations (Thompson et al. 1996, Morris et al. 2016), permanency (Taylor et al. 1999), and decommissioning (Madej 2001). However, few operational comparisons have evaluated sedimentation and BMP effectiveness from haul road stream crossing approaches. Therefore, the objectives of this study were to: (1) directly measure sediment delivery from haul road stream crossing approaches with varying road characteristics and BMP implementation; and, (2) examine relationships between road attributes, Virginia's BMP audit assessment, and measured sediment.

2.3 Methods

2.3.1 Study Sites

We selected 37 permanent forest haul road stream crossing approaches in the Piedmont (21) and Ridge and Valley (RV) (16) physiographic regions of Virginia. Total rainfall data were recorded daily from five weather stations within Appomattox, Buckingham, Giles, and Montgomery Counties (Figure 2.1). Precipitation amounts were 1,619 and 1,174 mm yr^{-1} with 97% and 91% falling as rainfall for the Piedmont and RV study sites, respectively. Stream crossing approaches were defined as the road area sloping towards a stream crossing. Piedmont sites were selected from seven intensively managed loblolly pine (*Pinus taeda* L.) plantations made available by Mead-Westvaco (now owned by Weyerhaeuser) in Virginia. We instrumented the 21 Piedmont haul road stream crossing approaches in June 2012. Rolling hills with moderate slopes are characteristic of the topography in this region (USDA NRCS 2012). Soil series identified along road approaches were Appomattox-Cullen complex, Chewacla, Codorus-Hatboro complex, Mecklenburg-Poindexter complex, Grassland-Delanco complex, Tatum-Manteo complex, Spears Mountain, and Spears Mountain-Bugley complex (USDA NRCS 2012). Soil erodibility factors (K-values), which can range from 0.02-0.69, were analyzed using composite soil samples and soil erodibility calculation within the Universal Soil Loss Equation manual (Wischmeier and Smith 1978). K-values for approaches in the Piedmont area ranged from 0.14-0.30. Permanent haul roads and stream crossings were constructed or maintained within 5-25 years for continued forest management. All stream crossings were constructed across intermittent or perennial streams and Virginia Department of Forestry (VDOF) BMP guidelines recommended SMZs were to be left adjacent to stream crossings (VDOF 2011). Roadside ditches were present along most approaches in the Piedmont, but all ditches were disconnected from the stream with wing-ditches located at least 7.6 m prior to the stream crossing according to VDOF recommendations. Traffic was low volume primarily from hunt clubs for most Piedmont roads during this study; however, four instrumented road approaches collected sediment data from approximately 1,500–1,600 loaded log truck passes for a one-month period. During this month, approximately 129 ha were harvested; 10–15 cm of #3 gravel was applied to the haul road approaches; and the approaches were widened approximately 0.9 m.

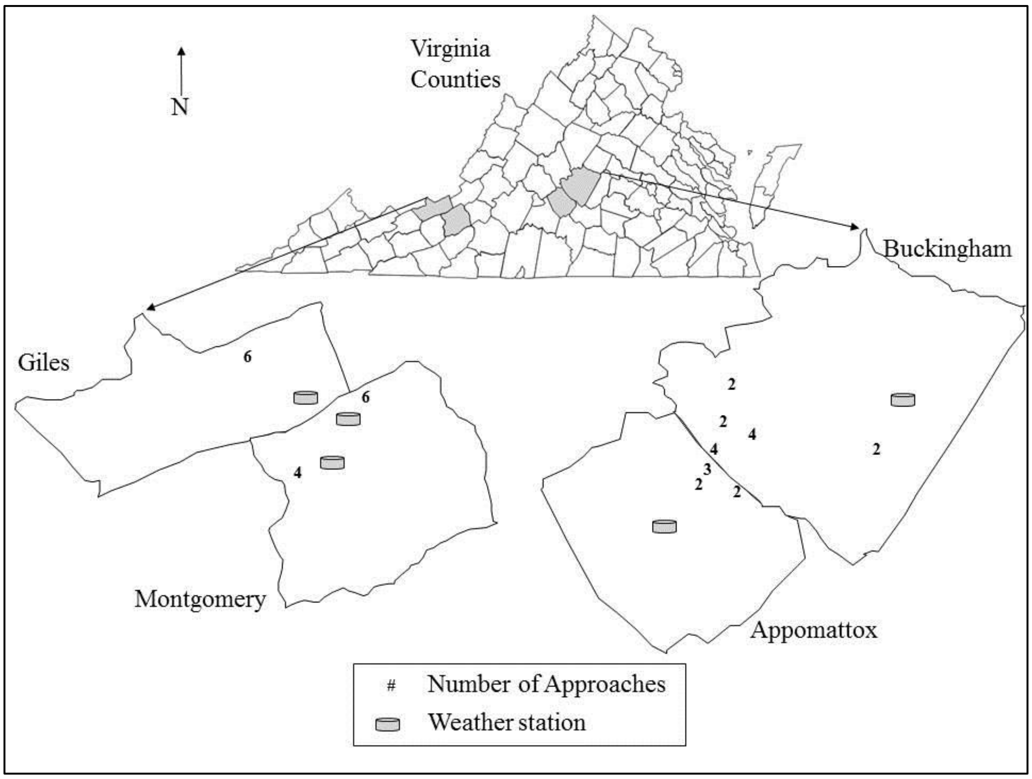

Figure 2.1. General vicinity map of the 37 stream crossing approaches and 5 weather stations in Appomattox, Buckingham, Giles, and Montgomery Counties, Virginia. Note: figure not to scale. Numbers represent the number of stream crossing approaches.

An additional sixteen stream crossing approaches were located in the RV region of Virginia. This region is characterized by broad valleys separated by long linear ridges with low to moderate slopes (USDA NRCS 2012). Sites were selected from three road segments made available by the United States Department of Agriculture (USDA) Forest

Service (FS) and Virginia Tech. Twelve stream crossing approaches were instrumented on two road segments (White Rocks and Turkey Nest roads) on Jefferson National Forest and the remaining four on Virginia Tech school forestland (Fishburn Forest). All haul road stream crossing approaches that were constructed through intermittent or perennial streams for these areas were instrumented. The following soil series were identified along road approaches in this region: Jefferson, Berks and Weikerts, Oriskany, Laidig, and Craigsville (USDA NRCS 2012). K-values for approaches in the RV area ranged from 0.10-0.49. The current designated uses for the FS roads are recreation (no public vehicle access), crop tree release activities, and small firewood cuttings, although the roads were originally constructed for timber harvests. The Virginia Tech school forest road serves access for periodic teaching exercises and maintenance for a municipal water and private cellphone towers. The last known maintenance to road surfaces on both National Forest and Virginia Tech school forest was 2-5 years prior to project installation.

2.3.2 Treatment Installation

Narrow trenches were hand-excavated between 30°-45° angle across the road surface and a thick rubber conveyor belt was buried leaving approximately 15 cm of belt exposed above the road surface (Figure 2.2). The excavated materials were cast downslope of the conveyor belt to minimize the impact of installation on sediment measures. The location of conveyor belts varied depending upon where the lowest elevations nearest the crossing structure were located. Water and sediment generated by overland flow were diverted from the road surface by the belt into an adjacent ditch or roadside silt fence catchment area. This design was used to allow vehicular traffic and approximate sediment delivery. Conveyor belts were installed over a six-month period

(June to November, 2012) and were measured on approximately two month intervals for one year.

Figure 2.2. Rubber conveyor belt diverter used to divert runoff into silt fence catchment areas.

2.3.3 Field Measurements of Sediment Deposits

Within the silt fence sediment collection areas, a series of 10 rebar pins marked the locations for repeated elevation measurements of trapped sediment (Figure 2.3). Additional sediment pins were added as sediment accumulated to better estimate deposited sediment. Elevations behind pins were measured using differential leveling with a total station (Sokkia total station model SET-520, Tokyo, Japan). Elevation gains (m) (sediment deposits) and depositional area (m^2) were recorded during each site visit and multiplied to calculate sediment volumes (m^3). Three bulk density samples were collected using the soil core method (Grossman and Reinsch 2002) for each sediment catchment area and analyzed after one year of sediment accumulation. Sediment volume increases were multiplied by mean bulk density (Mg m^{-3}) to calculate sediment mass (Mg). Sediment masses corresponding to each repeated measure were expressed as mass per unit area (Mg ha^{-1}) by dividing the road surface area (area from the conveyor belt to the nearest water control structure), and later summed over one year to express sediment delivery on an annual basis (Mg ha^{-1} yr^{-1}).

Figure 2.3. Silt fence catchment area with rebar pins marking the location of measurement.

2.3.4 Approach Attributes

Approach characteristics and attributes that have been shown to influence erosion (Swift 1984, Luce and Black 2001, Grace and Zarnoch 2013) were recorded following erosion belt installation. Road prism dimensions (distance from crossing to water control structure, width, slope, and cutslope ratio) were surveyed using a total station. Width of SMZs and depth of gravel were measured with a measuring tape. Bare soil percentages were collected seasonally and quantified by walking a zigzag pattern from the belt to the top of the approach and counting the number of steps where the toe of the boot contacted bare soil (i.e. count of bare soil steps / total number of steps x 100 = percent bare soil) (Brown et al. 2013). Road cover means were determined from the four seasonal measurements to account for the effect of vegetative growth and tree litter fall. Gravel coverage was also quantified in a similar manner and was categorized into bare (0-10%), sparsely graveled (11-50%), and graveled (>50%). Canopy covers were measured seasonally using a spherical densitometer and averaged for data analysis. Surface soil composites were collected from running surfaces and evident sediment source areas of approaches (i.e., bare soil areas along the approach that were not on the running surface [e.g., cutslopes]). Soil samples were analyzed for particle size classes using the hydrometer method (Gee and Or 2002). In addition to quantitative parameters, approach qualitative parameters recorded were road template (insloped, outsloped, or crowned) and road shape (flat, concaved, convex, and s-shaped).

2.3.5 Best Management Practice Audit Scores

BMP audits have been used in research investigations to compare BMP implementation and effectiveness (Sugden et al. 2012, Nolan et al. 2015). Evaluations of stream crossing approach BMPs in this investigation were conducted using a subset of the VDOF's BMP audit questions. BMP audits in Virginia are conducted post-harvest on 240 sites per year by a VDOF water quality specialist (Lakel and Poirot 2014). Audit questions consist of yes or no answers and cover 10 categories of forest operations. For our examination, BMP audit scores were calculated from the percent of 16 audit questions. The subset of questions specifically addressed the stream crossings and approaches for their condition and adequacy and included questions about road attributes, road template, water control, and stream crossing structure (Table 2.1).

Table 2.1. Subcategories included in the BMP audit rating of existing BMP implementation at each stream crossing approach based on Virginia Department of Forestry BMP manual criterion (Virginia Department of Forestry, 2011).

Category	Evaluation criteria
Road attributes	Are grades between 2% and 10% except for necessary deviations?
	Are roads day lighted where needed and feasible?
	Is access being controlled with functional gate?
	Is gravel or vegetation present to protect water bars from erosion?
	Is water being turned out into surrounding landscape with appropriate structures?
	Are turnouts functioning properly?
Road template	Is the road entrenched?
	Does the road template (insloped, outsloped, crowned) shed water from road surface in minimal amounts?
Water control	Are water control structures spaced adequately based on road grade?
	Do water control structures reduce rill formation by redirecting surface runoff from road surface in small amounts?
	Do water control structures redirect surface runoff away from the stream?
Stream crossing	Is a stream crossing location favorable for gentle approaches, stable streambanks, crossing at a 90° angle, and/or avoiding excessive fill?
	Is culvert fill sufficient to withstand expected traffic volumes and loads?
	Is a culvert diameter sufficient for water conveyance during storm events?
	Does the culvert allow fish passage?
	Did the logger minimize SMZ gaps for stream crossings?

2.3.6 Overall Road Quality Ranking

Overall road quality rankings were assigned to approaches by categorizing road gradient, percent bare soil, distance to the nearest water control structure (WCS), and road traffic into highly acceptable (High), acceptable (Standard), and does not meet recommended BMPs (Low) categories (Table 2.2). Our overall road quality rankings differ from the

BMP audit questions in that our rankings were assigned based on measurable approach characteristics, while BMP audit scores evaluated BMP compliance (yes or no). Individual audit category ratings (road gradient, bare soil, distance to the nearest WCS, traffic, and armoring) were used collectively to assign an overall rank. Overall road quality ranks were based on the greatest frequency of subcategory ratings (Table 2.2). For questionable sites with balanced ratings, the final decisions regarding road quality rank were made using BMP audit scores. In Cristan et al. (2016), the overall BMP rates for regulatory, non-regulatory, and quasi-regulatory states were ≥ 90%. Additionally, they reported 89-80% implementation rates for stream crossings. Therefore, audit scores between 100-90, 89-80, and less than 80 were assigned High, Standard, and Low ranks, respectively. Representative photographs of overall road quality ranks for Piedmont and RV regions are presented in Figure 2.4.

Table 2.2. Criteria for overall road quality ranking for approach attributes.

Approach attributes	High	Standard	Low
Road gradient (%)	Less than 5%	6-10%	Greater than 10%
Mean bare soil (%)	Less than 25%	26-50%	51-100%
Distance to WCS* (m)	Less than 15 m	n/a	Greater than 15 m
Traffic frequency	Seldom	Often	Logged
Surface armoring/ hardening	Graveled (> 51%)	Sparsely graveled (10 – 50%)	None (< 10%)
Overall road quality	Greatest frequency of High, Standard, and Low Tie breakers made using BMP audit scores		

*Water control structure

Piedmont

Haul Road Stream Crossings

Ridge and Valley

Haul Road Stream Crossings

Low

n/a

Standard

High

Figure 2.4. Representative photographs of overall road quality rankings for Piedmont and Ridge and Valley stream crossing approaches.

2.3.7 Data Analysis

The dataset consisted of independent variables characterizing approach attributes and two dependent variables (mass of delivered sediment [kg yr^{-1}] and sediment delivery rate [Mg ha^{-1} yr^{-1}]). The dependent data had unequal variances (determined with Levene's test) and non-normal distributions (Shapiro-Wilk test) (Zar 2010). Thus, median differences in dependent variables by overall road quality rankings and regions were tested using nonparametric Kruskal-Wallis and Wilcoxon tests. Significant differences were separated using a post-hoc Wilcoxon each pair test or Steel-Dwass test using an $\alpha = 0.05$. Independent variables characterizing approach attributes were tested using ANOVA methods. Significant differences were separated using Tukey-Kramer HSD tests (Zar 2010). Statistical analyses were performed using JMP (SAS Institute Inc. 2012) statistical software program.

2.4 Results and Discussion

2.4.1 Road Attributes by Region and Road Quality Rank

All sites received several precipitation events of sufficient intensities to cause soil erosion if BMPs were not adequate. Instrumented approaches represented road attributes that ranged widely and covered a spectrum of BMP applications. Piedmont sites primarily had crowned (48%) or insloped (38%) approach templates with flat (38%) or concave (43%) approach shapes. Approach templates on RV sites were insloped (56%) or outsloped (44%), with the majority of approach shapes being concave (44%) (Table 2.3). Most approaches received at least one application of gravel (90%); however, of those approaches 22% had less than 50% coverage (sparsely graveled) (Table 2.3). Soil texture of cutslope and ditch areas in the Piedmont tended to have greater clay content, while RV cutslopes and ditches had greater sand and silt contents

(Table 2.3). Soil texture on the approach surfaces was variable, but followed a similar pattern of clay content as that of the cutslopes and ditches (Table 2.3).

Table 2.3. Number and percentage of instrumented haul road stream crossing approach attributes in the Piedmont and Ridge and Valley region of Virginia.

Approach category	Approach attributes	Piedmont		Ridge and Valley		Total	
		n	%	n	%	n	%
Road template	Crowned	10	48	0	0	10	27
	Insloped	8	38	9	56	17	46
	Outsloped	3	14	7	44	10	27
Slope shape	Flat	8	38	3	19	11	30
	Concave	9	43	7	44	16	43
	Convex	2	10	2	13	4	11
	S-shaped	2	10	4	25	6	16
Surface armoring/ hardening	Bare	6	29	0	0	4	16
	Sparsely graveled	7	33	6	38	8	35
	Graveled	8	38	10	63	25	49
Soil texture of sediment sources (e.g. cutslopes and adjacent non-road areas)	Clay	6	29	0	0	6	16
	Clay loam	7	33	1	6	8	22
	Sandy clay loam	3	14	0	0	3	8
	Loam	2	10	3	19	5	14
	Fine sandy loam	2	10	10	63	12	32
	Coarse sandy loam	0	0	1	6	1	3
	Sandy loam	1	5	0	0	1	3
	Loamy coarse sand	0	0	1	6	1	3
Soil texture of approach surfaces (e.g. road surface)	Clay	3	14	0	0	3	8
	Clay loam	4	19	0	0	4	11
	Sandy clay loam	2	10	0	0	2	5
	Loam	3	14	1	6	4	11
	Fine sandy loam	4	19	3	19	7	19
	Coarse sandy loam	1	5	7	44	8	22
	Sandy loam	4	19	4	25	8	22
	Loamy sand	0	0	1	6	1	3

On average, Piedmont road approaches had significantly wider approach widths, greater bare soil levels, and longer distances to the nearest water control structure than sites in the RV region ($P \leq 0.1000$, Table 2.4). Approaches in the Piedmont had mean bare soil levels of 52%, while RV approaches had only 19%. Mean distance to the nearest water control structure was shorter on Piedmont approaches (45.2 m) than on RV approaches (88.2 m). It should be noted that these road attributes could be explained by differences in land management objectives and prior land use history. Currently, Piedmont approaches provide access for intensive harvesting, which potentially requires more water control to abate erosion, wider roads for trucking, and judicious applications of gravel to minimize costs. Greater bare soil on approaches in the Piedmont may have also been a function of greater traffic frequency. Many roads within this study and the Piedmont region overall are not designed, rather they are reopened legacy roads used to minimize the cost of constructing a new road (Brown et al. 2013). Brown et al. (2013) noted that upgrading legacy roads to modern road standards might require significant design improvements and application of modern BMPs to reduce soil erosion rates. Conversely, RV region roads in this study were designed and management objectives required greater levels of gravel application.

Table 2.4. Descriptive statistics for the Piedmont and Ridge and Valley regions categorized by road gradient, mean bare soil, distance to water control structure (WCS), and road width. P-values are displayed above each metric. Different letters within each column represent significant differences between regions at $\alpha \leq 0.10$ based on parametric T-tests.

		$P \leq 0.6378$; Road gradient (%)	$P \leq 0.0001$; Mean bare soil (%)	$P \leq 0.0843$; Distance to WCS (m)	$P \leq 0.0130$; Road width (m)
Piedmont	Max.	16.0	93.8	129.5	6.4
	Mean	6.6 a	52.0 b	45.2 a	3.8 b
	Median	6.0	50.0	30.5	3.7
	Min.	2.0	8.0	12.8	2.1
	S.D.	3.5	23.9	35.1	1.1
	S.E.	0.8	5.2	7.7	0.2
Ridge & Valley	Max.	13.0	42.5	426.7	3.7
	Mean	7.2 a	19.3 a	88.2 b	3.0 a
	Median	7.0	14.6	61.7	3.0
	Min.	1.0	1.8	6.1	2.4
	S.D.	3.8	12.5	104.0	0.4
	S.E.	0.9	3.1	26.0	0.1

Mean BMP audit scores were similar between Piedmont and RV region sites (77% and 78%, respectively). However, stream crossings in the RV received reduced BMP audit scores because they lacked water control structures and roadside ditches delivered runoff directly to streams. Conversely, in the Piedmont, no sites had ditches directly connected to streams, but many culverts were improperly sized, perched, or required additional maintenance for water turnouts.

Tables 2.5 and 2.6 show summary statistics for Piedmont and RV regions, respectively, by overall road quality rank. Mean bare soil was significantly different (P ≤

0.0105) within each region for overall road quality rank, while road gradient, distance to the nearest water control structure, and road width had similar values (Tables 2.5 and 2.6). Both regions collectively displayed significant difference among road quality rankings for mean bare soil ($P \leq 0.0001$), but no significant differences among road quality rankings existed for road gradient, distance to the nearest water control structure, and road width ($P \geq 0.1000$. Table 2.7).

Table 2.5. Descriptive statistics for Piedmont region approaches categorized by overall road quality for road gradient, mean bare soil, distance to water control structure (WCS), and road width. P-values are displayed above each road characteristic. Means followed by letters are significantly different at $\alpha \leq 0.10$ based on the Tukey-Kramer multiple comparison tests.

Road attributes		Overall road quality	N	Max.	Mean		Median	Min.	S.D.	S.E.
$P \leq 0.3402$; Road gradient	(%)	High	2	12.0	7.0	a	7.0	2.0	7.1	5.0
		Std.	12	8.0	5.7	a	5.5	3.0	1.9	0.5
		Low	7	16.0	8.1	a	7.0	4.0	4.6	1.7
$P \leq 0.0001$; Mean bare soil	(%)	High	2	19.3	13.6	a	13.6	8.0	8.0	5.6
		Std.	12	67.5	44.2	b	44.4	26.3	12.9	3.7
		Low	7	93.8	76.3	c	80.0	52.5	17.0	6.4
$P \leq 0.5842$; Distance to WCS	(m)	High	2	23.7	23.3	a	23.3	22.9	0.6	0.5
		Std.	12	129.5	50.9	a	29.0	12.8	43.6	12.6
		Low	7	80.8	41.7	a	39.6	24.4	19.5	7.4
$P \leq 0.1151$; Road width	(m)	High	2	5.5	5.3	a	5.3	5.2	0.2	0.2
		Std.	12	6.4	3.8	a	3.7	2.3	1.2	0.4
		Low	7	4.6	3.5	a	3.7	2.1	0.7	0.3

Table 2.6. Descriptive statistics for Ridge and Valley region approaches categorized by overall road quality for road gradient, mean bare soil, distance to water control structure (WCS), and road width. P-values are displayed above each road characteristic. Means followed by letters are significantly different at α ≤ 0.10 based on Students T-tests.

Road attributes		Overall road quality	N	Max.	Mean		Median	Min.	S.D.	S.E.
P ≤ 0.3448; Road gradient	(%)	High	7	12.0	6.1	a	7.0	1.0	3.6	1.3
		Std.	9	13.0	8.0	a	7.0	1.0	3.9	1.3
P ≤ 0.0105; Mean bare soil	(%)	High	7	18.8	10.8	a	13.8	1.8	6.3	2.4
		Std.	9	42.5	26.0	b	26.3	8.8	12.3	4.1
P ≤ 0.3824; Distance to WCS	(m)	High	7	137.0	61.5	a	47.2	8.2	45.9	17.3
		Std.	9	426.7	109.1	a	64.0	6.1	132.6	211.0
P ≤ 0.2234; Road width	(m)	High	7	3.7	3.2	a	3.0	2.4	0.5	0.2
		Std.	9	3.4	2.9	a	3.0	2.7	0.2	0.7

Table 2.7. Descriptive statistics for all approaches categorized by overall Road quality rankings for measured road gradient, mean bare soil, distance to water control structure (WCS), and road width. Numbers followed by letters are significantly different at α ≤ 0.10 based on the Tukey-Kramer multiple comparison tests.

Road attributes		Overall road quality	N	Max.	Mean		Median	Min.	S.D.	S.E.
P ≤ 0.5723; Road gradient	(%)	High	9	12.0	6.3	a	7.0	1.0	4.0	1.3
		Std.	21	13.0	6.7	a	7.0	1.0	3.1	0.7
		Low	7	16.0	8.1	a	7.0	4.0	4.6	1.7
P ≤ 0.0001; Mean bare soil	(%)	High	9	19.3	11.4	a	13.8	1.8	6.3	2.1
		Std.	21	67.5	36.4	b	37.5	8.8	15.4	3.4
		Low	7	93.8	76.3	c	80.0	52.5	17.0	6.4
P ≤ 0.5286; Distance to WCS	(m)	High	9	137.0	53.0	a	39.6	8.2	43.2	14.4
		Std.	21	427.0	75.8	a	36.6	6.1	94.6	20.6
		Low	7	80.8	41.7	a	39.6	24.4	19.5	7.4
P ≤ 0.8188; Road width	(m)	High	9	5.5	3.7	a	3.7	2.4	1.0	0.4
		Std.	21	6.4	3.4	a	3.0	2.1	1.0	0.2
		Low	7	4.6	3.5	a	3.7	2.1	0.7	0.3

2.4.2 Sediment Delivery Range from Haul Road Stream Crossing Approaches

Mean sediment delivery mass from all approaches ranged from <0.1 to 2719.1 kg during the study (Table 2.8). Despite direct roadside ditch connections, median sediment delivery mass was significantly less ($P \leq 0.0252$) in the RV (10.3 kg) than Piedmont region (43.8 kg) (Table 2.8). Sediment delivery rate also followed the same pattern ($P \leq 0.0142$, Table 2.8). Collectively, all sediment traps collected a total of 7.24 Mg of sediment over the one-year period. Subjective evaluation of the approaches indicates that two approaches accounted for 59% of the total sediment mass and the top five approaches accounted for 82%. The approach that generated the most sediment mass (2219.1 kg) was located in the Piedmont on a 13% slope, with a mean bare soil of 91% (Table 2.9). Clay soil textures were determined for both road and cutslope surface soils. The nearest water control structure was a culvert cross drain (81 m from the stream crossing) and did not properly disperse effluent into adjacent forest cover. Instead, runoff was directed back onto the travel-way and concentrated into a shallow rut (< 8 cm) creating an erosion rill. This approach had minimal to no gravel application, resulting in 91% mean bare soil (Figure 2.5). Despite producing the most sediment mass, this approach did not have the highest sediment delivery rate (73.6 Mg ha^{-1} yr^{-1}, Table 2.9). The second greatest amount of delivered sediment mass (1550.2 kg) originated from an approach also located in the Piedmont with a 7% slope and mean bare soil of 80%. This approach exhibited rutting and lacked proper dispersal of storm runoff onto the adjacent forest floor. Rutting was 8–12 cm deep and water turnouts located 25 m from the stream crossing failed due to inadequate maintenance and excessive sediment deposition. This approach concentrated runoff into road ruts, which initiated rill erosion. Due to the readily eroded soil, improper water control maintenance, and large area contributing to erosion (227 m^2) this approach had the greatest sediment delivery rate (290.7 Mg ha^{-1} yr^{-1}, Table 2.9).

Table 2.8. Sediment delivery estimates and approach attributes for 37 stream crossing approaches categorized by overall road quality and sorted from greatest to least by measured sediment mass.

Overall road quality	Measured sediment mass	Measured sediment	Slope	Mean bare soil	Distance to WCS*	Traffic	BMP audit score	Road surface soil texture
	(kg)	(Mg ha^{-1} yr^{-1})	(%)	(%)	(m)			
Low	2719.1	73.6	13.0	91.3	80.8	Often	50	Clay
	1550.2	290.7	7.0	80.0	25.0	Often	69	Clay Loam
	768.0	43.1	7.0	65.0	48.8	Logged	69	Fine Sandy Loam
	451.5	28.9	4.0	60.0	42.7	Logged	88	Fine Sandy Loam
	222.6	21.8	16.0	52.5	30.5	Often	13	Loam
	86.0	6.5	5.0	93.8	39.6	Often	75	Clay
	3.0	0.3	5.0	91.3	24.4	Often	81	Clay
Std.	427.4	115.0	8.0	55.0	15.2	Often	25	Clay Loam
	174.2	93.7	7.0	37.5	6.1	Often	81	Fine Sandy Loam
	171.9	12.9	4.0	60.0	21.3	Often	69	Loam
	150.2	9.0	6.0	67.5	30.5	Often	75	Loam
	56.9	2.9	9.0	28.8	59.4	Seldom	79	Coarse Sandy Loam
	46.9	0.8	12.0	25.0	201.2	Often	80	Loam
	43.8	3.9	8.0	26.3	24.4	Often	100	Sandy Loam
	37.9	3.8	1.0	11.3	36.6	Seldom	79	Sandy Loam
	28.5	10.2	12.0	42.5	9.1	Often	75	Sandy Loam
	24.5	5.7	7.0	26.3	12.8	Often	94	Sandy Clay Loam
	19.3	0.6	5.0	35.0	103.6	Often	81	Clay Loam
	18.8	4.8	4.0	50.0	18.3	Often	88	Clay Loam
	18.3	0.1	13.0	38.8	426.7	Often	73	Sandy Loam
	15.6	1.4	7.0	34.8	27.4	Often	100	Sandy Loam
	11.5	0.4	7.0	8.8	97.5	Seldom	79	Coarse Sandy Loam
	11.0	0.9	3.0	46.3	32.0	Logged	100	Fine Sandy Loam
	9.4	0.2	3.0	48.8	118.9	Logged	100	Sandy Loam
	9.0	0.5	6.0	15.0	64.0	Seldom	79	Coarse Sandy Loam
	6.0	0.2	5.0	42.5	129.5	Often	81	Fine Sandy Loam
	5.4	0.2	5.0	26.3	80.8	Seldom	79	Loamy Sand
	2.5	0.1	8.0	39.8	76.2	Often	63	Sandy Clay Loam
High	91.5	3.8	5.0	10.5	97.5	Seldom	79	Coarse Sandy Loam
	47.4	3.6	2.0	19.3	23.8	Often	94	Coarse Sandy Loam
	8.8	0.6	7.0	1.8	39.6	Seldom	79	Fine Sandy Loam
	3.4	0.3	12.0	8.0	22.9	Often	93	Sandy Loam
	1.4	0.1	8.0	18.8	81.0	Seldom	79	Sandy Loam
	0.8	0.3	1.0	14.0	8.2	Seldom	79	Coarse Sandy Loam
	0.2	<0.1	7.0	13.8	137.0	Seldom	79	Coarse Sandy Loam
	<0.1	<0.1	12.0	2.8	47.2	Seldom	71	Coarse Sandy Loam
	<0.1	<0.1	3.0	14.3	19.8	Seldom	79	Fine Sandy Loam

*Water control structure

Table 2.9. Descriptive statistics for all approaches categorized by overall road quality for measured sediment delivery and sediment delivery rates. Nonparametric Kruskal-Wallis tests and Wilcoxon tests were used to compare Road quality rankings. Numbers followed by letters are significantly different at $\alpha \leq 0.10$ based on the Steel-Dwass all pairs median separation tests.

	Measured sediment delivery	Overall road quality	N	Max.	Mean	Median	Min.	S.D.	S.E.
Piedmont	P-value ≤ 0.0553;	High	2	47.4	25.4	25.4 b	3.4	31.1	22.0
	Sediment mass	Std.	12	427.4	75.0	19.1 a	2.5	124.7	36.0
	(kg)	Low	7	2719.1	829.6	451.5 b	3.0	986.6	372.9
	P-value ≤ 0.0499;	*High*	*2*	*3.6*	*2.0*	*2.0 b*	*0.3*	*2.4*	*1.7*
	Sediment rate	*Std.*	*12*	*115.0*	*12.9*	*2.7 a*	*0.1*	*32.4*	*9.4*
	(Mg ha^{-1} yr^{-1})	*Low*	*7*	*290.7*	*66.4*	*28.9 b*	*0.3*	*101.9*	*38.5*
Ridge & Valley	P-value ≤ 0.0172;	High	7	91.4	14.7	0.8 a	<0.1	34.0	12.9
	Sediment mass	Std.	9	174.2	43.2	28.5 b	5.4	52.2	17.4
	(kg)	Low	0	-	-	-	-	-	-
	P-value ≤ 0.0500;	*High*	*7*	*3.8*	*0.7*	*0.1 a*	*<0.01*	*1.4*	*0.5*
	Sediment rate	*Std.*	*9*	*93.7*	*12.5*	*0.8 b*	*0.1*	*30.6*	*10.2*
	(Mg ha^{-1} yr^{-1})	*Low*	*0*	-	-	-	-	-	-
All	P-value ≤ 0.0011;	High	9	91.4	17.1	1.4 a	<0.1	31.8	10.6
	Sediment mass	Std.	21	427.4	61.4	19.3 b	2.5	99.5	21.7
	(kg)	Low	7	2719.1	828.6	451.5 c	3.0	986.6	372.9
	P-value ≤ 0.0017;	*High*	*9*	*3.8*	*1.0*	*0.3 a*	*<0.1*	*1.6*	*0.5*
	Sediment rate	*Std.*	*21*	*115.0*	*12.7*	*1.4 b*	*0.1*	*30.9*	*6.7*
	(Mg ha^{-1} yr^{-1})	*Low*	*7*	*290.7*	*66.4*	*28.9 c*	*0.3*	*101.8*	*38.5*

Figure 2.5. Photographs of the most erosive approach displaying an 8 cm deep erosion rill.

Sediment delivery of the two most erosive approaches in this study is comparable to the highest erosion rates noted for legacy road approaches and skid trails within the Piedmont (Sawyers et al. 2012, Wade et al. 2012a,b, Brown et al. 2013). In Brown et al. (2013) study, the most eroded approach had > 95% bare soil, an approach length (130 m) that exceeded BMP guidelines, but a 4% slope. Despite the gentle 4% slope, sediment delivery reached 287 Mg ha^{-1} yr^{-1} or approximately 11.1 Mg yr^{-1}. In a bladed skid trail study, Wade et al. (2012a,b) measured erosion from newly closed trails with varying levels of cover BMPs (grass seed only, grass seed and mulch, hardwood slash, and pine slash) on 10 to 20% slopes. Not surprisingly, their control treatments, bare soil with only waterbars, produced the greatest mean annual soil erosion (138 Mg ha^{-1} yr^{-1} or 0.63 Mg yr^{-1}). Sawyers et al. (2012) measured erosion from recently closed overland skid trails in the Piedmont region with treatments that included bare soil, grass seed only, grass seed and mulch, hardwood slash, and pine slash. The experiment also found bare soil treatments produced the greatest mean annual soil erosion (24.24 Mg ha^{-1} yr^{-1} or 0.14 Mg yr^{-1}). Both skid trails studies examined fixed 15.2 m skid trail lengths, while the current and legacy road study evaluated varying road lengths. Researchers from these studies demonstrated that roads with contemporary BMPs (proper water control spacing and application of soil stabilization methods) significantly reduced soil erosion and sediment delivery. Sediment delivery rates and sediment mass from the two most eroded approaches in the current study also exemplify the importance of maintaining water control structures, applying gravel or grass seed to minimize bare soil, and proper road location (i.e., avoiding road placement on excessive slopes), especially for stream crossings.

Seventy-five percent of approaches monitored in this study generated less than 100 kg or 10 Mg ha^{-1} yr^{-1} of sediment (Table 2.9). These approach slopes ranged from 1 to 13%; bare soil ranged from 2 to 94%; and distances to nearest water control structure ranged from 8.2 to 427.0 m. Such wide spectrum of road attributes for the approaches that eroded less than 100 kg of sediment indicate that contemporary BMPs can offset problematic road attributes and reduce erosion and sediment delivery. For example, a lower standard road with a higher level of BMP usage could produce sediment levels more similar to a higher standard road with standard BMP implementation. Nolan et al. (2015) compared BMP implementation levels on forest roads and found a clear linkage between BMP use and potential erosion. Furthermore, the variation of BMP applications supports site specific BMPs assigned by professional forest managers. For example, gentle approaches may require less surface coverage than steep approaches when erosion contributing areas are equivalent. All High ranked approaches produced less than 4 Mg ha^{-1} yr^{-1} and had mean bare soil less than 20% (Table 2.9).

2.4.3 Sediment Delivery by Road Quality Rankings

Eighty-one percent (30 out of 37) of monitored approaches measured were ranked as standard or High road quality (Table 2.9). Mean sediment delivery rates for Low rated approaches were approximately 48 times greater than High rated and 3.5 times greater than Standard rated approaches. Sixty-eight percent of Standard ranked approaches produced sediment delivery rates less than 2 Mg ha^{-1} yr^{-1} and 92% less than about 10 Mg ha^{-1} yr^{-1} (Table 2.9). The highest two sediment producing approaches within the Standard rank were just inside of the set parameters for road quality categorization. During sediment surveys, it was noted that the nearest water control structures for these two approaches (15.2 and 6.1 m) were improperly functioning.

Standard quality approaches in the Piedmont had significantly less median eroded sediment mass than High and Low quality approaches ($P \leq 0.0553$, Table 2.10). This unexpected result may be explained by the number of Piedmont approaches ranked as

High (n = 2). Road quality ranking in the RV indicated significant differences between High and Standard ranks for delivered sediment mass ($P \leq 0.0172$, Table 2.10). Road quality ranks were found to have significantly different amounts of trapped sediment ($P \leq 0.0011$). A post-hoc Steel-Dwass test showed that the median sediment mass for Low (451.5 kg) was significantly greater than Standard (19.3 kg) and High (1.4 kg). Additionally, Standard ranked approaches were significant greater than High.

Table 2.10. Descriptive statistics for the Piedmont and Ridge and Valley regions categorized by sediment delivery mass and sediment delivery rate. P-values are displayed above each metric. Different letters within each column represent significant differences between regions at $\alpha \leq 0.10$ based on non-parametric Wilcoxon tests.

	Region	Max.	Mean	Median	Min.	S.D.	S.E.
$P \leq 0.0252$; Sediment mass (kg)	Piedmont	2719.1	321.5	43.8 b	2.5	660.2	144.1
	Ridge & Valley	174.2	30.7	10.3 a	<0.1	46.2	11.5
$P \leq 0.0142$; Sediment rate (Mg ha^{-1} yr^{-1})	*Piedmont*	290.7	29.7	4.8 b	0.1	66.4	14.5
	Ridge & Valley	93.7	7.3	0.5 a	<0.1	23.2	5.8

2.5 Conclusions

Collectively, road characteristics and BMPs control variation in sediment yield. Low quality roads (i.e. inadequately spaced, constructed, or maintained water control structure, inadequately stabilized surfaces, or unrestricted road access) and unfavorable road characteristics (i.e. steep and/or excessive road prism dimensions and poor road location and drainage) have a greater erosion potential. Most stream crossing approaches monitored in this study appropriately applied BMPs to reduce sediment delivery. Stream crossing approaches in the RV were directly connected to streams through ditches, but sediment delivery remained relatively low due to the lack of traffic, greater canopy cover, and greater soil cover on running surfaces and cutslopes. The majority of the RV roads were designed roads having good locations and grades. Piedmont stream approaches tended to have greater sediment delivery because of traffic frequency, bare soil, and inadequately maintained water control structures. Sediment delivery varied according to site specific conditions, but tended to increase on approaches with bare soil exceeding 50% and improperly installed or maintained water control structures. Piedmont roads were often poorly designed legacy roads where BMPs were used to overcome the poor locations.

Stream crossing approaches with faulty water control introduced some subjectivity in defining erosion/contributing area. Estimating the eroding and water contributing areas to a specific point is a function of rainfall and minor topographic and road characteristics (Montgomery 1994). Poor road designs, construction, and maintenance further increase erosion and sediment delivery potential by increasing hydrologic networks connected to streams (Megahan et al. 2001, Clinton and Vose 2003). Failure to define contributing area may lead to less optimal management decisions. If the objective is to reduce total sediment reaching streams, managers should also consider total sediment mass (loading) and potential increases in contributing area caused by water control failures. For example, our greatest sediment mass measure (2.72 Mg) was the fourth highest sediment delivery rate (73 Mg ha^{-1} yr^{-1}). Additional implementation of BMPs would have a greater effect on the most erosion prone stream crossings and indicates that manager should target erosion prone crossings in order to maximize water quality protection with limited resources.

Approaches ranked as Standard and High clearly reduced sediment delivery relative to stream approaches characterized as Low road quality. Overall, this study demonstrates the effectiveness of BMPs for forest roads; however, implementing BMPs on roads cannot overcome the negative consequences of inadequate or improperly implemented water control structures on the road surface. An approach that handles an oversupply of water, is too steep, or has other unfavorable conditions that accelerate erosion should be targeted for additional BMP implementation and/or shorter routine maintenance schedules. In contrast, road approaches having favorable characteristics, such as low grades, gravel cover, and suitable water runoff loads may not require additional BMP implementation and may have more time between scheduled road maintenance periods.

2.6 Acknowledgements

Funding for this work was provided in part by the National Council for Air and Stream Improvement (NCASI), Inc. and McIntire-Stennis program of the National Institute of Food and Agriculture. MeadWestvaco Corporation, Plum Creek Timber Company, Virginia Tech Department of Forest Resources and Environmental Conservation, and United States Forest Service provided study sites.

2.7 Literature Cited

Anderson, C.J. and B.G. Lockaby. 2011. Research gaps related to forest management and stream sediment in the United States. *Environ. Manage.* 47(2), 303-313.

Aust, W.M., Bolding, M.C., and Barrett, S.M., 2015. Best management practices for low-volume forest roads in the Piedmont region. *Transp. Res. Record.* 2472: 51-55.

Aust, W.M., M.B. Carroll, M.C. Bolding, and C.A. Dolloff. 2011. Operational forest stream crossing effects on water quality in the Virginia Piedmont. *South. J. Appl. For.* 35(3): 123-130.

Aust, W.M., R. Visser, T. Gallagher, and E.T. Roberts. 2003. Cost of six different stream crossing options in the Appalachian area. *South. J. Appl. For.* 27(1): 66-70.

Brown, K.R., Aust, W.M., and McGurie, K.J., 2013. Sediment delivery from bare and graveled forest road stream crossing approaches in the Virginia Piedmont. *For. Ecol. Manage.* 310, 836-846. doi: 10.1016/j.foreco.2013.09.031.

Brown, K.R., K.J. McGuire, W.M. Aust, W.C. Hession, and C.A. Dolloff. 2015. The effect of increasing gravel cover on forest roads for reduced sediment delivery to stream crossings. *Hydrol. Process.* 29(6): 1129-1140.

Chappell, N.A., I. Douglas, J.M. Hanapi, and W. Tych. 2004. Sources of suspended sediment within a tropical catchment recovering from selective logging. *Hydrol. Process.* 18(4): 685-701.

Clinton, B.D. and J.M. Vose. 2003. Differences in surface water quality draining four road surface types in the southern Appalachians. *South. J. Appl. For.* 27(2): 100-106.

Conrad, J.L. IV., W.S. Ford, M.C. Groover, M.C. Bolding, and W.M. Aust. 2012. Virginia Tech forest road and bladed skid trail cost estimation method. *South. J. Appl. For.* 36(1): 26-32.

Cristan, R., W.M. Aust, M.C. Bolding, S.M. Barrett, J.F. Munsell, and E.B. Schilling. 2016. Effectiveness of forestry best management practices in the United States: literature review. *For. Ecol. Manage*. 360: 133-151.

Croke, J., S. Mockler, P. Hairsine, and P. Fogarty. 2006. Relative contributions of runoff and sediment from sources within a road prism and implications for total sediment delivery. *Earth Surf. Proc. Land.* 31(4): 457-468.

Croke, J.C. and P.B. Hairsine. 2006. Sediment delivery in managed forests: a review. *Environ. Rev.* 14(1): 59-87.

Edwards, P.J. and K.W.J. Williard. 2010. Efficiencies of forestry best management practices for reducing sediment and nutrient losses in the eastern United States. *J. For.* 108(5): 245-249.

Ellefson, P.V., M.A. Kilgore, and M.J. Phillips. 2001. Monitoring compliance with BMPs: the experience of state forestry agencies. *J. For.* 99(1): 11-17.

Forman, R.T. and L.E. Alexander. 1998. Roads and their major ecological effects. *Annual Rev. Ecol. Syst.* 29(1): 207-231.

Fransen, P.J.B., C.J. Phillips, and B.D. Fahey. 2001. Forest road erosion in New Zealand: overview. *Earth Surf. Proc. Land.* 26(2): 165-174.

Gee, G.W. and D. Or. 2002. Particle-size analysis. In Methods of soil analysis. Part 4. Physical methods. 278-284.

Grace, J.M. 2005. Forest operations and water quality in the south. *Trans. ASAE.* 48(2): 871-880.

Grace, J.M., and S.J. Zarnoch. 2013. Influence of forest road buffer zones on sediment transport in the Southern Appalachian Region. *In*: Proceedings of the 15th Biennial Southern Silvicultural Research Conference. E-Gen. Tech. Rep. SRS-GTR-175. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. pp. 487-493.

Grossman, R.B. and T.G. Reinsch. 2002. Methods of soil analysis: bulk density and linear extensibility, core method. *Soil Sci. Soc. Amer. Book Series* 5: 207-209.

Henley, W.F., M.A. Patterson, R.J. Neves, and A.D. Lemly. 2000. Effects of sedimentation and turbidity on lotic food webs: a concise review for natural resource managers. *Reviews Fish. Sci.* 8(2): 125-139.

Ice, G.G. 2004. History of innovative best management practice development and its role in addressing water quality limited waterbodies. *J. Environ. Eng.* 130(6): 684-689.

Ice, G.G., E. Schilling, and J. Vowell. 2010. Trends in forestry best management practices implementation. *J. For.* 108(6): 267-273.

Jordán, A. and L. Martínez-Zavala. 2008. Soil loss and runoff rates on unpaved forest roads in southern Spain after simulated rainfall. *For. Ecol. Manage.* 255: 913-919.

Kochenderfer, J.N. and J.D. Helvey. 1987. Using gravel to reduce soil losses from minimum-standard forest roads. *J. Soil Water Conserv.* 42(1): 46-50.

Kochenderfer, J.N., G.W. Wendel, and H.C. Smith. 1984. Cost of and soil loss on “Minimum-standard” forest truck roads constructed in the central Applachians. US Department of Agriculture, Forest Service, Northeastern Forest Experiment Station, Research Paper NE-544.

Kreutzweiser, D.P., S.S. Chapell, and K.P. Good, K.P. 2005. Effects of fine sediment inputs from logging road on stream insect communities: a large-scale experimental approach in a Canadian headwater stream. *Aquat. Ecol.* 39(1): 55-66.

Lakel, W.A. III and M. Poirot. 2014. Silvicultural best management practices implementation monitoring for Virginia-2013. Virginia Department of Forestry, Charlottesville, VA. 10p.

Lane, P.N.J. and G.J. Sheridan. 2002. Impact of an unsealed forest road stream crossing: water quality and sediment sources. *Hydrol. Process.* 16(13): 2599-2612.

Megahan, W.F. and G.L. Ketcheson. 1996. Predicting downslope travel of granitic sediments from forest roads in Idaho. *Water Res. Bulletin.* 32(2): 371-382.

Megahan, W.F., M. Wilson, and S.B. Monsen. 2001. Sediment production from granitic cutslopes on forest roads in Idaho, USA. *Earth Surf. Process. Land.* 26(2): 153-163.

Montgomery, D.R. 1994. Road surface drainage, channel initiation, and slope instability. *Water Resour. Res.* 30(6): 1925-1932.

Morris, B.C., M.C. Bolding, W.M. Aust, K.J. McGuire, E.B. Schilling, and J. Sullivan. 2016. Differing levels of forestry best management practices at stream crossing structures affect sediment delivery and installation costs. *Water.* 8(3): 92.

Nolan, L., W.M. Aust, S.M. Barrett, M.C. Bolding, K.B. Brown, and K.J. McGuire. 2015. Estimating costs and effectiveness of upgrades in forestry best management practices for stream crossings. *Water*.7(12): 6946-6966.

North Carolina Forest Service, 2014. *A guide for forest access road construction and maintenance in the southern appalachian mountains.* Publication number WQ-02-14.

Ryan, P.A. 1991. Environmental effects of sediment on New Zealand streams: a review. *New Zeal. J. Mar. Fresh.* 25(2): 207-221.

SAS Institue, Inc., 2012. Cary, NC: SAS Institute Inc.

Sawyers, B., M.C. Bolding, W.M. Aust, and W.A. III Lakel. 2012. Effectiveness and implementation costs of overland skid trail closure techniques in the Virginia Piedmont. *J. Soil Water Conserv.* 67(4): 300-310.

Shepard, J.P. 2006. Water quality protection in bioenergy production: the US system of forestry best management practices. *Biomass Bioener.* 30(4): 378-384.

Sheridan, G.J. and P.J. Noske. 2007. A quantitative study of sediment delivery and stream pollution from different forest road types. *Hydrol. Process.* 21(3): 387-398.

Sidle, R.C., S. Sasaki, M. Otsuki, S. Noguchi, and A.R. Nik. 2004. Sediment pathways in a tropical forest: effects of logging roads and skid trails. *Hydrol. Process.* 18(4): 703-720.

Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. Web Soil Survey. Available online at http://websoilsurvey.nrcs.usda.gov/. Accessed July 2012.

Stuart, G.W. and P.J. Edwards. 2006. Concepts about forest and water. *North. J. Appl. For.* 23(1): 11-19.

Sugden, B.D., R. Ethridge, G. Mathieus, P.E.W. Heffernan, G. Frank, and G. Sanders. 2012. Montana's forestry best management practice program: 20 years of continuous improvement. *J. For.* 110(6): 328-336.

Virginia Department of Forestry, 2011. *Virginia's forestry best management practices for water quality: Technical manual 2011.* Charlottesville, Virginia. 216p.

Walbridge, T.A., Jr., 1997. *The location of forest roads.* Forestry Department, Virginia Tech, Blacksburg, VA. 91p.

Wear, L.R., W.M. Aust, M.C. Bolding, B.D. Strahm, and C.A. Dolloff. 2013. Effectiveness of best management practices for sediment reduction at operational forest stream crossings. *For. Ecol. Manage.* 289: 551-561.

Wischmeier, W.H. and D.D. Smith. 1978. *Predicting rainfall erosion losses- A guide to conservation planning.* 62 p.

Wood, P.J. and P.D. Armitage. 1997. Biological effects of fine sediment in the lotic environment. *Environ. Manage.* 21(2): 203-217.

Zar, J.H., 2010. *Biostatistical Analysis*. Pearson-Prentice Hall, Upper Saddle River, NJ.

Ziegler, A.D., J.N. Negishi, R.C. Sidle, T. Gomi, S. Noguchi, A.R. Nik. 2007. Persistence of road runoff generation in a logged catchment in Peninsular Malaysia. *Earth Surf. Proc. Land.* 32(13): 1947-1970.

3.0 Comparing Sediment Trap Data with Erosion Models for Evaluation of Forest Haul Road Stream Crossing Approaches

A.J. Lang[a], W.M. Aust[b], M.C. Bolding[c], K.J. McGuire[d], and E.B. Schilling[e]

3.1 Abstract

Soil erosion and sediment delivery models were developed to estimate the inherent complexities of soil erosion, but most models are not specifically modified for forest operation applications. Three erosion models, the Universal Soil Loss Equation for forestry (USLE-forest), Revised Universal Soil Loss Equation Version 2 (RUSLE2), and Water Erosion Prediction Project (WEPP) were compared to trapped sediment data for 37 forest haul road stream crossings. We assessed model performance from five variations of the three erosion models, USLE-roadway, USLE-soil survey, RUSLE2, WEPP-default, and WEPP-modified. Each road approach was categorized into one of four levels of erosion (Very Low, Low, Moderate, and High) based on trapped erosion rate data and erosion rates reported in recent peer reviewed literature. Model performance metrics included: 1) summary statistics and a nonparametric analysis, 2) linear relationships, 3) percent agreement within erosion categories and tolerable error ranges, and 4)

[a] Graduate Research Assistant, Department of Forest Resources and Environmental Conservation (FREC), Virginia Tech, 305 Cheatham Hall, 310 West Campus Drive, Blacksburg, Virginia, 24061.
[b] Professor of Forestry, FREC, Virginia Tech, 228 Cheatham Hall 310 West Campus Drive, Blacksburg, Virginia, 24061.
[c] Associate Professor of Forest Engineering FREC, Virginia Tech, 228 Cheatham Hall 310 West Campus Drive, Blacksburg, Virginia, 24061.
[d] Associate Professor of Hydrology, FREC and Virginia Water Resource Research Center, Virginia Tech 210 Cheatham Hall, 310 West Campus Drive, Blacksburg, Virginia, 24061.
[e] Senior Research Scientist, National Council for Air and Stream Improvement Inc., Aubrey Texas 76227.

contingency table metrics. Sediment trap data varied from negligible (< 0.1) to hundreds of Mg ha^{-1} yr^{-1}. Soil erosion models evaluated could estimate erosion within 5 Mg ha^{-1} yr^{-1} for most approaches having erosion rates less than 11.2 Mg ha^{-1} yr^{-1}, while models estimates varied widely for approaches that eroded at rates above 11.2 Mg ha^{-1} yr^{-1}. The Kruskal-Wallis nonparametric analyses revealed that only WEPP-modified estimates were significantly similar to trapped sediment data ($p \geq 0.107$). While WEPP-modified ranked best for most model performance metrics, the time, effort, modeling expertise, and uncertainty associated with model results may discourage the use of the WEPP model as a forest management tool. WEPP is better suited for researchers and government agencies that have the capability to measure extensive parameter data. Additional sensitivity analysis is needed to expand default parameters for forest roads within the WEPP and USLE models.

3.2 Introduction

Sediment is often the most significant nonpoint source (NPS) water pollutant associated with forest operations in the US (SAF, 1995; EPA, 2005; Stuart and Edwards, 2006; Anderson and Lockaby, 2011). Substantial sedimentation adversely affects aquatic ecosystems, stream geomorphology, and water chemistry (Jackson et al., 2005; Jones et al., 2012; Pechenick et al., 2014). Erosion and subsequent sediment delivery to streams associated with silvicultural operations generally stem from the access system (i.e., haul roads, skid trails, stream crossings, and decks) (Elliot, 2000; Gucinski et al., 2001). The access system can generate surface runoff quickly because native soils are disturbed and compacted during construction. Reduced infiltration rates and altered hillslope hydrology increase erosion rates, especially when heavily trafficked, and increase connectivity between terrestrial and aquatic ecosystems (Sheridan and Noske, 2007; NRC, 2008). Surface runoff is managed by road and trail designs and implementation of forestry best management practices (BMPs) (Walbridge, 1997). Forestry BMPs minimize erosion by increasing cover over disturbed soils and controlling runoff volume and velocity (Shepard, 2006; Edwards et al., 2015). Forest road BMPs include proper planning (e.g. road location, road design, timing of operations, and scheduled road maintenance) (Walbridge, 1997; Swift and Burns, 1999), increasing surfacing (e.g. adding gravel or vegetative cover) (Brown et al., 2013), and closing roads and trails following operations (Madej, 2001; Sawyers et al., 2012; Wade et al., 2012). Since the adoption of the Federal Water Pollution Control Act (Clean Water Act [CWA]) of 1972, state NPS control programs have relied on BMPs to reduce erosion and sediment delivery from roads (Ice, 2004).

The Environmental Protection Agency (EPA) enforces the CWA statutes and determines appropriate policy for permitting exemptions. The EPA exempts many normal, ongoing silvicultural activities from the point source permitting process for discharges of dredge or fill material in wetlands and streams of the US (CWA, section 404). However, the direct connection of roads to waterways at stream crossings has sparked legal debate over the EPA's silviculture exemption. Specifically, legal cases (Environmental Defense Center v. US EPA, 344 F.3d 832 (9th Cir. 2003)) have focused on the EPA's decision to not regulate stormwater discharges from forest roads under the Phase II stormwater regulations. The US Supreme Court considered whether stormwater discharge permits should be required for logging roads under the CWA (Boston, 2012; MacCurdy and Timmons, 2013), and ultimately ruled in favor of the EPA's authority to determine appropriate permitting exemptions. The EPA maintained that under section 402(p)(6) of the CWA, flexible approaches are well suited to address the complexity of forest road ownership, management, and use (EPA, 2012). However, legal cases regarding this matter continue (EPA, 2015).

Researchers have also reported high sediment delivery rates at road stream crossings with inadequate BMPs (Taylor et al., 1999; Lane and Sheridan, 2002; Croke et al., 2005; Aust et al., 2015). BMPs for stream crossings include stabilizing bare soil near the stream, maintaining adequate streamside management zones (SMZs), and choosing a proper structure, design, and location of the crossing (Aust et al., 2011). When appropriate and correctly implemented stream crossing BMPs are applied, sediment delivery rates decrease (Aust et al., 2011; Brown et al., 2013; Nolan et al., 2015). The magnitude of sediment delivery from stream crossings is a complex issue governed by factors involving road attributes, soil composition, and infiltration characteristics (Luce and Black, 1999; Lane and Sheridan, 2002; Brown et al., 2013). Soil erosion and sediment delivery models were developed to estimate values for environmental parameters, but most models were not specifically designed for forest road applications. Researchers have modified agriculture models for forestry applications (e.g., forest version of the Universal Soil Loss Equation (USLE-forest) (Dissmeyer and Foster, 1984) and Water Erosion Prediction Project (WEPP) (Elliot and Hall, 1997), but further model improvements are necessary for forest roads (Fu et al., 2010). In light of erosion model deficiencies, forest managers are drawn to use models because results can be easily generated. Additionally, model results may further justify the costs associated with BMP implementation. Traditionally, two categories of soil erosion models have been applied in forest operation research: empirical and physical (Fu et al., 2010).

Empirical models are derived from experiments or observations and have simpler model structure compared to physically-based erosion models. Empirical models typically have fewer input parameters and are simpler to compute than physical models (Merritt et al., 2003). Because data requirements and site characterizations are often less, empirical models are easier to implement into forest management decisions (Christopher and Visser, 2007). Some popular examples of empirical erosion models include the USLE and the revised USLE (RUSLE). The USLE method of estimating soil loss is a widely used erosion model because of its ease and ability to generate estimations in the field (Christopher and Visser, 2007). USLE was originally designed to predict long-term annual soil loss from agricultural fields, but was adapted for forest lands to predict sheet and rill erosion to the bottom of a hillslope (USLE-forest, Dissmeyer and Foster, 1984). The RUSLE version 2 (RUSLE2) is an enhanced version of the USLE and RUSLE models (Renard et al., 1997). RUSLE2 was developed by the US Department of Agriculture-Agriculture Research Service (USDA ARS), the USDA Natural Resource Conservation Service (NRCS), and the Biosystems Engineering and Environmental Science Department of the University of Tennessee. RULSE2 uses the same process of estimating erosion as the USLE-forest, but allows for additional detail within input parameters such as, average temperature, monthly precipitation, slope profile breaks, permanent barriers, and specific management and support operations and their timing (Renard et al., 1991). Unlike the USLE, RUSLE2 is theoretically applicable to all land uses where raindrop impact and overland flow causes soil erosion (USDA ARS, 2008).

Researchers have also tested physically based erosion models for use in soil and water conservation, planning, and assessment (Flanagan et al., 2007).

Physically-based models are driven by the understanding of mass, energy, and momentum conservation. The detailed model parameters allow for further exploration of individual model components and their influence on erosional processes and sediment yields for the site they are applied. However, the number of inputs and differences among input measures make the application of physical models a more cumbersome tool for forest management. The water erosion prediction project (WEPP) was developed by the USDA ARS and Purdue University for federal government agencies managing sites for soil and water conservation (Flanagan et al., 2007). WEPP is a physically-based computer model that incorporates field observations and site level information for predicting erosion and sediment yield. WEPP can be run for either a hillslope profile or a watershed structure (Flanagan et al., 2012). Researchers have improved and estimated uncertainty of WEPP model runs to estimate sediment delivery from forest roads (Elliot et al., 1995; Laflen et al., 1997; Foltz et al., 2009), but management decisions should not be solely based on model predictions of sediment yield alone (Brown et al., 2016).

Comparisons of model predictions to observations are rare due to the associated costs of direct sediment measurements (Robichaud and Brown, 2002). Soil loss models and equations provide a time-efficient and cost-effective approach to evaluate BMPs and road designs (Fu et al., 2010). However, erosion estimates from models must be interpreted with caution, as issues with uncertainty are prevalent (Brazier et al., 2000). Few studies have compared model predictions to observed erosion using readily measureable field characteristics. Comparisons of erosion models with direct measures are necessary to assess and challenge existing model structure and input parameter data for the situations in which forest managers may use them (e.g., stream crossing, haul and skid roads). Revisions to model structure and parameter modifications within a model may improve agreement between observations and predictions, but cannot be verified, validated, or fully confirmed in the natural world (Oreskes et al., 1994). Several researchers have designed experiments to compared erosion models with direct sediment measures in order to evaluate specific BMPs (Sawyers et al., 2012; Wade et al., 2012; Wear et al. 2013; Brown et al., 2016). However, few studies have evaluated erosion models using simple field measures on operational stream crossing approaches that do not have specific BMP treatments.

Overall, researchers indicate that both empirical and physical models are useful tools in predicting erosion, identifying erosion prone areas, and ranking forestry BMP performances (Tiwari et al., 2000; Amore et al., 2004; Laflen et al., 2004; Croke and Nethery, 2006; Wade et al., 2012). Although USLE, RUSLE2, and WEPP erosion model were not specifically developed for forest road applications, numerous researchers have used them to estimate erosion and sediment yield (Megahan et al., 2001; Aust et al., 2015; Nolan et al., 2015). Each model has limitations and scope of usefulness that must be considered for sediment yield estimations (Brazier et al., 2000; Fu et al., 2010). Empirical models, such as USLE and RUSLE may be better suited for managers with limited site information. Physical models (e.g., WEPP) require more input parameters, but offer opportunities to modify the input parameters, which may allow for closer model estimates to observed values than empirical models on a wider range of sites.

3.2.1 Study Objectives

Previous research clearly indicates that forest stream crossings have above average potentials for delivery of sediment to streams and erosion models may prove useful for identifying erosion prone areas. Ranking erosion prone areas with erosion models could support management decisions to apply additional or more appropriate BMPs. The objectives of this manuscript were to: 1) evaluate model performance of three erosion models using field-informed measures by comparing model simulations to trapped sediment measurements, and 2) assess model utility for identifying stream crossing approaches that may require additional BMPs. The models evaluated were versions of the USLE-Forest (Dissmeyer and Foster, 1984), RUSLE2 (Toy et al., 1999), and WEPP (Elliot, 2004).

3.3 Methods

3.3.1 Study Areas

We selected and instrumented all 37 permanent forest haul road stream crossing approaches made available to us by MeadWestvaco Corporation (now WestRock), Virginia Tech, and US Forest Service. These approaches were located in the Virginia Piedmont and Ridge and Valley Physiographic regions. These regions are generally representative of topography in the Mountainous, Piedmont, and upper Coastal Plain regions of the southeastern United States and are the primary areas of concern with regard to forest road erosion (Aust et al., 1996). All stream crossings were originally constructed across intermittent or perennial streams with road standards that would permit the weight of large operational equipment transport trucks and log truck traffic. Streamside management zones (SMZs) were utilized adjacent to harvests, following Virginia Department of Forestry (VDOF) BMP guidelines (VDOF, 2011).

Twenty-one Piedmont approaches were instrumented in Appomattox and Buckingham Counties of Virginia in June of 2012. Topography of the Piedmont region is characterized by rolling hills with moderate slopes (USDA NRCS, 2012). Typical January temperatures near these study areas range from -4 to 8°C and July temperatures from 21 to 30°C. Average annual precipitation from historic weather data (1981 to 2010) is 120 cm occurring over 120 days (NOAA, 2015). Total rainfall data were collected daily from two personal weather stations affiliated with the Weather Underground service (WU, 2013) for Buckingham and Appomattox Counties. Soil series identified along road approaches were Appomattox-Cullen complex, Chewacla, Codorus-Hatboro complex, Mecklenburg-Poindexter complex, Grassland-Delanco complex, Tatum-Manteo complex, Spears Mountain, and Spears Mountain-Bugley complex (USDA NRCS, 2012). Piedmont sites were selected from seven intensively managed loblolly pine (*Pinus taeda* L.) plantations made available by MWV (now owned by WestRock). Although these roads were primarily constructed for timber management activities, low volumes of traffic from hunt clubs were the primary uses of most Piedmont approaches during this study. However, four instrumented approaches collected runoff data from frequent log truck passes in April, 2013. During this month, approximately 129 ha were harvested and additional gravel was applied to the haul road and the roads were widened.

Sixteen Ridge and Valley stream crossing approaches were also instrumented in Montgomery and Giles Counties of Virginia between July and November of 2012. The Ridge and Valley region is characterized by broad valleys separated by long linear ridges with low to moderate slopes (USDA NRCS, 2012). Typical January temperatures near these study areas range from -6 to 5°C and July temperatures from 16 to 28°C. Average annual precipitation from historic weather data (1981 to 2010) is 104 cm occurring over 130 days (NOAA, 2015). Total rainfall data were collected daily from three personal weather stations affiliated with the Weather Underground service (WU, 2013) within Giles and Montgomery Counties. The following soil series were identified along road approaches in this region: Jefferson, Berks and Weikert, Oriskany, Laidig, and Craigsville (USDA NRCS, 2012). Sites were selected from 12 stream crossings on two haul roads (Turkey Nest and White Rocks Roads) on the George Washington/Thomas Jefferson National Forest and four stream crossings on the Virginia Tech Fishburn Forest. The current purpose of the Forest Service roads is recreation (restricted public vehicle access), crop tree release activities, and small firewood cuttings, although the roads were originally constructed for timber harvests. The Virginia Tech School Forest road serves access for periodic teaching exercises and maintenance for municipal water and private cellphone towers, but the road has served for three separate timber harvests. The last maintenance to road surfaces on both National Forest and Virginia Tech school forest was 2-5 years prior to project installation.

3.3.2 Treatment Installation and Sediment Measurements

Stream crossing approaches were defined as the road areas sloping towards stream crossings. At the lowest elevation nearest the stream crossing, a narrow trench was hand-excavated at a 30°-45° angle across the road and a thick rubber conveyor belt was partially buried in the trench, leaving approximately 15 cm of belt exposed above the road surface (fig. 3.1). Water and sediment generated by overland flow were diverted from the road surface by the belt into an adjacent roadside silt fence catch area. This design was used to allow vehicular traffic and approximate sediment delivery to the stream. We assumed that sediment caught within the silt fence traps would have otherwise been deposited into the stream due to the close proximity. Conveyor belts were installed over a six-month period (June to November, 2012).

Figure 3.1. Rubber conveyor belt used to divert runoff into silt fence catchment areas.

Within the silt fence sediment collection areas, a series of 10 rebar pins marked the locations for repeated elevation measurements of trapped sediment (fig. 3.2). Additional sediment pins were added as sediment accumulated in order to better estimate deposited sediment. Elevations behind pins were measured using differential leveling with a total station (Sokkia total station model SET-520, Tokyo, Japan). Sediment deposits were measured approximately every two months over a one-year period, with all sites measured exactly one year after installation. Elevation change (m) (sediment deposits) and depositional area (m^2) were recorded during each site visit and multiplied to calculate sediment volumes (m^3). Three bulk density samples were collected after one year of sediment accumulation using the soil core method (Grossman and Reinsch, 2002) for each sediment catch area and analyzed. Sediment volume accumulation was multiplied by mean bulk density (Mg m^{-3}) to calculate sediment mass (Mg). Sediment mass corresponding to each repeated measure was expressed as mass per unit area (Mg ha^{-1}) by dividing the road surface area (area from the conveyor belt to the nearest water control structure), and later summed over one year to express potential sediment delivery on an annual basis (Mg ha^{-1} $year^{-1}$).

Figure 3.2. Silt fence catchment area with rebar pins marking the location of measurement.

Following erosion belt installation, approach characteristics and attributes hypothesized to influence erosion were recorded. Road prism dimensions (distance to water control structure, width, slope, and cutslope ratio) were surveyed using a total station. Width of SMZs and depth of gravel were measured with a tape. Bare soil percentages were collected seasonally and quantified by walking a zigzag pattern on the road running surface to the nearest waterbar and counting the number of steps where the toe of the boot contacted bare soil (i.e. count of bare soil steps / total number of steps x 100 = percent bare soil) (Brown et al., 2013). Percent cover on the road surface was determined during four seasonal measurements then averaged to account for the effect of vegetative growth and tree litter fall. Gravel coverages were also quantified in a similar manner. Canopy cover was measured seasonally using a spherical densitometer. Surface soil composites were collected from running surfaces and anticipated sediment source areas (i.e. bare soil areas along the approach that were not on the running surface [e.g. cutslopes]). Soil samples were analyzed for particle size classes using the hydrometer method (Gee and Or, 2002). Approach categorical variables were road template (insloped, outsloped, or crowned) and road shape (flat, concaved, convex, and s-shaped).

3.3.3 USLE-Forest

The forestland version of the USLE is comprised of six components that are multiplied together to estimate the amount of soil displacement to the bottom of a slope (Dissmeyer and Foster, 1984). The six components are: rainfall and runoff factor (R), soil erodibility factor (K), slope-length factor (L), slope-steepness factor (S), cover and management factor (C), and support practice factor (P). The rainfall and runoff factor (R) for Piedmont and Ridge and Valley sites was 180 and 145, respectively. These values were derived from the isoerodent map provided in the USLE forest manual (Dissmeyer and Foster, 1984). A soil erodibility factor (K) was calculated for each approach using two methods. We evaluated two techniques of the USLE-Forest using soil data obtained from the road surfaces (USLE-roadway) versus soil data estimated using the NRCS soil surveys (USLE-soil survey). The (USLE-roadway) utilized the percent sand, silt, and clay determined in the particle size analysis (Gee and Or, 2002). The following equation was used to calculate the soil erodibility factor (K) for this method:

$$K = 2.1M^{1.14}\left(10^{-6}\right)\left(12 - a\right) + 0.0325\left(b - 2\right) + 0.025\left(c - 3\right)$$

where M = (percent silt + very fine sand)(100-percent clay), a = percent organic matter, b = structure code, and c = profile permeability class. The second method (USLE-soil survey) utilized the USDA NRCS soil survey erodibility values given for the B horizon identified at each approach, as the road construction process removes the top soil and exposes subsoils. A total station was used to measure slope length (L) and steepness (S). The values collected were used in the following equation to calculate the LS factor:

$$LS = \left(\lambda / 72.6\right)^{m}\left(65.41\sin^{2}\theta + 4.65\sin\theta + 0.065\right)$$

where λ = slope length (feet), θ = the slope angle (degrees), and m = 0.2 for <1% slopes, 0.3 for 1 to 3% slopes, 0.4 for 3.5 to 4.5% slopes, and 0.5 for ≥5% slopes (Dissmeyer and Foster, 1984). The cover, management, and support practices (CP) values for a tilled soil were used to approximate topsoil disturbance and traffic effects of haul roads. CP subfactors include: (1) bare soil, residual binding, and soil reconsolidation; (2) canopy effect; (3) steps; (4) on-site storage; (5) invading vegetation; and (6) contour tillage. The bare soil, residual binding, and soil reconsolidation subfactor was estimated using the bare soil estimates measured seasonally during site visits. The remaining CP subfactors were estimated according to each approach condition. Erosion rates were calculated seasonally for each approach. This provided four erosion rates per approach over one year. The four erosion rates of each approach were averaged to determine mean annual erosion rate (Mg ha^{-1} yr^{-1}).

3.3.4 RUSLE2

Erosion estimates were calculated using RUSLE2 (version 2.5.7.7). Unlike USLE, RUSLE2 allows for additional parameter inputs that can be adjusted to more adequately mimic site conditions. First, climate data were accessed from the NRCS National RUSLE2 database (USDA NRCS, 2015) for Appomattox, Buckingham, Giles, and Montgomery counties, Virginia. These datasets included average daily and monthly values between 1961 to 1990 for rainfall and temperature. We used these datasets because we did not measure rainfall intensity, which is needed to calculate erosivity density within RUSLE2 (USDA ARS, 2008). Second, a soil file can be downloaded from RUSLE2 database or created for site specific conditions. Within the soil files, there is information on soil texture, soil erodibility index, hydrologic class utilizing curve number methodology, and acceptable soil loss rates. New soil files were created by modifying generic soil files with general soil textures (i.e. sand, silt, and clay combinations) and qualitative levels of soil organic matter (i.e. low, moderate, high) and permeability (slow, moderate, fast). For this study, soil files for each approach were generated according to soil particle size determined in lab analysis of soil texture. Due to the original construction template and traffic, we assumed low soil organic matter and slow permeability for all approaches. All other soil parameters remained at default settings. Third, slope topography can be created having no breaks or multiple breaks within the slope profile. Elevation data were collected at points along each approach where obvious changes in the slope shape or grade occurred and these elevations were used to create detailed profiles within the model. The fourth component was a management file that described surface conditions. Within the software, there are management files that describe highly disturbed land. The gravel surface file most closely approximated forest haul roads, but the operations within these files do not specifically reflect forest road activities (e.g. traffic level, surface condition, and road template). Management files were created for each road approach by modifying the "gravel surface, fresh" management file. Within this file, two default operations are implemented, "blade fill material" and "add mulch". The blade fill material operation can be changed to several different options, but only "blade cut material" or "blade fill material" seemed suitable for characterizing forest haul roads. The "blade cut material" generates less erosion than "blade fill material". The second operation ("add mulch") allows users to choose from a variety of mulching agents. The type of mulch applied in this study was "mulch materials\rock". The rate of application parameter was adjusted until the resulting "cover from addition" described the mean approach soil coverage. Application rates varied from 11.4 Mg ha^{-1} to 700 tonnes ha^{-1} to achieve mean soil coverage from 6 to 98% in RUSLE2. Sediment delivery estimates were made for a one-year model run.

3.3.5 WEPP

WEPP (version 2012.8) was used to estimate erosion at each road approach. Files related to slope, climate, soils, and vegetation management were modified or created to develop hillslope profiles for each of the 37 road approaches. Similar to RUSLE2, WEPP contains an extensive database for climate, soils, and vegetation management files. However, there are more adjustable parameters within WEPP than RUSLE2. The user may implement or modify files to fit a specific hillslope or watershed. We wished to compare the use of WEPP as generally available with little modification (WEPP-default) and WEPP with extensive modifications (WEPP-modified), thus two methods of attaining sediment yield estimates with WEPP were assessed. For both methods, slope files were created in the WEPP model similarly to the slopes created in RUSLE2. In addition to the RUSLE2 slope profile, WEPP has parameters regarding road surface width, road vertical shape (concave, convex, linear, or S-shaped), and aspect. These parameters were collected during the initial road survey using a total station and were input into both WEPP methods.

For the WEPP-default method, default or minimal adjustments were made to the soil and management files with the premise of utilizing easily accessible field information. Default management files were chosen based on their description and were not modified. Soil files were chosen based on mapped soil survey information. Climate data were simulated for 100 years using the CLIGEN weather generator. This method produces daily time series estimates for climate parameters in a selected location based on the nearest weather station with historic mean monthly measurements. The resulting WEPP-default files were used in the simulation.

For the second method (WEPP-modified), parameter values for climate, soils, and vegetation management procedures were modified based on measurements of site characteristics and communication with William Elliot (USDA Forest Service, personal communication, September 2015). Daily rainfall, minimum air temperature, and maximum air temperature data were collected from five personal weather stations affiliated with Weather Underground service (WU, 2013) located nearest the five groups of approaches. Data were recorded for the period of January 1, 2012 to December 31, 2013. To simulate rainfall, WEPP requires additional climate parameters including duration, time to peak intensity, peak intensity, daily solar radiation, wind velocity, and wind direction. These parameters were generated using the “Add climate location” function in WEPP. We selected the nearest weather station with associated CLIGEN files for each group of approaches to create a parameter (PAR) file. This method matches the observed data (e.g. rainfall and temperature) and averages the non-recorded data from the selected weather station(s) climate records. The five resulting climate files were used in the analysis. Within the WEPP soil database, 16 default files specify the term "road surface". These files are differentiated with soil texture, road template, and/or surfacing. Soil files to be modified were selected by the known road surface texture, then surfacing (e.g. gravel, vegetated, bare), and road template for each approach. WEPP-modified soil files were modified by adjusting percent sand, silt, and clay and keeping the default settings for interrill erodibility, rill erodibility, critical shear, and effective hydraulic conductivity. Within the WEPP vegetation management database, eight road management files could potentially be appropriate for haul roads. Management files require input for initial conditions that start on January 1. Nine initial condition files could be suited for haul roads. Within the initial condition files, 21 parameters may be adjusted. One of the 21 parameters is the initial plant condition, which has another subset of 33 parameters grouped into plant growth and harvest, temperature and radiation, canopy, leaf area index, and root, senescence, and residue parameters. The combined effects of gravel and litterfall on the road surface cover were modeled by modifying parameters within one of the eight default files. Parameters values adjusted for initial conditions were canopy cover, interrill and rill covers, and initial plant. Canopy cover (%) was set to the recorded value during winter site visits. Interrill and rill covers were assigned the inverse values of percent bare soil recorded for winter site visits. Within the subset of initial plant parameters, canopy cover coefficient, decomposition rate, and biomass energy ratio were adjusted by trial and error to mimic canopy and road cover over the one-year study period. The resulting management files were used in the simulation.

3.3.6 Data Analysis

Our goal was to compare model estimates to trapped sediment erosion rates. A variety of methods were used to compare trapped sediment and modeled sediment erosion estimates including: 1) summary statistics and a nonparametric analysis, 2) linear relationships, 3) percent agreement within erosion categories and tolerable error ranges, and 4) a contingency table.

We tested for statistically significant differences between the trapped sediment data and each modeling method. Trapped sediment erosion rates were not normally distributed nor varied equally. Data transformations of the sediment trap measures did not result in equal variance (Levene's test). Therefore, nonparametric Kruskal-Wallis methods were used to compare erosion rates between trapped sediment data and modeled estimates (Zar, 2010). Separations of significantly different medians were based on Wilcoxon each pair tests (Zar, 2010). All statistical analyzes were completed in JMP® version 11 software package (JMP®, 2016).

We categorized trapped sediment data into four erosion categories (Very Low, Low, Moderate, and High), where Very Low erosion rates were < 10 Mg ha^{-1} yr^{-1}, Low erosion rates were 10.1 – 25 Mg ha^{-1} yr^{-1}, Moderate erosion rates were 25.1 – 50 Mg ha^{-1} yr^{-1}, and High erosion rates were > 50 Mg ha^{-1} yr^{-1}. Similar erosion rates were reported by Morris et al. (2014) and Nolan et al. (2015) for forest haul road stream crossings with various levels of BMPs. Next, we assigned tolerable ranges of error that the model estimates could deviate from trapped data. For the tolerance ranges, we assigned erosion rates (Mg ha^{-1} yr^{-1}) as $\pm$20, 10, 5, 3, and 1 for trapped erosion rates greater than 100, 99.9-50, 49.9-10, 9.9-1, and < 1, respectively. For example, a measured erosion rate of 75 Mg ha^{-1} yr^{-1} could be modeled between 85 and 65 Mg ha^{-1} yr^{-1} and be within our acceptable/tolerable range of error. An erosion rate modeled above 85 Mg ha^{-1} yr^{-1} or below 65 Mg ha^{-1} yr^{-1} would be categorized as above and below the tolerable range of error, respectively.

Another method to assess model performance is with contingency tables. Contingency tables can be used to report the behavior of environmental models at a specific level, known as a "trigger" or "alarm" (Bennett et al., 2013). For this study, we chose 11.2 Mg ha^{-1} yr^{-1} as the trigger. We chose the trigger based on the NRCS allowable erosion rates (T-factor) for the soil series involved in the study, which range from 2.2 to 11.2 Mg ha^{-1} yr^{-1}. The T-factor is the tolerable rate of soil loss at which the quality of a soil as a medium for plant growth can be maintained (USDA NRCS, 2011). Additionally, we examined literature on road erosion rates for high versus low erosion rates on roads and determined that 11.2 Mg ha^{-1} yr^{-1} generally captured the concept of high versus low erosion rates for forest roads. Brown et al. (2013) and Kochenderfer and Helvey (1987) found that forest roads with adequate BMPs typically produce erosion rates below our trigger. We calculated the number of occurrences in which trapped sediment data and model estimates were both above the trigger (hits), both below the trigger (correct negatives), trapped sediment above and modeled estimate below the trigger (miss), and trapped sediment below and modeled estimate above the trigger (false alarm). From these results, it is possible to derive a number of measures of model performance (Bennett et al., 2013). We reported model accuracy, bias score, probability of detection, false alarm ratio, and success index metrics from the contingency table results for each model. From a management perspective, erosion models that can correctly identify approaches above a threshold would be beneficial for determining which approaches would benefit from additional BMPs.

3.4 Results

3.4.1 Summary Statistics and Nonparametric Analysis of Model Performance

Summary statistics of approach characteristics by region are provided in table 3.1. Additional site descriptions and BMP levels for roads in this study were previously reported in Lang et al. (In Review). In general, approaches in the Piedmont region tended to have shorter lengths, wider widths, gentler slopes, and greater average bare soil than approaches in the Ridge and Valley region. Differences could be attributed to management objective differences (e.g. intensive, short rotation pine silviculture management in the Piedmont and extensive long rotation hardwood management in the Ridge and Valley).

Table 3.1. Descriptive statistics of stream crossing approaches in the Piedmont, Ridge and Valley, and both physiographic regions.

Region[a]	Measure	Max.	Q_{75}	Mean	Median	Q_{25}	Min.
Piedmont	Approach length (m)	130	62.5	45.2	30.5	23.3	12.8
	Approach width (m)	6.40	4.57	3.82	3.66	3.20	2.13
	Slope (%)	16.0	8.00	6.62	6.00	4.00	2.00
	Bare soil (%)	93.8	66.3	52.0	50.0	34.4	8.00
Ridge & Valley	Approach length (m)	427	97.5	88.2	61.7	24.0	6.10
	Approach width (m)	3.66	3.28	3.05	3.05	2.74	2.44
	Slope (%)	13.0	11.3	7.19	7.00	5.00	1.00
	Bare soil (%)	42.5	28.1	19.3	14.6	10.7	1.75
Both regions	Approach length (m)	427	80.9	63.8	39.6	23.3	6.10
	Approach width (m)	6.40	3.66	3.48	3.35	2.74	2.13
	Slope (%)	16.0	8.00	6.86	7.00	4.50	1.00
	Bare soil (%)	93.8	53.8	37.9	35.0	14.6	1.75

[a]Twenty-one stream crossing approaches located in the Ridge and Valley region and 16 in the Piedmont region.

Trapped sediment data ranged from 291 to less than 0.01 Mg ha^{-1} yr^{-1}, with a mean and median of 20 and 1.44 Mg ha^{-1} yr^{-1}, respectively (table 3.2). All models over predicted trapped erosion rates at the 75^{th} quantile, median, and 25^{th} quantile. At the 75^{th} quantile, model estimates were greater than trapped erosion rates by more than 9 Mg ha^{-1} yr^{-1}. At the median, overestimates ranged from 1.59 to 10.6 Mg ha^{-1} yr^{-1}. At the 25^{th} quantile, overestimates were all less than 6 Mg ha^{-1} yr^{-1}. These results indicate better model and trapped sediment agreement at lower rather than higher erosion rates. Means of modeled erosion rates deviated from the trapped sediment mean by 0.7 to 5.5 Mg ha^{-1} yr^{-1} and displayed high measures of variance and positive skewedness (table 3.2). WEPP-modified had the closest median estimate to trapped sediment data, followed by RUSLE2, USLE-roadway, USLE-soil survey, and WEPP-default.

Table 3.2. Descriptive statistics for trapped sediment data and modeled estimates categorized by all erosion rates and threshold erosion rates above and below 11.2 Mg ha^{-1} yr^{-1}.

	Estimation method	Max.	Q_{75}	Mean	Median	Q_{25}	Min.	Variance	Skew
All	Trapped sediment	290.7	9.6	20.0	1.4	0.3	<0.1	2800	4.2
	USLE-roadway	206.0	20.6	19.3*	8.1	2.0	0.1*	1373	4.0
	USLE-soil survey	201.0	19.6	22.1	9.0	3.2	0.1	1428	3.5
	RUSLE2	100.0	19.0*	15.5	6.3	2.3	0.4	1428	3.5
	WEPP-modified	273.0*	23.0	22.7	3.0*	0.9*	0.4	2539	3.8
	WEPP-default	192.0	26.0	25.5	12.0	6.2	2.8	1411	3.2
Above threshold	Trapped sediment	290.7	110	84.9	58.3	23.6	12.6	8223	2.0
	USLE-roadway	206.0	84.0	53.0	17.6	8.0	4.0*	4822	1.8
	USLE-soil survey	201.0	94.1	58.9	29.5*	9.9	2.0	4632	1.5
	RUSLE2	100.0	38.0	27.7	16.0	6.9	2.3	1022	2.0
	WEPP-modified	273.0*	83.9	60.4*	27.7	4.0	0.8	8409	2.2
	WEPP-default	192.0	99.9*	51.6	20.8	11.6*	2.8	4722	1.7
Below threshold	Trapped sediment	10.2	3.8	2.1	0.6	0.2	<0.1	8	1.5
	USLE-roadway	49.7*	14.5	10.0*	5.7	1.1	0.1*	145	1.8
	USLE-soil survey	54.4	16.7	12.0	7.7	2.3	0.1	184	1.8
	RUSLE2	71.0	16.5	12.1	4.8	1.9	0.4	305	2.2
	WEPP-modified	107.0	5.6*	12.3	2.7*	0.8*	0.4	645	2.7
	WEPP-default	84.3	23.3	18.3	11.8	5.9	3.5	384	2.1

* estimate that most closely approximates trapped sediment data within the column of each category

For approaches above the erosion threshold (> 11.2 Mg ha^{-1} yr^{-1}), all models tended to under predict erosion rates for all summary statistics (table 3.2). WEPP-modified was closest to the trapped sediment at the maximum and mean. WEPP-default was closest to the trapped sediment at the 75th and 25th quantiles. USLE-soil survey had the closest median estimate to trapped data (29.5 Mg ha^{-1} yr^{-1}). For approaches below the erosion threshold, all models tended to over predict erosion rates for all summary statistics (table 3.2). WEPP-modified was furthest from trapped sediment at the maximum, but closest at the median, 75th, and 25th quantiles. WEPP-modified had the closest modeled estimates to over half of the approaches with erosion rates below the threshold. USLE-roadway had the closest mean estimate to trapped sediment data below the threshold, albeit a 7.86 Mg ha^{-1} yr^{-1} difference.

The spread between each modeling method and trapped sediment data was relatively low at the mean and median, likely because 78% of stream crossing approaches produced low erosion values (< 11.2 Mg ha^{-1} yr^{-1}). Each modeling method estimated erosion rates within the range of trapped sediment data (fig. 3.3). WEPP-default had the highest mean magnitude of error at 32.4 Mg ha^{-1} yr^{-1}, where the magnitude of error is calculated as the absolute value of the difference between the model predictions and the sediment trap rates. The second highest mean magnitude of error was WEPP-modified at 29.3 Mg ha^{-1} yr^{-1}. The third and fourth highest mean magnitude of error was USLE-soil survey and USLE-roadway at 24.8 and 24.4 Mg ha^{-1} yr^{-1}, respectively. The lowest mean magnitude of error was RUSLE2 at 22.9 Mg ha^{-1} yr^{-1}.

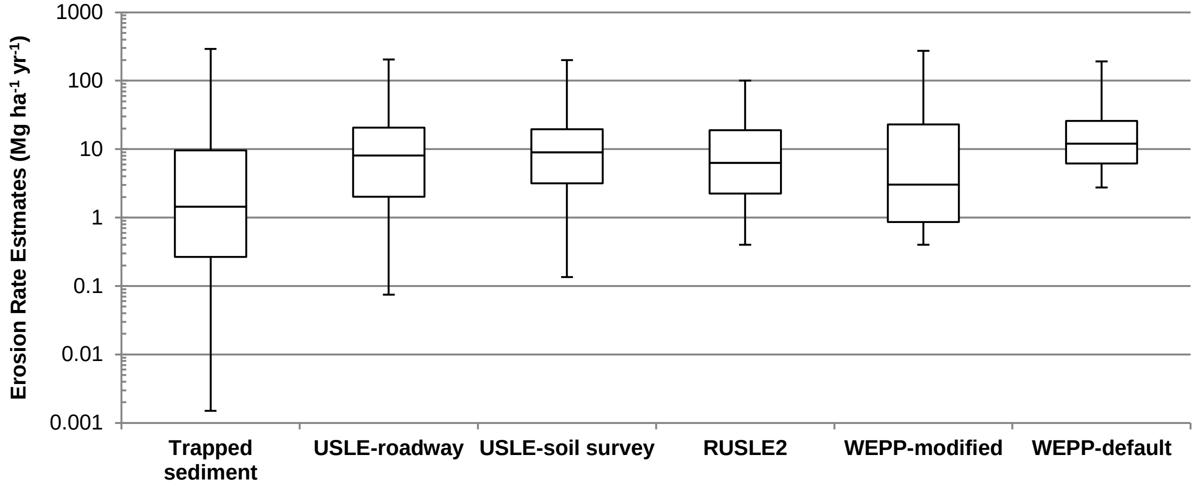

Figure 3.3. Boxplots of erosion rate estimates for trapped sediment data and each modeling method. Box and whisker plots display first, second, and third quartile and maximum and minimum values. Note the log scale.

We compared all model erosion estimates to the sediment trap data using Kruskal-Wallis statistical test (Zar, 2010). The Kruskal-Wallis statistical test indicated significant differences between trapped sediment data and model estimates ($p < 0.001$). The results of the Wilcoxon's comparison test show significant differences between trap sediment data and USLE-roadway ($p \leq 0.019$), USLE-soil survey ($p \leq 0.005$), RUSLE2 ($p \leq 0.010$), and WEPP-default ($p \leq 0.001$). WEPP-modified was not significantly different from trapped sediment data ($p \geq 0.107$), therefore WEPP-modified performed better than all other modeling methods examined.

3.4.2 Linear Relationships

All models had coefficients of determination (R^2) below 0.5 (fig. 3.4). Weak linear relationships between sediment trap data and each model indicated poor overall model performances on haul road stream crossing approaches. USLE-soil survey had the strongest linear relationship with trapped sediment (r = 0.46 and R^2 = 0.21) followed by USLE-roadway (r = 0.40 and R^2 = 0.16). RUSLE2, WEPP-modified, and WEPP-default models had correlation coefficients less than 0.3 and coefficients of determinations less than 0.10. Overall, the linear relationships between the trapped sediment and model estimates did not justify additional exploration of these relationships. These relationships clearly indicate the need for additional development/improvement of erosion models for prediction of erosion associated with stream crossing approaches.

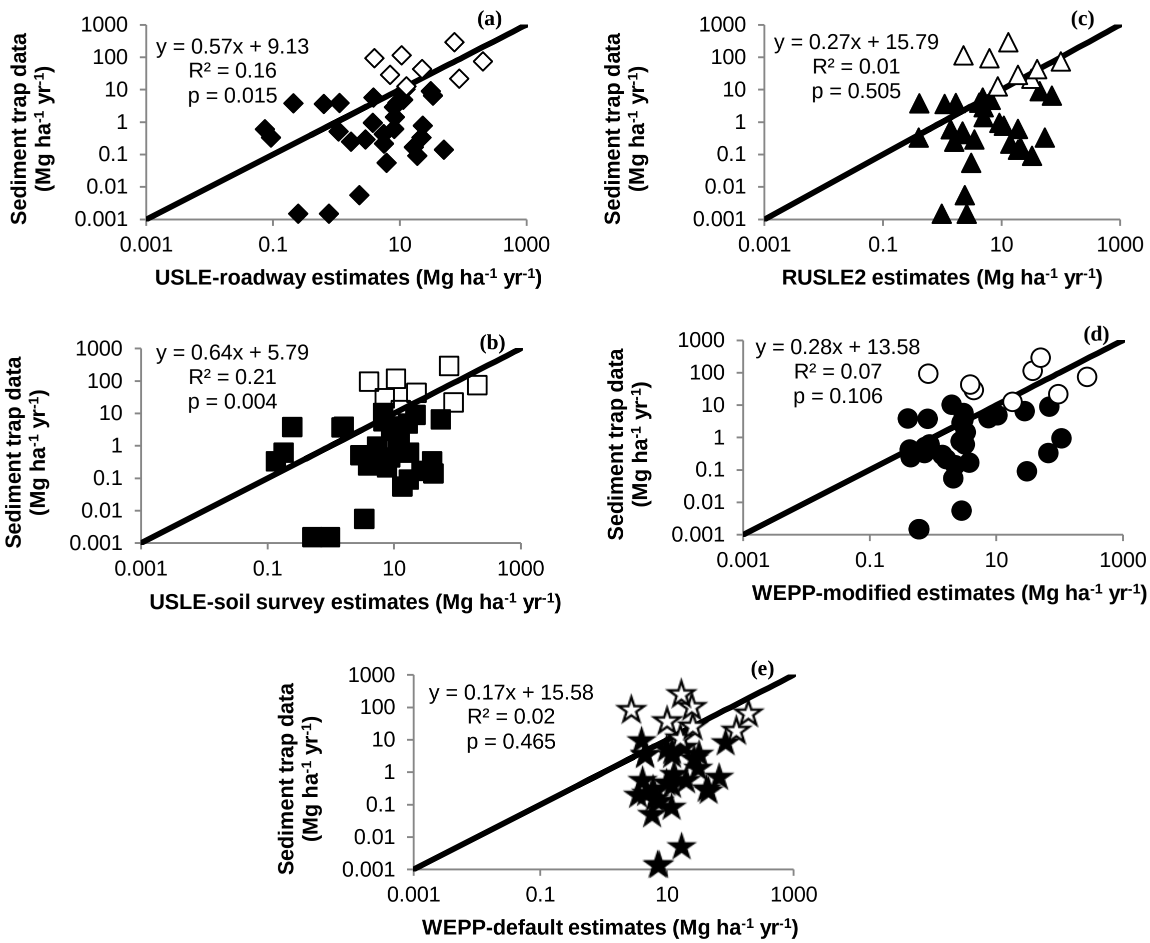

Figure 3.4. Linear relationship between sediment trap data and USLE-roadway (a), USLE-soil survey (b), RUSLE2 (c), WEPP-modified (d), and WEPP-default (e) estimates. Filled and hollow data points represent sediment trap values below and above the 11.2 Mg ha^{-1} yr^{-1} threshold, respectively. Data above and below the 1:1 line indicate model under and over estimations, respectively.

3.4.3 Percent Agreement between Trapped Sediment Data and Modeled Estimates

Stream crossing approaches were ranked into erosion rate categories (Very Low, Low, Moderate, and High) according to trapped sediment erosion rates (table 3.3). For these stream crossing approaches, 54% were Very Low followed by Low (27%), High

(11%), and Moderate (8%). All soil erosion models had the best agreement at the Very Low level. WEPP-modified correctly identified 85% of Very Low approaches, followed by USLE-roadway (75%), RUSLE2 (65%), USLE-soil survey (60%), and WEPP-default (50%). At the Low level, all models performed slightly worse than at the Very Low level. WEPP-modified and USLE-roadway each correctly identified 70% of approaches followed by RUSLE2 (60%), USLE-soil survey (60%), and WEPP-default (50%). All soil erosion models inadequately categorized approaches in the Moderate and High categories. Of the seven approaches in these two erosion categories, USLE-soil survey correctly identified three, while the three remaining models correctly identified only two approaches. Overall, WEPP-modified correctly identified 70% of erosion categorized approaches, followed by USLE-roadway (65%), RUSLE2 (57%), USLE-soil survey (54%), and WEPP-default (43%) (table 3.3).

Table 3.3. Percentage of correct BMP category[1] identifications by each soil erosion model.

Erosion category	n	Erosion range	Trapped sediment range	Correct BMP category placement by				
				USLE-roadway	USLE-soil survey	RUSLE2	WEPP-modified	WEPP-default
		($Mg\ ha^{-1}\ yr^{-1}$)				(%)		
Very Low	20	< 10	< 0.1 - 2.9	75%	60%	65%	*85%	50%
Low	10	10.1 - 25	3.6 - 12.6	*70%	50%	60%	*70%	40%
Moderate	3	25.1 - 50	21.8 - 43.1	0%	*33%	*33%	0%	*33%
High	4	> 50.1	73.6 - 290.7	*50%	*50%	25%	*50%	25%
All Categories	37	-	< 0.1 - 290.7	65%	54%	57%	*70%	43%

[1] BMP categories based on Nolan et al. (2015) and Brown et al. (2013).
* greatest percent correct within the row of each BMP category

Regarding the sediment trapped from the 37 stream crossing approaches, two collected more than 100 $Mg\ ha^{-1}\ yr^{-1}$; two collected between 99 and 50 $Mg\ ha^{-1}\ yr^{-1}$; five collected between 49 and 10 $Mg\ ha^{-1}\ yr^{-1}$; ten collected between 9.9 and 1.0 $Mg\ ha^{-1}\ yr^{-1}$; and 18 collected less than 1 $Mg\ ha^{-1}\ yr^{-1}$. WEPP-modified and RUSLE2 had the highest percent agreement (32%), where percent agreement is the frequency in which modeled estimates were within the tolerable error ranges of trapped sediment data. USLE-roadway and USLE-soil survey had the next highest percent agreement (27%) followed by WEPP-default (11%). Seventy-six percent of WEPP-default estimates were above the extent of the tolerable error ranges, while fewer were observed from WEPP-modified (49%), RUSLE2 (54%), USLE-roadway (57%), and USLE-soil survey (59%). The frequencies of estimates below the extent of the reasonable error ranges for all models were much less. WEPP-modified most frequently underestimated trapped sediment values (19%), followed by USLE-roadway (16%), USLE-soil survey (14%), RUSLE2 (14%), and WEPP-default (14%).

3.4.4 Contingency Table Assessment

Model performance is seldom evaluated by a single metric, thus we calculated five model performance metrics (table 3.4) as provided by Bennett et al. (2013). The five selected metrics were accuracy, bias score, probability of detection, false alarm ratio, and success index. All five metrics require a “trigger”. We established a trigger (Bennett et al., 2013) rate of 11.2 Mg ha^{-1} yr^{-1} for the 37 stream crossing approaches. We determined that 29 approaches had trapped erosion rates below this trigger, while eight approaches had erosion rates greater than this trigger rate of 11.2 Mg ha^{-1} yr^{-1}.

Model accuracy ranged from 0.54 to 0.78 (table 3.4). WEPP-modified had the best accuracy (0.78), followed by RUSLE2 (0.70), USLE-roadway (0.68), USLE-soil survey (0.65), and WEPP-default (0.54). Bias scores (Bennett et al., 2013) indicate whether the model will underestimate (bias < 1) or overestimate (bias > 1). All models tended to overestimate more frequently than underestimate in instances where trapped sediment rates were less than the trigger. WEPP-default had the highest bias score (2.38), followed by USLE-soil survey (2.13), USLE-roadway (1.75), RUSLE2 (1.63), and WEPP-modified (1.25). The probability of detection rate (Bennett et al., 2013) was greatest with WEPP-default and USLE-soil survey models (0.75), while the other three models had equivalent detection rates (0.63). False alarm ratios ranged from 0.50 to 0.79. WEPP-default most frequently overestimated erosion rates below the trigger (0.79) followed by USLE-soil survey (0.65), USLE-roadway (0.64), RUSLE2 (0.62), and

WEPP-modified (0.50). The success index was greatest for WEPP-modified (0.73), followed by USLE-soil survey (0.69), RUSLE2 (0.67), USLE-roadway (0.66), and WEPP-default (0.62) (table 3.4). Based on these metric of model performance, WEPP-modified was the “best” model in four of the five metrics (accuracy, bias score, false alarm ratio, and success index) and WEPP-modified was second best in the probability of detection. However, it is important to note that WEPP-default was the poorest performer with regard to these five metrics and was the “worst” performing model in all other metrics of model performance.

Table 3.4. Model performance metrics that compare sediment trap data with model estimates.

Estimation method	Accuracy[a]	Bias score[a]	Probability of detection[a]	False alarm ratio[b]	Success index[a]
USLE-roadway	0.68	1.75	0.63	0.64	0.66
USLE-soil survey	0.65	2.13	0.75	0.65	0.69
RUSLE2	0.70	1.63	0.63	0.62	0.67
WEPP-modified	0.78	1.25	0.63	0.50	0.73
WEPP-default	0.54	2.38	0.75	0.79	0.62

[a] Ideal value = 1
[b] Ideal value = 0

3.5 Discussion

The overall goal of this project was to evaluate the sediment delivery potential from forest stream crossings and compare results to model estimates. As compared to the sediment trap erosion data, all soil erosion models tended to overestimate median sediment values. However, all soil erosion models frequently performed sufficiently well to properly place the erosion estimates into correct erosion categories. Proper erosion category placement/ranking corresponds well with other forest road studies in Virginia that compared measured erosion with modeled estimates of erosion. Brown et al. (2013) measured and modeled sediment delivery from stream crossing approaches and reported measured sediment delivery rates from 10 to 16 Mg ha^{-1} yr^{-1} for graveled approaches and 34 to 287 Mg ha^{-1} yr^{-1} for bare soil approaches. They concluded that both USLE-forest and WEPP performed well in identifying problem road approaches, but WEPP performed better USLE-forest. Wade et al. (2012) reported mean erosion rates less than 32 Mg ha^{-1} yr^{-1} for bladed skid trails with seed, mulch, or slash closure BMPs and 138 Mg ha^{-1} yr^{-1} for bare soil with water bar control treatments. Similarly, USLE-forest, RUSLE2, and WEPP models adequately ranked soil erosion rates for trails with and without BMPs. However, WEPP did not perform as well as RUSLE2 and USLE-forest. Sawyers et al. (2012) compared trapped sediment data to WEPP and USLE-forest models for overland skid trails and found that WEPP performed better than USLE-forest, but both models generally agreed with trapped sediment data. Our results and those from other similar studies, clearly indicate that forest operations with inadequate BMPs or poor maintenance have the potential to produce excessive erosion, yet BMPs often reduce sediment yield to the stream to amounts similar to geologic rates (Patric, 1976). Additionally, versions of the USLE and WEPP models perform sufficiently to identify problem areas, but have varying results in adequately quantifying sediment from forest roads.

Our sediment trap data indicated that erosion rates on haul road stream crossing approaches vary from negligible to hundreds of Mg ha^{-1} yr^{-1}. Four of the 37 stream crossing approaches eroded at High rates (exceeded 50.1 Mg ha^{-1} yr^{-1}) (table 3.3). These road approaches had evidence of water control; however, the water control structures were no longer effectively controlling stormwater runoff and were directly connected to the stream channel. Approaches categorized as High would benefit from road maintenance and would require extensive corrective BMPs to control erosion. Croke et al. (2005) described direct connectivity in their study as occurring when channels or gullies formed, leading directly to stream channels, typically below cross drains with excessive contributing areas. Direct connections to stream channels in this study stemmed from the road surface. The combination of the four most eroded approaches contributed 573 Mg ha^{-1} yr^{-1}, which is 3.4 times greater than the sum of all erosion from the remaining 33 stream crossing approaches. Nolan et al. (2015) used the USLE-forest to estimate erosion at haul road stream crossings across Virginia and found that erosion rates were 13x greater for site with inadequate BMPs as compared to site with correctly implemented BMPs.

Another objective of the study was to evaluate model performance of three erosion models with different levels of field informed detail by comparing model simulations to trapped sediment measurements. We used 12 different model performance metrics to compare the ability of the five model types/methods for prediction of erosion from forest road stream crossing approaches. Table 3.5 provides the findings from the nonparametric analysis and model ranks for the remaining 11 model performance metrics. Overall, USLE-roadway, USLE-soil survey, and RUSLE2 had similar cumulative rankings (26, 30, and 26) where the best possible ranking could be 11 and the worst potential ranking could be 55. However, WEPP-modified had the best overall ranking with a cumulative score of 15 and was the “best” model for 9 of the 11 metrics. Kruskal-Wallis nonparametric analyses revealed that WEPP-modified sediment estimates were not significantly different from trapped sediment data ($p \geq 0.107$) and WEPP-modified better approximated the trapped sediment volumes than did all other models. In contrast, WEPP-default was found to have the highest cumulative score which indicated that this model was the least desirable of the five models that we evaluated.

Table 3.5. Model ranks for 12 model performance metrics. (1 = best performing model, 5 = poorest performing model, identical numbers indicate equal performance). S.D. (Significantly Different) and N.S.D. (Not Significantly Different).

Performance Metrics	USLE-roadway	USLE-soil survey	RUSLE2	WEPP-modified	WEPP-default
Descriptive: all	2	4	3	1	4
Descriptive: above	2	2	3	1	1
Descriptive: below	1	2	2	1	2
Linear relationships	2	1	3	3	3
Correctly placed erosion categories	2	4	3	1	5
Percent agreement with tolerable error ranges	2	2	1	1	3
Contingency table: accuracy	3	4	2	1	5
Contingency table: bias score	3	4	2	1	5
Contingency table: probability of detection	2	1	2	2	1
Contingency table: false alarm ratio	3	4	2	1	5
Contingency table: success index	4	2	3	1	5
Nonparametric analysis	S.D.	S.D.	S.D.	N.S.D.	S.D.
Total cumulative score	**26**	**30**	**26**	**15**	**39**

The general WEPP model incorporates numerous parameters that are intended to reflect the environment, such as climate, hydrology and water budget, plant growth, soil composition, and various forestry and agricultural operations (Amore et al., 2004). The intention of WEPP was to succeed the USLE method, as the demand for greater accuracy increased (Flanagan et al., 2007). Numerous researchers have found that the WEPP model performed well for agricultural (Abaci and Papanicolaou, 2009), rangeland (Simanton et al., 1991), construction (Moore et al., 2007), and various forest operations (Forsyth et al., 2006; Dun et al., 2009; Wade et al., 2012; Elliot, 2013), yet considerable modifications to input parameters and calibration are generally required in order to allow satisfactory model performance. Even when predicted model estimates are consider satisfactory, uncertainty and equifinality remain an issue (Brazier et al., 2000). Brown et al. (2016) evaluated the uncertainty in WEPP estimates of sediment yield from forest road stream crossing approaches and concluded that WEPP's utility for estimating BMP effectiveness is limited to predicting relative differences in vastly different BMP treatments. Additionally, they were unable to confidently recommend parameter ranges specific to bare or gravel surface treatments for forest road approaches. However, our results of most model performance metrics and statistical analyses indicated that WEPP-modified had the greatest potential of the soil erosion models evaluated to estimate soil erosion for forest road stream crossing approaches. Reasons why WEPP may have performed better than the empirical models include greater model specificity (e.g. incorporation of observed rainfall, soil, road dimension, ditch, and vegetative parameter details). Greater model specificity is known to lead to greater uncertainty and equifinality (Brazier et al., 2000; Brown et al., 2016). Brazier et al. (2000) evaluated WEPP performance on two data sets and concluded that there was significant uncertainty in input parameters and model structure of WEPP. Skaugset et al. (2011) found that WEPP and RUSLE2 erosion models overestimated soil erosion on roads in the Pacific Northwest region of the United States by 2 to 8 times because of the road surface hydrology was not adequately represented within the models. They suggest that further consideration of the hydrology component is necessary for forest roads. Within the current study, we were able to incorporate observed precipitation data into WEPP-modified, which likely lead to greater model performance compared to all other tested models. The ability to increase the specificity of physical models is promising for the future of soil erosion estimation, but there needs to be better, easier, and enhanced ways to measure the parameters that drive these models. Oreskes et al. (1994) identified parameter scale as an important consideration of model uncertainty. They state that multiple scales are used in the development model parameters, which affect their relationships and are always uncertain.

WEPP-default had the lowest potential to model soil erosion as compared to the other models. Modified versions of WEPP have been successfully applied in previous research studies without characterization of the climate data (Liu et al., 1997; Tiwari et al., 2000). Tiwari et al. (2000) applied WEPP in an uncalibrated method and compared model estimates with USLE and RUSLE. Each model had Nash-Sutcliffe model efficiency measures greater the 0.71, which indicate exceptional model performances. Wade et al. (2012) concluded that the greatest strength and weakness of WEPP is its flexibility of parameters, but that the default parameters may not be suited for a particular intent. Forestry practitioners require a soil erosion model with readily attainable parameters to estimate site specific road conditions. Therefore, WEPP is better suited for researchers and government agencies that have the capability to measure the most influential parameter data. This suggests that additional sensitivity analysis is needed to better understand the influences of default parameters for forest roads within the WEPP model. Erosion models are representations of scientific hypothesis and therefore guide future research. The primary value of WEPP, USLE, and RUSLE2 is heuristic (Oreskes et al., 1994). Incorporating physical models such as WEPP into BMP inspections and assessments of management alternatives would be difficult because of the number of different parameters and calibration required.

The USLE is the most widely used soil erosion models for a variety of sites and management conditions (Kinnell, 2010). The different levels of parameter detail within the USLE-roadway, USLE-soil survey, and RUSLE2 did not improve model estimates when compared to trapped sediment data ($p < 0.10$). Sheridan et al. (2008) simulated rainfall over forest roads and compared mapped erosivity values with observed data and found that observed data for model K-values did not provide greater accuracy than did modeled erosivity values. Using the soil particle size data to calculate the erosivity factor (K-values) in USLE-roadway and RUSLE2 marginally improved model estimates over USLE-soil survey, which simply utilized K-values from the soil survey for site subsoils. USLE-soil survey was the easiest and simplest of the methods evaluated to estimate soil erosion. USLE-soil survey is a useful method for field comparisons compared to RUSLE2, which requires computer access (Christopher and Visser, 2007). However, RUSLE2 offers a database that will allow the user to download and store a variety of soil, climate, and management files that can be used repeatedly thereafter. Compared to the USLE-forest versions, RUSLE2 allows users to incorporate more complex surfacing, slopes, and soils, and observed precipitation files. However, it would appear that WEPP has a greater potential than RUSLE2 to quantify sediment above 11.2 Mg ha^{-1} yr^{-1}.

Empirical models in this study did not perform well to quantify sediment at stream crossings. Empirical models estimate erosion over a long-term (10+ years), continuous basis and were not intended to be applied to single events or short-term estimates as applied in our study. They continue to be applied in such studies because of the ease of use and underlining theory. One of the limitations of the USLE and its modifications for forest road erosion modeling is that its development was based on agricultural soils. Road surfaces are highly compacted and have variable surface roughness ratios depending on traffic level (Sheridan and Noske, 2007). Traffic level and specific BMP are not directly considered in the USLE models and are important factors contributing to soil erosion from forest roads (Reid and Dunn, 1984; Fu et al., 2010). Croke and Nethery (2006) reported that RUSLE overestimated soil loss during a 10-year storm event on skid trails by nearly an order of magnitude. Furthermore, they concluded that further research on the major erosion factors and subfactors are necessary for RUSLE to be applicable to forest road conditions. It is important to note that models give results within the range of conditions from which they were developed and have different levels of utility depending on the intended use and the data available.

We also used this project as an opportunity to assess the utility of the models for identifying stream crossing approaches that reflected different categories of BMP implementation. For example, what level of erosion might occur for forest roads having either poor or excellent levels of BMP implementation? This type of information could be useful to state agency personnel attempting to strategically inspect forest harvest operations or to forest managers who were tasked with ensuring BMP implementation with limited resources. Furthermore, these models could have utility for instructors attempting to instruct students in the general effects of BMP usage on stream sediment. Ideally, a model would provide good estimates and be relatively straightforward to use. Unfortunately, the best model (WEPP-modified) requires extensive parameter adjustments before it performed well. The ability to conduct such adjustment requires extensive knowledge of erosion processes, the WEPP model, and how WEPP responds to adjustments. Such modifications are simply beyond the time constraints and personnel qualifications of many organizations, yet WEPP-default does not appear to be an acceptable substitute. Commonly, managers do not have sufficient time or technical expertise to evaluate and modify model components and change default settings. Therefore, use of the standard WEPP is often too onerous for simple decisions of identifying situations that may require additional BMPs. We evaluated the benefits and problems associated with the remaining three models and concluded that the USLE-forest (i.e. USLE-soil survey and USLE-roadway) is the simplest and most time efficient to apply, while it provided the second best ability to place erosion values within the correct BMP category (table 3.3).

3.6 Conclusions

After several decades of BMP research, there are still numerous information gaps that confound our ability to accurately assess the impact of forestry BMPs on stream sediment and models could help us fill these gaps in an expeditious manner. Research indicates that forestry BMPs generally minimize erosion from forest operations, yet forest stream crossings may have higher sediment delivery potential than all other forest operations. Thus we evaluated the performance of five variations of three common erosion models for operational stream crossing approaches. Despite the importance of stream crossings, there have been few research projects that evaluated the effects of stream crossings across a range of erosion potentials and few projects that have evaluated use of erosion models for stream crossing evaluation. Thus, this project was developed. Our major conclusions are:

1. Soil erosion models evaluated could estimate erosion and sediment delivery within 5 Mg ha^{-1} yr^{-1} for most approaches with erosion rates less than 11.2 Mg ha^{-1} yr^{-1}, while models estimated varied widely for approaches that eroded above 11.2 Mg ha^{-1} yr^{-1}. Based on the results from 12 evaluations of model performance, WEPP-modified consistently ranked higher compared to all other models, yet its performance in estimating erosion above 11.2 Mg ha^{-1} yr^{-1} clearly needs additional improvement. WEPP requires input parameters that are not readily available and easily measured yet, modifications to input parameters are necessary fit the site conditions. Using the default parameter to simulate erosion (WEPP-default) ranked worst in 75% of model performance metrics compared to all other models. WEPP-default over-predicted erosion by $\geq$ 5 Mg ha^{-1} yr^{-1} for 82% of approaches with erosion rate less than 11.2 Mg ha^{-1} yr^{-1} and only correctly placed 43% of erosion estimates into appropriate erosion categories.

2. Additional field evaluations and improvement of erosion models are needed for stream crossings. Sensitivity analyses are needed to better understand the influences of default parameters for forest roads within the WEPP and USLE models.

3. State agencies with significant forestland in the eastern United States conduct BMP inspections for forest road stream crossings. These inspections identify erosion problems and recommend corrective BMPs, yet these agencies are interested in use of models for estimation of road and ditch BMP effectiveness and sediment delivery. Two

models, WEPP-modified and USLE-roadway correctly placed ≥ 70% of approaches into Very Low and Low erosion levels. Due to the complexity of input parameters and scaling issues of WEPP-modified, USLE-roadway may be a more practical choice for many managers and agency personnel. Using the soil survey to obtain K-values may be misleading for roads because road construction changes soil composition. Texturing soils in the field or lab will better characterize the site.

3.7 Acknowledgements

Funding for this work was provided in part by the National Council for Air and Stream Improvement (NCASI), Inc., the McIntire-Stennis Program of the National Institute of Food and Agriculture, and the Virginia Tech Forest Operations and Business Research Cooperative. MeadWestvaco Corporation, Plum Creek Timber Company, Virginia Tech Department of Forest Resources and Environmental Conservation, provided logistical support and the United States Forest Service provided study sites on the Washington/Jefferson National Forest.

3.8 References

Abaci, O., & Papanicolaou, A. N. (2009). Long-term effects of management practices on water-driven soil erosion in an intense agricultural sub-watershed: monitoring and modelling. *Hydrologic. Proc. 23*(19), 2818-2337.

Amore, E., Modica, C., Nearing, M. A., & Santoro, V. C. (2004). Scale effect in USLE and WEPP application for soil erosion computation from three Sicilian basins. *J. Hydrol., 293*(1), 100-114.

Anderson, C. J., & Lockaby, B. G. (2011). Research gaps related to forest management and stream sediment in the United States. *Environ. Mgmt.*, *47*(2), 303-313. doi: 10.1007/s00267-010-9604-1.

Aust, W. M., Bolding, M. C., & Barrett, S. M. (2015). Best Management Practices for Low-Volume Forest Roads in the Piedmont Region: Summary and Implications of Research. *Transport. Res. Record: J. Transport. Res. Board*, (2472), 51-55.

Aust, W. M., Carroll, M. B., Bolding, M. C., & Dolloff, C. A. (2011). Operational forest stream crossings effects on water quality in the Virginia Piedmont. *South. J. Appl. For.*, *35*(3), 123-130.

Aust, W. M., Shaffer, R. M., Burger, J. A. (1996). Benefit and cost of forestry best management practices in Virginia. *South. J. Appl. For.*, *20*(1), 23-29.

Bennett, N. D., Croke, B. F. W., Guariso, G., Guillaume, J. H. A., Hamilton, S. H., Jakeman, A. J., . . . Andreassian, V. (2013). Characterising performance of environmental models. *Environ. Model. Software*, *40*, 1-20. doi: 10.1016/j.envsoft.2012.09.011.

Boston, K. (2012). Impact of the Ninth Circuit Court ruling (Northwest Environmental Defense Center v. Brown) regarding forest roads and the Clean Water Act. *J. For.*, *110*(6), 344-346. doi: 10.5849/jof.11-069.

Brazier, R. E., Beven, K. J., Freer, J., & Rowan, J. S. (2000). Equifinality and uncertainty in physically based soil erosion models: application of the GLUE methodology to WEPP–the Water Erosion Prediction Project–for sites in the UK and USA. *Earth Surface Processes and Landforms*, *25*(8), 825-845.

Brown, K. R., Aust, W. M., & McGuire, K. J. (2013). Sediment delivery from bare and graveled forest road stream crossing approaches in the Virginia Piedmont. *For. Ecol. Mgmt., 310,* 836-846. doi: 10.1016/j.foreco.2013.09.031.

Brown, K. R., McGuire, K. J., Hession, W. C., & Aust, W. M. (2016). Can the water erosion prediction project model be used to estimate best management practice effectiveness from forest roads? *J. For., 114*(1), 17-26. doi: 10.5849/jof.14-101.

Christopher, E. A., & Visser, R. (2007). Methodology for evaluating post harvest erosion risk for the protection of water quality. *New Zealand J. For., 52*(2), 20.

Croke, J., & Nethery, M. (2006). Modelling runoff and soil erosion in logged forests: Scope and application of some existing models. *Catena, 67*(1), 35-49. doi: 10.1016/j.catena.2006.01.006.

Croke, J., Mockler, S., Fogarty, P., & Takken, I. (2005). Sediment concentration changes in runoff pathways from a forest road network and the resultant spatial pattern of catchment connectivity. *Geomorphology, 68*(3-4), 257-268. doi: 10.1016/j.geomorph.2004.11.020.

Dissmeyer, G. E., & Foster, G. R. (1984). A guide for predicting sheet and rill erosion on forest land. *General Tech. Publ. R8-TP 6*. Washington, D.C.: USDA Forest Service.

Dun, S., Wu, J. Q., Elliot, W. J., Robichaud, P. R., Flanagan, D. C., Frankenberger, J. R., ... & Xu, A. C. (2009). Adapting the Water Erosion Prediction Project (WEPP) model for forest applications. *J. Hydrol., 366*(1), 46-54.

Edwards, P. J., Schoonover, J. E., & Willard, K. W. J. (2015). Guiding principles for management of forested. agricultural, and urban watersheds. *J. Contemp. Water Res. Edu., 154*(4), 60-84.

Elliot, W. J. (2000). Roads and other corridors. In *Drinking water from forests and grasslands: a synthesis of the scientific literature,* Dissmeyer, George E. (ed.), Gen. Tech. Rep. SRS-39, Asheville, NC. USDA Forest Service. Southern Research Station, 85-101.

Elliot, W. J. (2013). Erosion processes and prediction with WEPP technology in forests in the Northwestern US. *Trans. ASABE, 56*(2), 563-579.

Elliot, W. J., & Hall, D. E. (1997). Water erosion prediction project (WEPP) forest applications. *General Tech. Publ.* Ogden, UT: USDA Forest Service, Intermountain Research Station.

Elliot, W. J., (2004). WEPP internet interfaces for forest erosion prediction. *J. American Water Resource. Assoc., 40*(2), 299-309. doi: 10.1111/j.1752-1688.2004.tb01030.x.

Elliot, W. J., Foltz, R. B., & Luce, C. H. (1995). Validation of water erosion prediction project (WEPP) model for low-volume forest roads. *In Proc. 6th Intl. Conf. Low-Volume Roads*, (pp. 178-186). Minneapolis, Minn.

Environmental Protection Agency (EPA). (2005). National management measures to control nonpoint source pollution from forestry.pp. 276. *EPA-841-B-05-001*. Washington, D.C.: Office of Water.

Environmental Protection Agency (EPA). (2012). Revisions to stormwater regulations to clarify that an NPDES permit is not required for stormwater discharges from logging roads. *Federal Register*, *77*(236), 72970-72975.

Environmental Protection Agency (EPA). (2015). Notice of opportunity to provide information on existing programs that protect water quality from forest road discharges. *Federal Register*, *80*(217), 69653-69660.

Flanagan, D. C., Frankenberger, J. R., & Ascough, J. C. (2012). WEPP: Model use, calibration, and validation. *Trans. ASABE, 55*(4), 1463-1477.

Flanagan, D. C., Gilley, J. E., & Franti, T. G. (2007). Water Erosion Prediction Project (WEPP): Development history, model capabilities, and future enhancements. *Trans. ASABE, 50*(5), 1603-1612.

Foltz, R. B., Copeland, N. S., & Elliot, W. J. (2009). Reopening abandoned forest roads in northern Idaho, USA: quantification of runoff, sediment concentration, infiltration, and interrill erosion parameters. *J. Environ. Mgmt., 90*(8), 2542-2550. doi: 10.1016/j.jenvman.2009.01.014.

Forsyth, A. R., Bubb, K. A., & Cox, M. E. (2006). Runoff, sediment loss and water quality from forest roads in a southeast Queensland coastal plain Pinus plantation. *For. Ecol. Mgmt., 221*(1-3), 194-206. doi: 10.1016/j.foreco.2005.09.018.

Fu, B., Newham, L. T., & Ramos-Scharron, C. (2010). A review of surface erosion and sediment delivery models for unsealed roads. *Environ. Model. Software, 25*(1), 1-14.

Gee, G. W., & Or, D. (2002). Particle-size analysis. In *Methods of soil analysis. Part 4. Physical methods*, (pp. 278-284). Madison, Wis.: Soil Science Society of America Inc.

Grossman, R. B., & Reinsch, T. G. (2002). Bulk Density and Linear Extensibility, Core Method. In *Methods of soil analysis. Part 4. Physical methods*, (pp. 207-210). Madison, Wis.: Soil Science Society of America Inc.

Gucinski, H., Furniss, M. J., Ziemer, R. R., Brookes, M. H. (2001). Forest roads: a synthesis of scientific information. Gen. Tech. Rep. PNW-GTR-509. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 103 p.

Ice, G. (2004). History of innovative best management practice development and its role in addressing water quality limited waterbodies. *J. Environ. Eng., 130*(6), 684-689.

Jackson, C., Martin, J., Leigh, D., & West, L. (2005). A southeastern piedmont watershed sediment budget: Evidence for a multi-millennial agricultural legacy. *J. Soil Water Cons., 60*(6), 298-310.

JMP®, Version 11. (2016). SAS Institute Inc., Cary, NC. 1989-2016.

Jones, J. I., Murphy, J. F., Collins, A. L., Sear, D. A., Naden, P. S., & Armitage, P. D. (2012). The impact of fine sediment on macro-invertebrates. *River Res. Applic., 28*(8), 1055-1071. doi: 10.1002/rra.1516.

Kinnell, P. I. A. (2010). Event soil, runoff and the Universal Soil Loss Equation family of models: A review. *J. Hydrol., 385*, 384-397.

Kochenderfer, J., & Helvey, J. (1987). Using gravel to reduce soil losses from minimum-standard forest roads. *J. Soil Water Cons.*, *42*(1), 46-50.

Laflen, J. M., Elliot, W., Flanagan, D., Meyer, C., & Nearing, M. (1997). WEPP-predicting water erosion using a process-based model. *J. Soil Water Cons.*, *52*(2), 96-102.

Laflen, J. M., Flanagan, D. C., & Engel, B. A. (2004). Soil erosion and sediment yield prediction accuracy using WEPP. *J. American Water Resource. Assoc.*, *40*(2), 289-297.

Lane, P. N. J., & Sheridan, G. J. (2002). Impact of an unsealed forest road stream crossing: water quality and sediment sources. *Hydrologic. Proc.*, *16*(13), 2599-2612. doi: 10.1002/hyp.1050.

Liu, B. Y., Nearing, M. A., Baffaut, C., & Ascough II, J. C. (1997). The WEPP watershed model: III. Comparisons to measured data from small watersheds. *Trans. ASABE*, *40*(4), 945-952.

Luce, C. H., & Black, T. A. (1999). Sediment production from forest roads in western Oregon. *Water Resource. Res.*, *35*(8), 2561-2570. doi: 10.1029/1999wr900135.

MacCurdy, M. G., & Timmons, D. L. (2013). Questions remain for the timber industry after Supreme Courts decision in Decker v. Northwest Environmental Defense Center. *Envtl. L.*, *43*, 827.

Madej, M. A. (2001). Erosion and sediment delivery following removal of forest roads. *Earth Surf. Proc. Land.*, *26*(2), 175-190.

Megahan, W. F., Wilson, M., & Monsen, S. B. (2001). Sediment production from granitic cutslopes on forest roads in Idaho, USA. *Earth Surf. Proc. Land.*, *26*(2), 153-163.

Merritt, W. S., Letcher, R. A., & Jakeman, A. J. (2003). A review of erosion and sediment transport models. *Environ. Model. Software, 18*(8), 761-799.

Moore, A. D., McLaughlin, R. A., Mitasova, H., & Line, D. E. (2007). Calibrating WEPP model parameters for erosion prediction on construction sites. *Trans. ASABE, 50*(2), 507-516.

Morris, B., Bolding, M. C., and Aust, W. M. (2014). Sediment contribution of haul road culverts in Virginia. (pp. 7). Moline, Ill.: COFE.

National Oceanic and Atmospheric Administration (NOAA). (2015). 1981-2010 U.S. Climate Normals. Retrieved from https://www.ncdc.noaa.gov/data-access/land-based-station-data/land-based-datasets/climate-normals/1981-2010-normals-data.

National Research Council (U.S.) (NRC). (2008). Committee on Hydrologic Impacts of Forest Management. *Hydrologic effects of a changing forest landscape*. Washington, D.C: National Academies Press

Nolan, L., Aust, W. M., Barrett, S. M., Bolding, M. C., Brown, K., & McGuire, K. (2015). Estimating costs and effectiveness of upgrades in forestry best management practices for stream crossings. *Water, 7*(12), 6946-6966.

Oreskes, N., Shrader-Frechette, K., & Belitz, K. (1994). Verification, validation, and confirmation of numerical models in the earth sciences. *Science*, *263*(5147), 641-646.

Pechenick, A. M., Rizzo, D. M., Morrissey, L. A., Garvey, K. M., Underwood, K. L., & Wemple, B. C. (2014). A multi-scale statistical approach to assess the effects of connectivity of road and stream networks on geomorphic channel condition. *Earth Surf. Proc. Land., 39*(11), 1538-1549.

Reid, L. M., & Dunn, T. (1984). Sediment production from forest road surfaces. *Water Resource. Res.*, *20*(11), 1753-1761.

Renard, K. G., Foster, G. R., Weesies, G. A., & Porter, J. P. (1991). RUSLE: Revised universal soil loss equation. *J. Soil Water Cons.*, *46*(1), 30-33.

Renard, K. G., Foster, G. R., Weesies, G. A., McCool, D. K., & Yoder, D. C. (1997). Predicting soil erosion by water: A guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). *Handbook, 703*, (pp. 404). Washington, D.C.: USDA Agriculture Research Service.

Robichaud, P. R., & Brown, P. R. (2002). Silt fences: An economical technique for measuring hillslope soil erosion. *General Tech. Publ. RMRS-GTR-94*. Washington, D.C.: USDA Forest Service.

Sawyers, B., Bolding, M. C., Aust, W. M., & Lakel, W. (2012). Effectiveness and implementation costs of overland skid trail closure techniques in the Virginia Piedmont. *J. Soil Water Cons., 67*(4), 300-310.

Shepard, J. P. (2006). Water quality protection in bioenergy production: the US system of forestry best management practices. *Biomass Bioenergy*, *30*(4), 378-384. doi: 10.1016/j.biombioe.2005.07.018.

Sheridan, G. J., & Noske, P. J. (2007). Catchment-scale contribution of forest roads to stream exports of sediment, phosphorus and nitrogen. *Hydrol. Proc.*, *21*(23), 3107-3122. doi: 10.1002/hyp.6531.

Sheridan, G. J., Noske, P. J., Lane, P. N. J., & Sherwin, C. B. (2008). Using rainfall simulation and site measurements to predict annual interrill erodibility and phosphorus generation rates from unsealed forest roads: Validation against in-situ erosion measurements. *Catena*, *73*(1), 49-62.

Simanton, J. R., Weltz, M. A., & Larsen, H. D. (1991). Rangeland experiments to parameterize the water erosion prediction project model: vegetation canopy cover effects. *J. Rang. Mgmt.*, *44*(3), 276-282.

Skaugset, A., Surfleet, C., Meadows, M., & Amann, J. (2011). Evaluation of erosion prediction models for forest roads. *Transport. Res. Record: J. Transport. Res. Board*, (2203), 3-12.

Society of American Foresters (SAF) Task Force on Reauthorization of the Clean Water Act, and Society of American Foresters. 1995. *Reauthorization of the clean water act : Forestry effects on water quality : A report to the membership of the Society of American Foresters*, SAF 95-10, Besthesda, MD, 24 p.

Stuart, G. W., & Edwards, P. J. (2006). Concepts about forests and water. *North. J. Appl. For.*, *23*(1), 11-19.

Swift, L. W., & Burns, R. G. (1999). The three Rs of roads: redesign, reconstruction, and restoration. *J. For.*, *97*(8), 40-44.

Taylor, S. E., Rummer, R. B., Yoo, K. H., Welch, R. A., & Thompson, J. D. (1999). What we know--and don't know--about water quality at stream crossings. *J. For.*, *97*(8), 12-17.

Tiwari, A., Risse, L., & Nearing, M. (2000). Evaluation of WEPP and its comparison with USLE and RUSLE. *Trans. ASAE*, *43*(5), 1129-1135.

Toy, T., Foster, G., & Renard, K. (1999). RUSLE for mining, construction and reclamation lands. *J. Soil Water Cons.*, *54*(2), 462-467.

United States Department of Agriculture, Agriculture Research Service (USDA ARS). (2008). User Reference Guide for Revised Universal Soil Loss Equation, Version 2 (RUSLE2). Draft. (pp. 444). Washington, D.C.: USDA-Agriculture Research Service.

United States Department of Agriculture, National Resources Conservation Service (USDA NRCS). (2011). Updated T and K Factors Q & A – Wisconsin. (pp. 2).

United States Department of Agriculture, National Resources Conservation Service (USDA NRCS). (2012). Soil Survey. Retrieved from Soil Survey Staff http://websoilsurvey.nrcs.usda.gov/.

United States Department of Agriculture, National Resources Conservation Service (USDA NRCS). (2015). Official NRCS Climate Database. Retrieved from http://fargo.nserl.purdue.edu/RUSLE2_ftp/Climate_data/.

Virginia Department of Forestry (VDOF). (2011). Virgina's Forestry Best Management Practices for Water Quality: Technical Manual. Fifth ed. Charlottesville, Va.: Virginia Department of Forestry.

Wade, C., Bolding, M., Aust, W., Lakel, W., & Schilling, E. (2012). Comparing sediment trap data with the USLE-Forest, RUSLE2, and WEPP-Road erosion models for evaluation of bladed skid trail BMPs. *Trans. ASABE, 55*(2), 403-414.

Walbridge, T. A. (1997). *The location of forest roads*. (pp. 91). Blacksburg, Va: Walbridge and Associates.

Wear, L. R., Aust, W. M., Bolding, M. C., Strahm, B. D., and Dolloff, C. A. (2013). Effectiveness of best management practices for sediment reduction at operational forest stream crossings. *For. Ecol. Mgmt., 289*, 551-561.

Weather Underground Service (WU). (2013). Weather history for June 2012 through December 2013. Retrieved from https://www.wunderground.com/history.

Zar, J. H. (2010). *Biostatistical Analysis*. (pp. 960). Upper Saddle River, NJ: Pearson-Prentice Hall.

4.0 Forestry best management practices for erosion control in haul road ditches near stream crossings

A.J. Lang[a], W.M. Aust[b], M.C. Bolding[c], K.J. McGuire[d], and E.B. Schilling[e]

4.1 Abstract

Poorly designed and maintained forest road stream crossings can directly link erosion sources to streams. Forestry Best Management Practices (BMPs) provide techniques that are useful for preventing sedimentation associated with ditch erosion. However, few studies have quantified ditch BMP sediment reductions. Thus, our primary objective was to evaluate erosion control effectiveness due to ditch BMPs and secondarily to quantify ditch BMP implementation costs. Sixty ditch segments, near stream crossings were reconstructed and five ditch BMP treatments were applied using a completely randomized design resulting in 11-13 replications per treatment. Ditch BMP treatments were (1) bare ditch (Bare), (2) grass seed with lime fertilizer (Seed), (3) grass seed with lime fertilizer and erosion control mat (Mat), (4) rock check dams (Dam), and (5) completely rocked (Rock). Silt fence sediment traps and sediment pins were measured over one year to determine treatment effectiveness. Trapped sediment deposits indicated

[a] Graduate Research Assistant, Department of Forest Resources and Environmental Conservation (FREC), Virginia Tech, 305 Cheatham Hall, 310 West Campus Drive, Blacksburg, Virginia, 24061.
[b] Professor of Forestry, FREC, Virginia Tech, 228 Cheatham Hall 310 West Campus Drive, Blacksburg, Virginia, 24061.
[c] Associate Professor of Forest Engineering FREC, Virginia Tech, 228 Cheatham Hall 310 West Campus Drive, Blacksburg, Virginia, 24061.
[d] Associate Professor of Hydrology, FREC and Virginia Water Resource Research Center, Virginia Tech 210 Cheatham Hall, 310 West Campus Drive, Blacksburg, Virginia, 24061.
[e] Senior Research Scientist, National Council for Air and Stream Improvement Inc., Aubrey Texas 76227.

that median erosion rates were greatest for Dam (6.14 Mg ha^{-1} y^{-1} [2.74 ton ac^{-1} yr^{-1}]), followed by Bare (4.92 Mg ha^{-1} y^{-1} [2.19 ton ac^{-1} yr^{-1}]), Rock (1.73 Mg ha^{-1} y^{-1} [0.77 ton ac^{-1} yr^{-1}]), Seed (1.04 Mg ha^{-1} y^{-1} [0.46 ton ac^{-1} yr^{-1}]), and Mat (0.82 Mg ha^{-1} y^{-1} [0.37 ton ac^{-1} yr^{-1}]). Results suggested that Mat treatments had significantly lower erosion rates than Bare and Dam, while Rock and Seed provided intermediate levels. Costs of BMP treatments were least expensive for Seed (\$6.10 approach^{-1}), followed by Mat (\$21.33 approach^{-1}), Dam (\$71.43 approach^{-1}), and Rock (\$141.08 approach^{-1}). Results suggest that erosion began to accelerate disproportionately when bare soil levels were between 30-50%, therefore minimum soil cover of 50% is recommended for ditches.

4.2 Introduction

Sediment is the most prevalent non-point source (NPS) pollutant from forest management activities (Patric 1976; Neary et al. 1988; Binkley and Brown 1993; Jackson et al. 2005; Edwards and Williard 2010). Sediment can negatively affect aquatic biota as an individual pollutant and may indirectly transport nutrients, pesticides, and other water pollutants (Wood and Armitage 1997). Extensive sediment deposition in streams can take years to export (Jackson et al. 2005), which can affect the recovery process of stream habitats (Jones et al. 2011). Erosion rates from mature forestlands are typically 0.11 to 0.22 Mg ha^{-1} y^{-1} (0.05 to 0.10 ton ac^{-1} yr^{-1}), which is lower than reported geologic norm erosion rates (0.40 to 0.67 Mg ha^{-1} y^{-1} [0.18 to 0.30 ton ac^{-1} yr^{-1}]) (Patric 1976). With regard to silvicultural operations, roads, skid trails, log decks, and stream crossings erode more readily than the general harvest area and can deliver sediment to streams if adequate erosion control measures are not implemented (Motha et al. 2003). Motha et al. (2003) found that unsealed roads contributed 20 to 60 times more sediment to streams than undisturbed forests on a per unit area basis in an Australian watershed. Conversely, careful forest operations can minimally affect water quality (Shepard 2006; McBroom et al. 2008). Yoho (1980) reviewed erosion rates from forest operation activities in the southern US and found that careful logging operations had low overall erosion rates ranging from 0.13 to 0.38 Mg ha^{-1} y^{-1} (0.06 to 0.17 ton ac^{-1} yr^{-1}). Barrett et al. (2016) estimated erosion from conventional and biomass harvests in Virginia, US and found that overall rates were less than 1.8 Mg ha^{-1} y^{-1} (0.80 ton ac^{-1} yr^{-1}).

Forest roads typically have exposed soil and may alter hillslope hydrology, which may affect the timing, quantity, and pathways of water flow through catchments (Dymond et al. 2014). The degree to which roads alter hydrologic connections fluctuates over time and varies due to road location, design, and use (Croke et al. 2005). Wemple et al. (1996) assessed hydrologic connections of roads to streams within two catchments located in the western Cascades of Oregon, US. Using a geographic information system, they estimated that 57% of road lengths connected to the stream network by ditches. Bilby et al. (1989) identified drainage points where road runoff reached streams within three forested watersheds in Washington, US. They found that approximately 34% of the road area delivered sediment to streams. Researchers have also reported low sediment delivery potentials from hydrologically connected roads because BMPs, such as special hardening of sensitive road segments, reduce sediment impacts (Martin 2009).

Potential sediment issues associated with forest roads have been recognized for decades (Kraebel 1936; Bailey 1948; Trimble and Sartz 1957). Current forest management activities routinely address such issues using a variety of erosion control practices and measures known as Best Management Practices (BMPs) (Anderson and

Lockaby 2011a). Development of forestry BMPs followed enactment of the Federal Water Pollution Control Act (i.e., Clean Water Act) and subsequent amendments (Aust and Blinn 2004). State forestry agencies continue to modify and improve recommendations for BMPs based on applicable research findings. Numerous researchers have demonstrated the effectiveness of BMPs for erosion control on forest roads and stream crossings. Specific examples include stream crossing structures (Aust et al. 2011), stream crossing approach surfaces (Brown et al. 2013), stream crossing closure techniques (Wear et al. 2013), streamside management zone widths (Lakel et al. 2010), skid trail closure techniques (Sawyers et al. 2012), road surfacing (Clinton and Vose 2003), cutslope and fillslope covers (Grace et al. 1998), road turnouts (Grace 2002), and road decommission techniques (Foltz 2012). Several extensive reviews of forestry BMP literature report that water quality effects are minimized when BMP guidelines are properly implemented (Aust and Blinn 2004; Ice 2004; Shepard 2006; Anderson and Lockaby 2011a; Cristan et al. 2016).

Researchers often attribute accelerated erosion rates and potential water quality issues from forested land use to poorly located roads, recently constructed roads prior to stabilization, stream crossing approaches with inadequate best management practices (BMPs), and/or poorly maintained road systems (Luce and Wemple 2001; Lang et al., 2015a; Edwards et al., 2015). Litschert and MacDonald (2009) assessed nearly 200 harvest units in Sierra Nevada and Cascade Mountains of California, US and found that the majority of connected sediment pathways to streams were initiated from poorly located skid trails. They recommended that sediment delivery could be reduced by locating skid trails further away from streams. Increases in sediment yield have been associated with new road construction for harvesting operations, particularly at stream crossings (Beschta 1978). Swank et al. (2001) found that a newly constructed road (before the hydroseed treatment could germinate) contributed approximately 45 Mg (50 ton) of sediment to the stream during one intense rainfall event at the Coweeta Laboratory in North Carolina, US. Aust et al. (2011) evaluated water quality parameters above and below 23 newly constructed haul road stream crossings in the Virginia Piedmont before timber harvests, during stream crossing installations, during timber harvests, and following timber harvest closures. They found a nearly 19x increase in mean total dissolved solid levels and a 75x increase in mean sediment levels during stream crossing installations compared to background levels, but sediment levels dropped following closure BMP application. They concluded that implementing BMPs on the stream crossing approaches immediately following installation and BMP maintenance during the timber harvest as opposed to after harvest completion would improve water quality. Lane and Sheridan (2002) estimated that the installation of a culvert stream crossing within a steep catchment in Australia contributed 1.8-2.7 Mg (2-3 tons) of bedload material. However, they concluded that basic BMPs could have been used to reduce sediment contributions during the stream crossing installation.

Stream crossings are perhaps the most susceptible segment of the road system for sediment delivery (Swift 1985; Wear et al. 2013; Brown et al. 2014; Edwards et al. 2015). Brown et al. (2013) reported sediment delivery rates between 10 to 16 Mg ha^{-1} y^{-1} (4.5 to 7.1 ton ac^{-1} yr^{-1}) for legacy road stream crossing approaches with gravel surfacing and 34 to 287 Mg ha^{-1} y^{-1} (15.2 to 128 ton ac^{-1} yr^{-1}) from legacy road stream crossing approaches without gravel surfacing. Nolan et al. (2015) estimated mean potential erosion rates of 11.9 and 24.2 Mg ha^{-1} y^{-1} (5.3 and 10.8 ton ac^{-1} yr^{-1}) on haul road and skid trail stream crossing approaches, respectively. Cristan et al. (forthcoming) surveyed state forestry agencies in all 50 states to compare monitoring strategies and implementation of forestry BMPs. They found that state regulation and overall BMP implementation rates were not significantly different for 32 states that reported implementation data. However, BMP implementation rates were significantly higher for the stream crossing category in western (90.4%) and southern states (89.2%) compared to northern states (80.1%). Limited information is available for either the number of existing or newly installed stream crossings associated with forest operations, which remains of interest to define the potential scope of erosion and sediment delivery from silvicultural stream crossings. Morris (2014) used Google Earth to estimate haul road stream crossings and concluded that approximately 774 new crossings were constructed in Virginia over one year.

Road maintenance and traffic disturb the road surface and can influence sediment production (Forsyth et al. 2006). Reid and Dunne (1984) found that a heavily used road contributed 130 times more sediment than an abandoned road. Bibly et al (1989) reported that the amount of sediment produced from a road segment on an hourly basis was related to traffic. Ziegler et al. (2001) reported that traffic disturbance to the road surface generated loose material that was subsequently eroded. Sheridan and Noske (2007) reported greater sediment delivery rates from well-graveled roads with less traffic than well-graveled roads with infrequent traffic. Furthermore, traffic level on the graveled roads explained 97% of the variation in annual sediment delivery. Drainage control is another important consideration regarding sediment production from forest roads.

Drainage control structures such as cross drains, broad based dips, rolling dips, water turnouts, and waterbars can be implemented to minimize the sediment load by directing water to the forest floor (North Carolina Forest Service 2014). By decreasing the water volume with frequent drainage structures, the shearing force is lessened and less soil is eroded (Forsyth et al., 2006).

BMPs for roadside ditches are important because ditches can be a significant source of sediment due to carrying concentrated overland flow that can have substantial erosive energy (Swift 1985). Insloped and crowned forest road templates are common designs that typically utilize ditches (Moll et al. 1997). Ditch maintenance is necessary to maintain drainage, but may result in increased erosion due to soil disturbance or removal of vegetative or other soil covers. Removing material within ditches loosens soil particles and makes them available for transport (Croke and Hairsine 2006). Luce and Black (1999) reported that removing vegetation from the ditch and cutslope increased sediment productivity by 7.4 fold. Researchers have examined various vegetative and rock stabilization techniques on cutslopes, fillslopes, and ditches to control erosion (Grace et al. 1998; Luce and Black 1999; Megahan et al. 2001). Vegetative and rock treatments can stabilize soil, but the extent of stabilization may vary over time and is heavily dependent on site conditions (Grace 1998; Swank et al. 2001). Overall, dispersing runoff from the road and ditch in small quantities is key for effective erosion control, since runoff volume and velocity drive erosive forces (Edwards et al. 2015).

Mulch materials are commonly recommended as a ditch BMP, yet sediment production data for such treatments are primarily from road surfaces (Edwards et al. 2015). Foltz (2012) compared sediment production rates among wood shred, wood strands, and straw mulches on decommissioned roads in the northern Rocky Mountains. Only the wood strand treatment during the first year following road decommission had significantly less sediment production than the control. Effective ground cover between 35% and 58% caused statistically indistinguishable sediment production from erosion plots. Seed and mulch are another commonly recommended ditch BMP treatment based on their effectiveness on forest roads and cutslopes. Megahan et al. (2001) evaluated erosion on newly constructed granite cutslopes on similar standard forest roads in Idaho. Treatments included dry seed on alluvial and shallow soils, hydroseed plus mulch, hydroseed plus mulch with terraces, and bare soil control measures. Dry seeding on the alluvial soil reduced erosion compared to all other treatments, but only 5% of the road length was appropriate for this treatment. Hydroseed plus mulch had significantly greater erosion than dry seed on alluvial soil, but had the lowest mean erosion rate of the remaining treatments and was the most cost effective treatment for 95% of the road.

Gravel or stone applications are recommended as ditch BMP erosion control methods and stone surfaces have proved to be effective for road erosion control. Clinton and Vose (2003) compared total suspended solids (TSS) along sediment pathways stemming from paved and graveled road sections to streams in the southern Appalachians. Paved road segments produced significantly less TSS than the unimproved graveled road segments. However, all road types had decreased TSS concentrations with distance from the road. Therefore, they concluded that dispersing runoff frequently and away from streams has significant implications for water quality. Accelerated erosion and sediment delivery rates have been reported from road systems with inadequate surfacing.

Sheridan and Noske (2007) measured sediment delivery from unpaved road surfaces in Australia and reported a maximum delivery rate of 37 Mg ha^{-1} yr^{-1} (84 ton ac^{-1} yr^{-1}).

Overall, a review of the literature indicates that forest roads, specifically road segments with ditched stream crossing approaches, have the strongest potential to introduce sediment to streams if appropriate BMPs are not correctly applied. BMPs for ditched forest roads are important for reducing runoff and controlling erosion, yet few research studies have compared the effectiveness of ditch BMP treatments. Such information would aid forest managers by providing insight into the effectiveness of specific BMPs. Therefore, the primary objective of this research was to evaluate the erosion control effectiveness of four commonly recommended ditch BMPs on forest haul road stream crossing approaches. A secondary objective was to quantify and compare costs associated with different BMP implementation to sediment reduction.

4.3 Materials and Methods

4.3.1 Study Sites

Sixty ditched road segments were selected along three insloped forest haul roads in the Ridge and Valley physiographic region of Virginia (Montgomery County). The Ridge and Valley region is characterized by broad valleys separated by long linear ridges with low to moderate slopes (USDA NRCS 2016). Typical January temperatures range from -6°C to 5°C (21°F to 41°F) and July temperatures from 16°C to 28°C (61°F to 82°F). Average annual precipitation from historic weather data (1981-2010) is 103.9 cm (40.9 in) occurring over 130 days (NOAA 2015). Total daily rainfall data were collected during the study period from a personal weather station affiliated with the Weather Underground Service. Two roads are managed by Virginia Tech (Water Tower Road and

Cassidy's Ridge Road). The third road is on the US Forest Service Washington/Jefferson National Forest (Turkey Nest Road). Each road is gated and vehicle access is restricted. Cassidy's Ridge Road and Water Tower Road access the Virginia Tech Fishburn Forest and have served several timber harvests. The Water Tower road also serves access for maintenance of municipal water and private cellphone towers. Cassidy's Ridge Road served as the main access for a 12.5 ha (31 ac) timber harvest from November 2014 to March 2015. During this time, 37 loaded log trucks traveled across the road (Lang et al. 2015b). Soil series along Cassidy's Ridge and Water Tower roads are Berks and Weikert soils and Berks-Clymer complex (USDA NRCS 2016). The soil survey indicated that the hazard ratings of soil loss from unsurfaced roads and trails were severe and moderate for Berks and Weikert soils and Berks-Clymer complex, respectively (USDA NRCS 2016). The current purposes of Turkey Nest road are crop tree release activities, small firewood cuttings, and recreation (restricted public vehicle access), although the road was originally constructed and used for timber harvests. Soil series along Turkey Nest Road were Jefferson extremely stony soil, Berks and Weikert soil, and Berks-Weikert complex (USDA NRCS 2016). Hazard ratings of soil loss from unsurfaced roads and trails were severe for each of these soil series. Severe ratings infer that road design must be carefully considered and that frequent road maintenance may be required. Additionally, costly erosion control measures may be necessary to maintain proper drainage and trafficability (USDA NRCS 2016).

4.3.2 Experimental Design

A John Deere 450 bulldozer and a New Holland TN750 Farm tractor were used to reconstruct sixty 15.2 m (50 ft) segments of ditch at culvert stream crossings between

May 28 and June 17, 2014. Ditches were reshaped and soil and organic debris were removed away from streams (upslope) within the ditch to improve ditch function, road drainage, and trafficability. Ditches were reconstructed according to site characteristics, but all were cleared to a minimum of approximately 15.2 m (50 ft) along each stream crossing approach. Where feasible, a rolling dip (shallow road surface drainage structure) was installed at the end of treatments furthest from the stream crossing to reduce road drainage area (Figure 4.1).

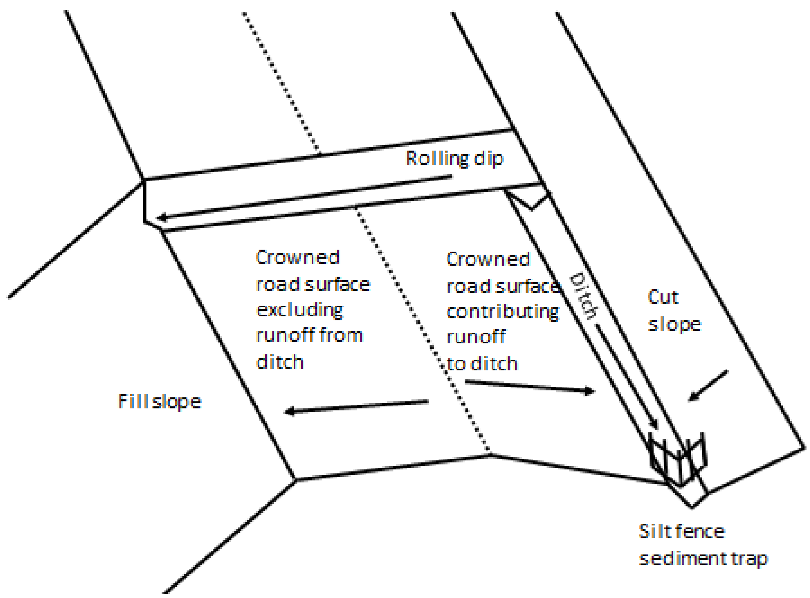

Figure 4.1. Idealized schematic of surface drainage to catchment areas.

A silt fence catchment was placed within each ditch section prior to the stream crossing (Robichaud and Brown 2002). A network of sediment pins were installed adjacent to the silt fence to allow for periodic measurement of sediment deposit depths and were used to estimate sediment delivery (Figure 4.2). Similar studies have used this technique to estimate sediment delivery (Robichaud and Brown 2002; Lakel et al. 2010; Brown et al. 2013; Lang et al. *In Review*).

Figure 4.2. Silt fence catchment area with rebar pins marking the location of measurement. Photo taken October 31, 2014 (about five months after study installation).

4.3.3 Ditch Treatments

One of five treatments was applied to each ditch section. Ten stream crossing approaches had ditch lengths over 30.4 m (100 ft). For these double segment situations, we took advantage of the additional segment and applied the same BMP treatment to the first and second 15.2 m (50 ft) section. For the sixty ditch segments, treatments were: (1) bare ditch (Bare), (2) grass seed with lime fertilizer (Seed), (3) grass seed with lime fertilizer and erosion control mat (Mat), (4) rock check dams (Dam), and (5) completely rocked (Rock) (Figure 4.3). Each treatment was replicated 11-13 times. Seed and Mat treatments were applied by hand spreading a Forest Service recommended grass seed and lime mixture that consisted of orchard grass (73 PLS [Pure Live Seed] kg ha^{-1} [65 PLS lbs ac^{-1}]), annual rye (28 PLS kg ha^{-1} [25 PLS lbs ac^{-1}]), white clover (2.2 PLS kg ha^{-1} [2 PLS lbs ac^{-1}]), and pelletized lime (336 kg ha^{-1} [300 lbs ac^{-1}]). Mat treatments were covered with straw erosion control mats, which were secured within the ditch with landscape staples. Seed and Mat treatments were seeded and fertilized only once following ditch reconstruction. Unwashed number 1 Surge stone (8.9-10.2 cm, 3.5-4 in) was used for Rock and Dam treatments. Stone was purchased from a local quarry and delivered on site. Rock and Dam treatments were applied using the front-end loading bucket of a New Holland TN750 farm tractor. Dam treatments consisted of two rock check dams that were constructed to Virginia BMP guideline specifications (VDOF 2011). Check dams were each approximately 1.5 m (5 ft) long and placed at 3.8 and 11.4 m (12.5 and 37.5 ft) distances from the catchment area.

Figure 4.3. Representative ditch best management practices on haul roads stream crossing approaches. Seed (top-left), Mat (top-right), Dam (bottom-left), Rock (bottom-right), and Bare (middle) treatments.

Ditch reconstruction and BMP installation was performed by Virginia Tech personnel. Material costs and person-hours were recorded and compiled. Quoted rates from a local excavating contractor for a bulldozer and tractor with operators were US$115 h^{-1} and US$65 h^{-1}, respectively. Laborer rates used for BMP applications were US$17.19 h^{-1}, which were based on the average 2014 national occupational employment and wage estimate rate for construction laborers (BLS 2014). Expenses for BMP treatments were converted to a cost per 15.2 m (50 ft) stream crossing approach.

4.3.4 Road Prism Attributes and Sediment Measures

Following ditch reconstruction and treatment installation, we surveyed road prism dimensions using a total station (Sokkia total station model SET-520, Tokyo, Japan). A total station is an electronic transit that measures distances and angles and calculates slopes and curvatures. Road prism measurements included: reconstructed ditch length and slope; road prism area draining to ditch; area of travelway; area and maximum height and slope of cutslope; and percent slope from the edge of the travelway to ditch bottom (mean of two measures) (Table 4.1). Additional road characteristics measured included: surface roughness (Saleh 1993), soil particle size within the ditch and on the travelway (Gee and Or 2002), road shape (convex, concave, uniform, or S-shaped), and percent bare soil (Brown et al. 2013). During each sediment measurement period, percentages of bare soil were estimated from the ditch, travelway, and cutslope.

Table 4.1. Summary statistics of site attributes for the 60 experimental units.

Site Attributes	Units	Max	Mean	Median	Min	SD
Ditch length	(m)	25.6	16.6	15.7	13.5	2.5
Ditch slope	(%)	16.1	7.5	7.5	0.1	4.1
Cutslope area	(m^2)	78.1	43.2	44.0	13.8	17.6
Travelway area	(m^2)	423.2	54.7	44.6	35.6	49.2
Drainage area*	(m^2)	125.0	38.1	29.3	14.0	23.6
Max cutslope height	(m)	5.4	2.3	2.2	0.4	1.2
Max cutslope gradient	(%)	102.5	69.2	69.2	31.7	16.8
Travelway-ditch transition	(%)	33.4	17.8	18.2	3.2	7.6
Mean bare soil in ditch†	(%)	63.0	19.7	19.5	3.0	12.1
Mean bare soil on cutslope†	(%)	69.0	20.7	15.5	3.0	16.4
Mean bare soil on travelway†	(%)	63.0	35.5	32.0	11.0	15.2
Surface roughness ratio	-	1.3	1.1	1.1	1.0	0.1

*Area of the road prism (cutslope, road surface, ditch) draining to the sediment trap
†Mean of 18 measurements over one year

Sediment pin heights and area of sediment deposition from each of the 60 sediment traps were measured to the nearest 0.16 cm (1/16th in) with a measuring tape. Eighteen total measurements were taken periodically from June, 2014 to June, 2015. Measurements in February 2015 were delayed due to snow cover. Bulk densities were collected from each catchment area containing sediment deposits following the core method (Blake and Hartge 1986). Sediment volumes were calculated by multiplying depositional area by depositional depth. Bulk densities of trapped sediment were multiplied by sediment volumes to calculate sediment load (Brown et al. 2013). Sediment loads were then divided by the drainage area and elapsed time since the previous measure to obtain erosion rates (Mg ha^{-1} time^{-1} [ton ac^{-1} time^{-1}]).

4.3.5 Statistical Analysis

Within the completely randomized design, each of the five treatments were replicated 11-13 times, totaling 60 experimental units. Trapped sediment erosion rates were not normally distributed (Shapiro-Wilk test) and not varied equally (Levene's test). Data transformations of the sediment trap measures did not result in equal variance. Therefore, nonparametric Kruskal-Wallis methods were used to compare erosion rates among ditch treatments (Zar 2010). Significant differences were based on alpha levels of 0.10 and subsequent median separations of significantly different treatments were based on Steel-Dwass all pairs test (Zar 2010). All statistical analyses were completed in JMP version 11 software package (JMP 2016).

4.4 Results and Discussion

4.4.1 Ditch Treatment Effects

Summary statistics of soil erosion rates for all treatments are provided in Table 4.2. Bare treatments had the largest range of erosion rates (0.54 to 231 Mg ha^{-1} y^{-1} [0.24 to 103 ton ac^{-1} yr^{-1}]). All ditch BMP treatments had some experimental units with trapped sediment less than 0.6 Mg ha^{-1} y^{-1} (0.27 ton ac^{-1} yr^{-1}), indicating that all treatments function well under certain circumstances. Maximum erosion rates for Seed, Mat, Dam, and Rock treatments were less than 35 Mg ha^{-1} y^{-1} (15.6 ton ac^{-1} yr^{-1}), which was 6.6x less than the maximum Bare treatment. The Kruskal-Wallis statistical test indicated that trapped sediment data for ditch BMP treatments were significantly different ($p \leq 0.0393$) (Table 4.2). The Steel-Dwass all-pairs comparison test among ditch BMP treatments indicated that Mat, Seed, and Rock treatments had similar median erosion rates below 1.8 Mg ha^{-1} y^{-1} (0.8 ton ac^{-1} yr^{-1}). Only Mat treatment medians had significantly different erosion rates from Dam and Bare treatments, while Bare, Seed, Dam, and Rock treatment medians showed no significant differences from each other (Table 4.2).

Table 4.2. Summary statistics for ditch BMP treatments based on one year of sediment deposits.

Ditch BMP Treatment	n	Erosion Rate (Mg ha^{-1} y^{-1})						
		Max.	Q75	Median*	Mean**	Q25	Min.	Std. Dev.
Bare	12	231.0	45.5	4.92 b	34.50	0.87	0.54	66.4
Seed	11	26.1	12.8	1.04 ab	7.07	0.33	0.09	8.55
Mat	12	5.7	1.6	0.82 a	1.21	0.14	0.08	1.57
Dam	13	34.9	24.5	6.14 b	11.70	1.36	0.06	12.5
Rock	12	30.2	9.7	1.73 ab	5.68	0.94	0.24	8.69

* $p < 0.10$
** Since nonparametric statistics were used for analyses, mean separation tests were not performed on mean values.

4.4.2 Road Site Differences

Differences in trapped sediment and road attributes by road sites are provided in Table 4.3. Water Tower and Turkey Nest roads had greater trapped sediment amounts than Cassidy's Ridge ($p \leq 0.0067$). Cassidy's Ridge had smaller drainage and cutslope areas and lower mean bare soil on the travelway compared to Water Tower road (Table 4.3). Compared to Turkey Nest road, Cassidy's Ridge had shorter maximum cutslope heights and less bare soil on cutslopes and the travelway (Table 4.3). The Kruskal-Wallis statistical test indicated that trapped sediment data for road sites by Mat treatments were significantly different ($p \leq 0.0260$). The Steel-Dwass all-pairs comparison test among road sites for Mat treatments indicated that Cassidy's Ridge had lower trapped sediment than Turkey Nest road ($p \leq 0.0365$).

Table 4.3. Median trapped sediment values and mean road attributes by road sites.

Site Attributes	Cassidy's Ridge	Water Tower	Turkey Nest	p-value
Trapped sediment* (Mg ha^{-1} y^{-1})	0.5a	11.1b	2.3b	0.0067
Ditch length (m)	16.7	16.2	16.7	0.7895
Ditch slope (%)	10.0b	10.9b	4.8a	0.0001
Drainage area* (m^2)	27.2a	73.2b	28.9a	0.0001
Travelway area (m^2)	45.6	63.2	45.7	0.3898
Cutslope area (m^2)	36.4a	57.0b	40.7a	0.0028
Max cutslope height (m)	1.7a	2.4ab	2.6b	0.0425
Max cutslope gradient (%)	65.1	73.7	64.0	0.1118
Travelway-ditch transition (%)	24.2b	22.6b	12.5a	0.0001
Ditch roughness ratio	1.1	1.2	1.1	0.2279
Mean bare soil (ditch)† (%)	16.1	23.9	19.7	0.2272
Mean bare soil (cutslope)† (%)	12.7a	11.9a	28.5b	0.0003
Mean bare soil (travelway)† (%)	20.2a	34.5b	43.8b	0.0001

*Median values
**Area of the road prism (cutslope, road surface, ditch) draining to the sediment trap
†Mean of 18 measurements over one year

4.4.3 Erosion Control

Total soil erosion values relative to precipitation amounts for all ditch BMP treatments are depicted for each measurement date in Figure 4.4. Erosion summations across all ditch BMP treatments were greatest immediately following ditch reconstruction and BMP implementation. The highest sum of erosion, 360 Mg ha^{-1} (161 ton ac^{-1}), occurred during this first measurement period (Figure 4.4) when construction effects were fresh. Erosion across all treatments summed to 55 Mg ha^{-1} y^{-1} (25 ton ac^{-1} yr^{-1}) for the August measure, although precipitation amounts were similar to the first measurement period. This reemphasizes that erosion had originally been influenced by construction. The higher initial erosion sum was most likely the result of ditch reconstruction, which disturbed the road surface and exposed mineral soil, thereby leaving loose, easily eroded material within ditches. Such materials have been found to move rapidly with surface runoff on newly constructed roads in Idaho (Megahan 1983). Luce and Black (1999) found that grading road surfaces and removing vegetation from the ditch and cutslope in western Oregon increased erosion 6.6x over grading the road surface only. Megahan (1974) found that soil erosion followed an exponential relationship and that the largest percentage of soil loss occurred within one to two years after soil disturbances. Additionally, over time, armoring via natural aggregate and vegetative succession occurs and erodibility of soil tends to decrease. Data collected by Luce and Black (1999) in western Oregon suggested that loose sediments were available in all ditches throughout their study. In our study, we observed loose sediment immediately following ditch reconstruction, but the abundance of loose sediment appeared to dissipate and ditch bottoms apparently became armored with time. These observations are supported by the rapid decline in trapped sediment (Figure 4.4).

Originally, we speculated that the Rock ditch BMP treatments would be the most effective for erosion control, yet the sediment data analyses did not find Rock to be significantly better than Bare. The #1 stones applied to the Rock ditches were unwashed and fine materials attached to the stone may have contributed increased sediment from the Rock treatment. Toman and Skaugset (2011) compared road surfacing treatments and found that surface aggregates with the greatest percentage of fine materials produced the largest amount of sediment. These fine textured attachments are particularly prone to erosion. Luce and Black (1999) found that finer textured soils produced 4-9 times more sediment than coarser textured soils.

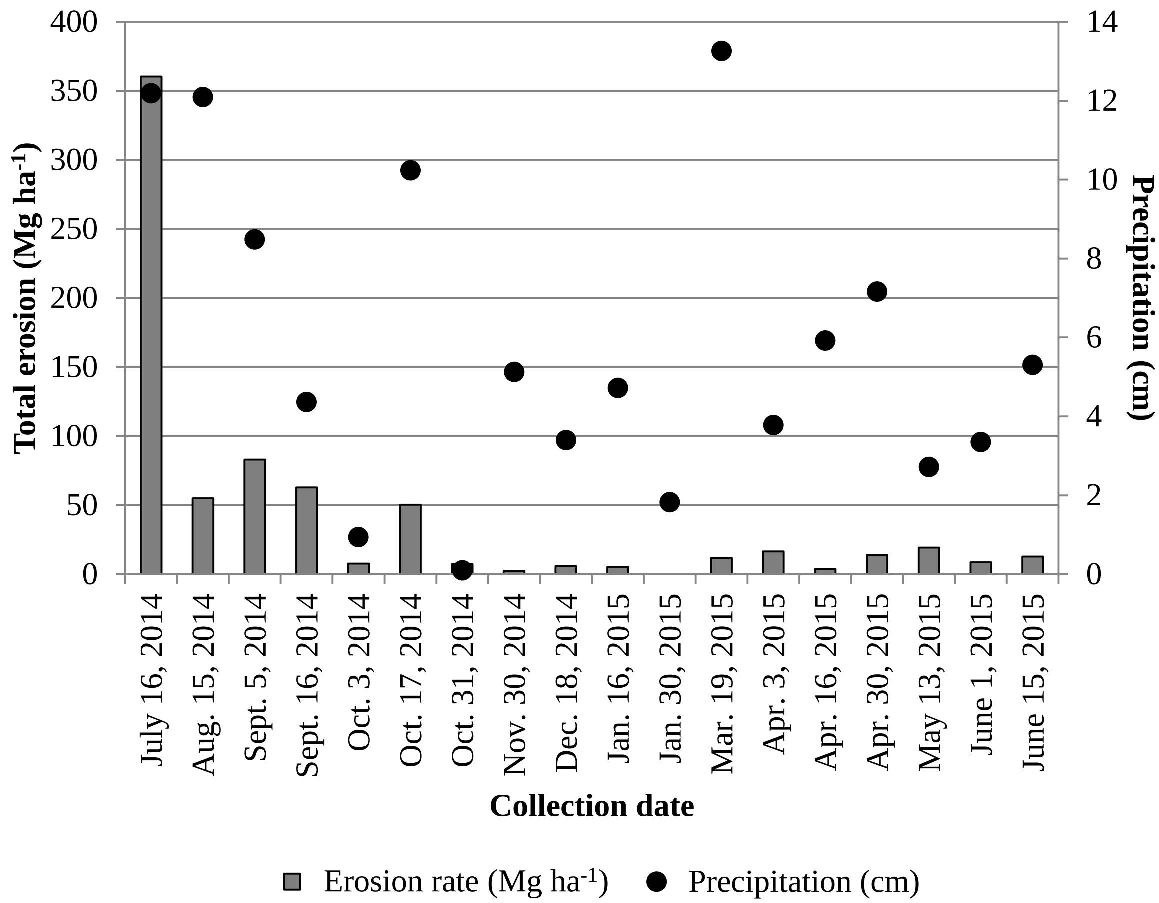

Figure 4.4. Total erosion for all ditch BMP treatments by collection period. The secondary y-axis depicts the precipitation totals for those same periods. Note that time between collection dates were not uniform.

Grading of the road surface is necessary in some instances for the maintenance of the road template and effectiveness of road drainage structures. However, frequent grading of road surfaces and ditches should be avoided unless it is necessary to repair severe drainage or soil erosion issues. Coe (2006) compared recently graded native surface roads with ungraded native surface roads and found that graded roads produced 2x more sediment per unit storm energy than ungraded roads. Splash erosion from simulated rainfalls accounted for 38-48% of the total sediment production on two road segments in Thailand and Hawaii (Ziegler et al. 2000). Thus, in order to maximize erosion control, BMPs should be implemented immediately after construction disturbances. The frequency of road grading can be minimized by adequately designing drainage systems and restricting wet weather traffic (MacDonald and Coe 2008). Provencher (1995) found that by identifying road segments in poor conditions and scheduling grading accordingly, maintenance costs could be reduced by 30% when compared to road grading on fixed time intervals. Coulter et al. (2006) combined road design and maintenance schedule considerations into a model (Analytic Hierarchy Process) to minimize stream impacts, optimize road design and location, and minimize maintenance costs. Other models have also been developed to help forest engineers locate and design forest roads, which may reduce the frequency of maintenance operations (e.g. Akay and Sessions 2005; US Forest Service 2009; Conrad et al. 2012).

Figure 4.5 provides the median values of erosion from the five ditch BMP treatments. Median values were assessed because the data was not normally distributed nor varied equally. Median erosion values were greatest for each treatment during the first measurement. Immediately following construction, Bare treatments had the highest erosion rates and accounted for 61% of the total erosion from all ditch sections. The top four most erosive Bare ditch BMP treatments immediately following construction as measured initially accounted for nearly 58% of the total erosion from all 60 road segments. The Dam treatments did little to provide soil cover, therefore the Bare and Dam treatments had extents of exposed bare soil. Additionally, Dam and Rock treatment installation may have caused some degree of ditch bank disturbance. We observed three

Dam and one Bare treatment sections in August, September, and October with small cutslope slumps, which probably influenced sediment delivery to traps.

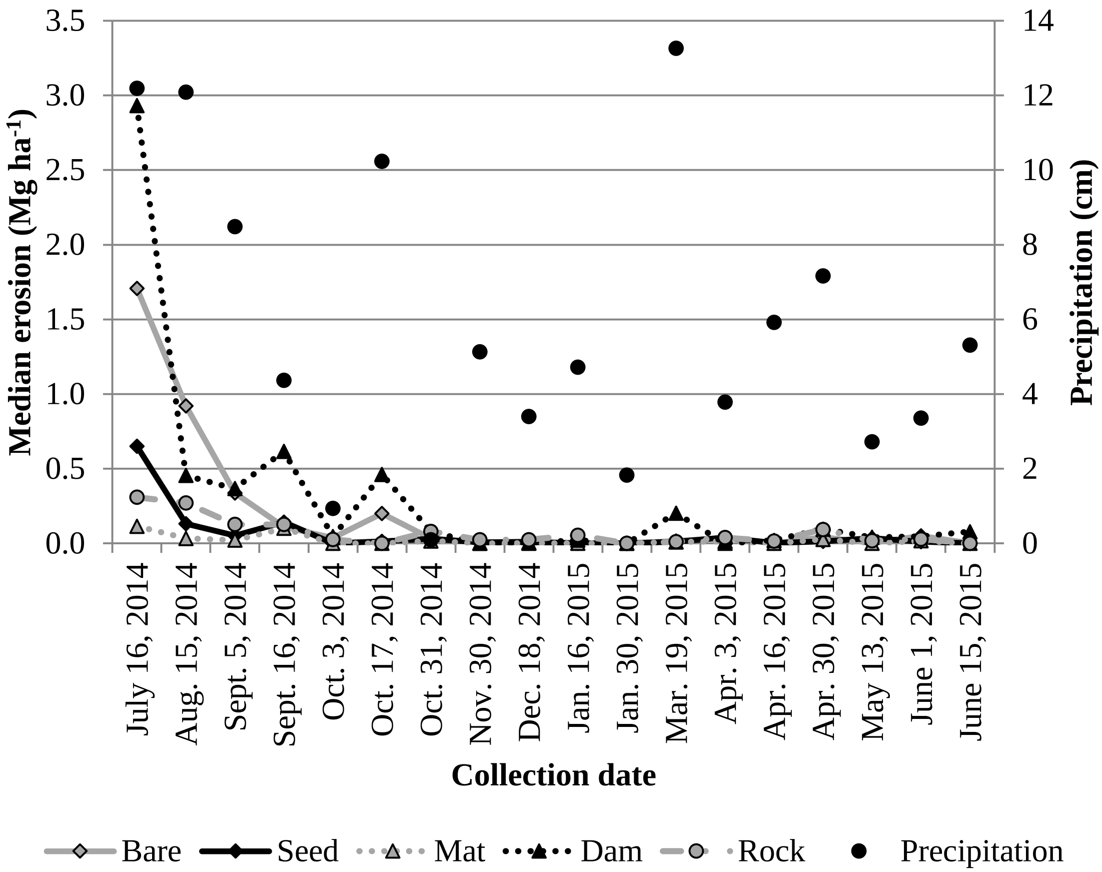

Figure 4.5. Median erosion for each ditch BMP treatment by collection period. The secondary y-axis depicts the precipitation totals for those same periods. Note that time between collection dates were not uniform.

Over the entire year, three of the Bare treatment segments had the greatest erosion rates (230, 77.6 and 52.6 Mg ha^{-1} y^{-1} [313, 103, 23.5 ton ac^{-1} yr^{-1}]). These three sites produced nearly 50% of the total sediment eroded from all 60 sites. Notable site characteristics on these three sites may explain the accelerated erosion rates. The ditch section with the greatest erosion rate intercepted ground water along a cutslope, which was collapsing and providing additional sediment to the ditch. Road cutslopes can intercept subsurface flow when the water table rises to cutslopes or if cutslopes are excavated at the depth of the permeant water table (Dutton 2000; Wemple and Jones 2003). Saturated soils on the cutslopes may be unstable and may fail by undercutting or slumping caused by ditch maintenance or erosion. The locations where subsurface flow will be intercepted by cutslopes are difficult to predict during road construction planning (Toman 2004). Megahan et al. (2001) provided a rough approximation of cutslope erosion following construction and concluded that cut bank erosion may last five times longer than erosion from other road prism components. In the current study, our data suggests the ditch BMPs were effective after seed germination and/or following the initial wash of disturbed soil. The second greatest erosion rate was associated with an excessively large drainage area (214 m^2 [2304 ft^2]) caused by a rutted travelway, lack of water control, and a steep slope (10.1%). Bedrock at this location was near the surface, which likely discouraged implementation of adequate cross drains during construction. The third highest erosion rate had a steep ditch bank slopes (33.4%) and steep slope (12.0%) along the ditch channel. Overall, ditch BMP treatment erosion rates generally followed precipitation patterns in September and October. By the end of October, leaf senescence occurred and litterfall covered most of the ditch sections. Subsequently, small variations in erosion were observed for the remainder of the study (Figure 4.5).

Soil erosion occurs where soil is exposed to kinetic energy generated by rainfall, wind, or other forces. Numerous studies have elucidated the value of providing soil coverage in order to control erosion from forest road surfaces. Brown et al. (2013) compared five bare surface roads with four gravel surface roads and found that the mean of bare roads produced 7.5x more sediment that of graveled. Brown et al. (2014) evaluated sediment delivery from six stream crossing approaches in the Virginia Piedmont and concluded that low gravel coverage (34-60%) reduced sediment delivery by a factor of two compared to bare road surface. Similarly, skid trail closure has satisfactorily achieved acceptable erosion rates when vegetative or slash cover are used (Sawyers et al. 2012; Wade et al. 2012; Wear et al. 2013). Our results indicate that ditch erosion control may require similar cover related BMPs that provide protection from erosive forces.

Figure 4.6 separates bare soil percentages of the various ditch BMP treatments for three sediment contribution areas (road surface, cutslope and ditch). Bare, Seed, and Dam treatments provided the least amount of soil cover within ditches following reconstruction (< 50%), which provided more erosion prone area than Rock and Mat treatments (Figure 4.6). Percent cover within the ditches increased about 20% for the Seed treatments following grass seed germination. Following reconstruction, exposure of bedrock or loose rock materials inhibited seed growth on several Seed treatments. Germination success was highly variable. Exposure of rock fragments in the ditch increased after the initial wash of material on some treatments. However, some sites remained highly suitable for seed germination (dependent on the presence of soil). Rock exposure may increase surface runoff by reducing hydraulic conductivity, but may also reduce erosion by providing additional soil cover (Jordan and Martinez-Zavala 2008; Vinson 2016). Arnáez et al. (2004) found that gradients, plant cover density, and rock fragment cover on cutslopes, fillslopes, and roadbeds had significant effects on runoff and erosion during a rainfall simulation experiment in Spain. They found that increased rock fragment content was positively correlated with soil loss on fillslopes. Conversely, Cerdá (2001) showed that rock fragments enhanced infiltration and reduced erosion during rainfall simulation experiments in southeast Spain. Jordan and Martinez-Zavala (2008) also found that increased rock fragment contents were negatively correlated with runoff rate on cutslopes. An overall evaluation of the bare soil from three contributing areas indicated that the bare treatment had the least cover during the entire time and the Mat treatment had the best cover.

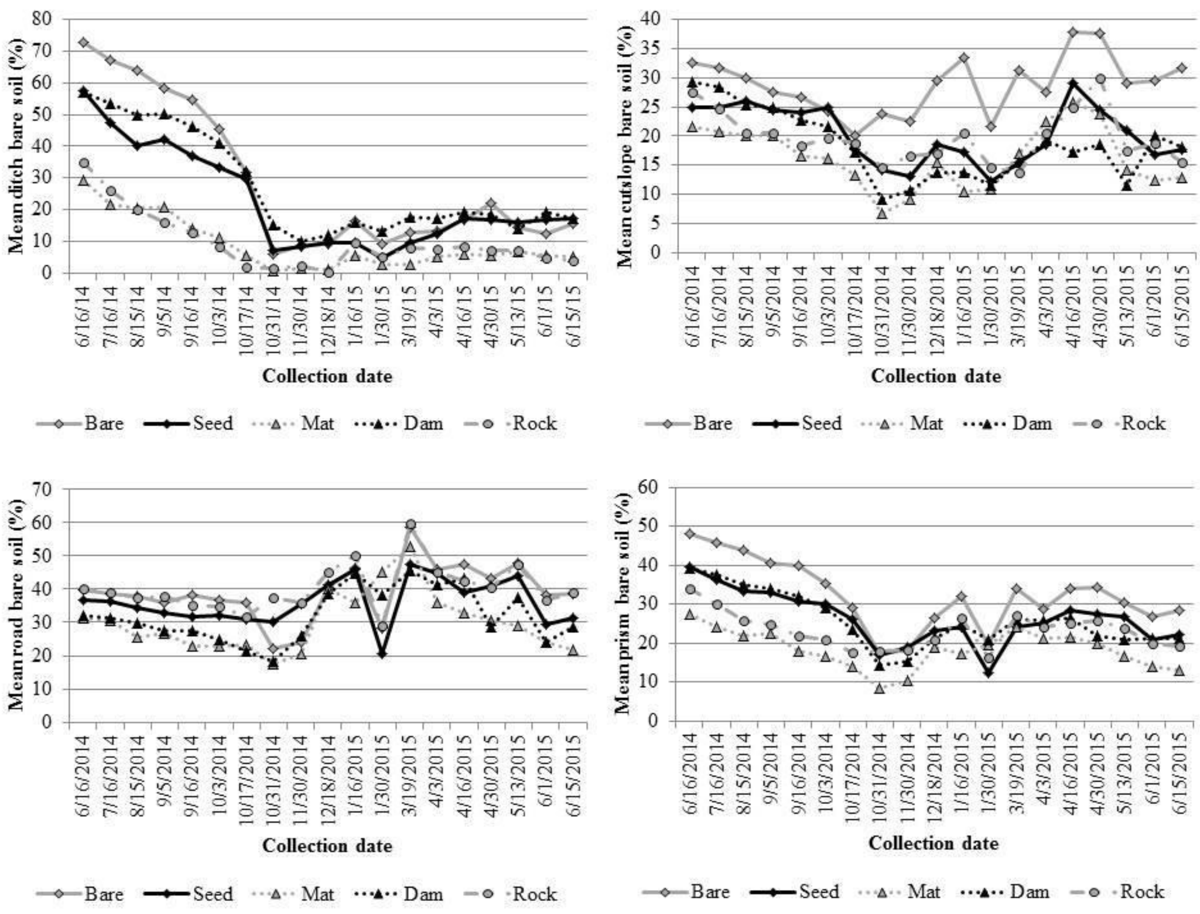

Figure 4.6. Percent mean bare soil for ditch (top left), cutslope (top right), road surface (bottom left), and overall mean of the three road components (bottom right) for each BMP treatment by collection period.

Grass can be an effective erosion control measure, but depends on seed germination. The success of grass was excellent on sites with appropriate soils and sparse on sites with rocky soils. Reseeding alone will have minimal effect because of the slow initial growth responses. Additionally soil protection is necessary during this period of high erosion potential. Researchers have shown that sediment production from ditches, cutslopes, and fill slopes can be reduced by mulch, grass seed, and natural vegetation (Carr and Ballard, 1980; Megahan et al., 2001; Swift 1984).

Litterfall on the road surface prevents soil erosion in a variety of ways. The litterfall covers bare soil and prevents soil detachment from rainfall. Litter increases may hold water on the road surface for longer periods, which may be slowly evaporated and reduce the amount of runoff generated. Litterfall increase soil roughness, which decreases water velocity, and may increase infiltration rates. Brown et al. (2013) found that sediment delivery rates were reduced on bare road plots when leaf litter covered a substantial portion of the road prism. Litter may also increase erosion rates by retaining moisture, especially if the road is trafficked. Retaining the soil moisture subjects the road surface to rutting, which is why daylighting (removing tree canopy shading the road surface) is a common recommendation for actively trafficked roads. Erosion rates on Bare ditches may not be as high of a concern in areas where litterfall can cover soil. In this study, Bare ditches may have been adequate because litterfall covered much of the ditch and road surface and low traffic volumes.

Figure 4.7 depicts mean erosion rates for each measurement period and ditch BMP by mean percent bare soil. Mean percent bare soil values were significant predictors of mean ditch erosion rates ($p < 0.0001$) and explained 58% of the total variation in erosion rates. The polynomial regression fit for all data (n = 90) indicates that mean erosion rates for each measurement period and ditch BMP (Mg ha^{-1} y^{-1}) = -0.0006 * Bare soil + 0.0033 * (Bare soil - 19.9889)^2.

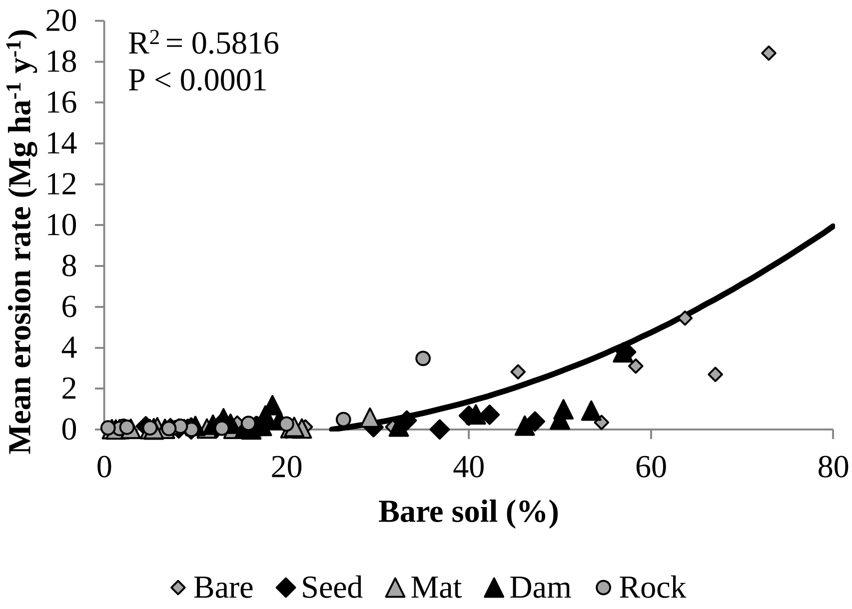

Figure 4.7. Percent mean bare soil for each measurement period and ditch BMP treatment (n = 18 for each BMP treatment [n = 90 total]) as a predictor of mean erosion rate in Mg ha^{-1} y^{-1}. Polynomial regression line: Predicted mean erosion rate = -0.0562 * Bare soil + 0.0023 * (Bare soil)^2.

4.4.4 Ditch Best Management Practices Treatment Costs

We used the Bare ditch treatment as the control for cost comparisons and machine, labor, and material costs for all ditch BMP treatments per stream crossing approach (15.2 m [50 ft] length) are provided in Table 4.4. Machine and labor costs were constant across the road and ditch reconstruction (US$674 approach^{-1}). Our costs match relatively well with similar published stream crossing and BMP improvement costs.

McKee et al. (2012) surveyed loggers regarding stream crossings and found that loggers estimated stream crossing BMP costs to be approximately US$445 approach^{-1}. Nolan et al. (2015) evaluated 20 haul road stream crossings in the Virginia Piedmont and estimated BMP costs that would improve minimal stream crossing BMPs to adequate or above adequate levels. Nolan et al. (2015) found that stream crossing BMP improvements, including water control improvements, seed, mats, and gravel would have an estimated cost between $450 and $480 approach^{-1}. Wear et al. (2013) obtained contractor BMP costs for closure of skidder bridge stream crossings in the Virginia Piedmont and reported costs ranging from US$120 to US$345 approach^{-1}. These cost data are useful when selecting BMP treatments. In our study, the Mat ditch BMP treatment was very effective and the cost data support use of this treatment which only costs US$21.33 more than leaving a bare ditch. However, the cost data also indicate that the Rock treatment should be used more judiciously because it increased the BMP cost per stream crossing approach by 20.9% ($141.08). Ice (2011) assessed methods to evaluate BMP effectiveness, including percent reduction in pollutant load, attainment of water quality or habitat standards, biological response, and economic efficiency. He concluded that a weight-of-evidence method that incorporates multiple dimensions of effectiveness could provide useful information about management decisions. Additionally, BMP implementation levels are improved when economic feasibility is included in the BMP recommendation.

Table 4.4. Cost estimates for ditch reconstruction and BMP techniques used for haul road stream crossing approaches in the Ridge and Valley during 2014.

Treatment	Machine and labor costs for ditch and crown reconstruction	Material costs for ditch treatments	Machine and labor costs for ditch treatments	Total ditch erosion control costs	Total costs for road maintenance and erosion control treatments	Percentage increase above Bare ditch BMP treatment
	US$ per 15.2 m stream crossing approach					%
Bare	$674.00	$0.00	$0.00	$0.00	$674.00	-
Seed	$674.00	$3.23	$2.87	$6.10	$680.01	0.9
Mat	$674.00	$12.73	$8.60	$21.33	$695.33	3.2
Dam	$674.00	$30.33	$41.10	$71.43	$745.43	10.6
Rock	$674.00	$63.19	$77.89	$141.08	$815.08	20.9

4.5 Summary and Conclusions

Our data support the premise that ditched roads leading to stream crossings have the potential to deliver large quantities of sediment to streams if appropriate BMPs are not utilized. BMPs can be effectively employed to reduce and control erosion within haul road ditches near stream crossings. BMPs can address specific circumstances such as large bare soil contributing areas with steep slopes, bare cutslopes with active seeps, and rutted road surfaces that allow runoff to become concentrated. Findings from this study suggest Mat, Seed, and Rock ditch BMP treatments were highly effective at reducing erosion, but the Mat was most effective since it provided immediate soil protection. However, Seed and Rock BMP treatments reduced bare soil in ditches and provided long-term erosion control. Any BMPs that reduced bare soil provided reductions in erosion and even natural site conditions, including litterfall and invasive vegetation were important to erosion control. Percent bare soil values were significant predictors of ditch erosion rates and explained 58% of the total variation in erosion rates. When planning which erosion control techniques to implement, it is important that managers remember that a variety of BMPs can be used to achieve similar levels of water quality protection. Furthermore, this provides managers with the flexibility to select ditch erosion control measures based on their cost while providing flexibility to site-specific circumstances. We found that the Mat treatment enhanced seed germination for a relatively small cost compared to seed alone. When BMP costs are included in the evaluation, Seed and Mat treatments clearly have advantages over the more expensive Dam and Rock ditch BMP treatments.

During the one-year investigation, the Mat treatments were slightly better for erosion control for the majority of our sites. However, it should be remembered that all of the erosion control treatments might be appropriate for a particular situation. For example, for one of the ditch segments, the ditch channel eroded to bedrock and neither the Seed nor Mat ditch treatments would have provided effective erosion control for the ditch sides. For this situation, the Rock or Dam treatments may have been more appropriate ditch BMPs.

Implementing BMPs for roadside ditches cannot overcome the negative consequences of inadequate or improperly implemented water control structures on the road surface. A road ditch segment that handles an oversupply of water, is too steep, or has other unfavorable conditions that generate erosion cannot be totally ameliorated by the use of ditch BMPs alone. In contrast, ditch segments having favorable characteristics, such as low grades and suitable water runoff loads may not require ditch BMPs beyond Bare treatments. In this study, litterfall alone provided adequate erosion control for some bare ditch segments with suitable construction design that were near published geologic erosion rates.

In conclusion, the Seed, Mat, and Rock ditch BMP treatments resulted in low ditch erosion rates. Litterfall reduce erosion rates for all ditch BMP treatments, including Bare. In some circumstances, such as ditches within a clearcut area, such litterfall would not be available to control ditch erosion. As with other BMP selection and implementation, the most effective ditch treatments BMPs will be those selected by a trained professional having adequate site knowledge.

4.6 Acknowledgements

Funding for this work was provided in part by the National Council for Air and Stream Improvement (NCASI), Inc., the McIntire-Stennis Program of the National Institute of Food and Agriculture, and the Virginia Tech Forest Operations and Business Research Cooperative. Virginia Tech Department of Forest Resources and Environmental Conservation, provided logistical support and study sites. The United States Forest Service provided study sites on the Washington/Jefferson National Forest.

4.7 References

Akay, A.E., and J. Sessions. 2005. Applying the decision support system, TRACER, to forest road design. Western Journal of Applied Forestry 20(3):184-191.

Anderson, C.J., and B.G. Lockaby. 2011a. The effectiveness of forestry best management practices for sediment control in the southeastern United States: A literature review.Southern Journal of Applied Forestry 35(4):170-177.

Anderson, C.J., and B.G. Lockaby. 2011b. Research gaps related to forest management and stream sediment in the United States. Environmental Management 47(2):303-313.

Arnáez, J., V. Larrea, L. Ortigosa. 2004. Surface runoff and soil erosion on unpaved forest roads from rainfall simulation tests in Northeastern Spain. Catena. 57:1-14.

Aust, W.M., and C.R. Blinn. 2004. Forestry best management practices for timber harvesting and site preparation in the eastern United States: An overview of water quality and productivity research during the past 20 years (1982–2002). Water, Air, and Soil Pollution: Focus 4(1):5-36.

Aust, W.M., M.B. Carroll, M.C. Bolding, and C.A. Dolloff. 2011. Operational forest stream crossings effects on water quality in the Virginia Piedmont. Southern Journal of Applied Forestry 35(3):123-130.

Bailey, R.W. 1948. Reducing runoff and siltation through forest and range management. Journal of Soil and Water Conservation 3:24-31.

Barrett, S.M., W.M. Aust, M.C. Bolding, W.A. Lakel, and J.F. Munsell. 2016. Estimated erosion, ground cover, and best management practices audit details for postharvest evaluations of biomass and conventional clearcut harvests. Journal of Forestry 114(1):9-16.

Beschta, R.L. 1978. Long-term patterns of sediment production following road construction and logging in the Oregon coast range. Water Resources Research 14:1011-1016.

Bilby, R.E., K. Sullivan, and S.H. Duncan. 1989. The generation and fate of road-surface sediment in forested watersheds in southwestern Washington. Forest Science 35(2):453-468.

Binkley, D., and T.C. Brown. 1993. Forest practices as nonpoint sources of pollution in North America. Water Resources Bulletin 29(5):729-740.

Blake, G.R., and K.H. Hartge. 1986. Bulk density. *In* Methods of soil analysis, Part 1, physical and mineralogical methods, 2nd ed., ed. A. Klute. Madison, WI: Soil Science Society of America, Inc.

Brown, K.R., K.J. McGuire, W.M. Aust, W.C. Hession, and C.A. Dolloff. 2014. The effect of increasing gravel cover on forest roads for reduced sediment delivery to stream crossings. Hydrological Processes 29(6):1129-1140.

Brown, K.R., W.M. Aust, and K.J. McGuire. 2013. Sediment delivery from bare and graveled forest road stream crossing approaches in the Virginia Piedmont. Forest Ecology and Management 310:836-846.

Bureau of Labor Statistics (BLS). 2014. Occupational Employment Statistics. 47-2061 Construction Laborers. Mean hourly wage. http://www.bls.gov/oes/current/oes472061.htm.

Carr, W.W., and T.M. Ballard. 1980. Hydroseeding forest roadsides in British Columbia for erosion control. Journal of Soil and Water Conservation. 35(1):33-35.

Cerdá, A. 2001. Effects of rock fragment cover on soil infiltration, interrill runoff and erosion. European Journal of Soil Science 52:59-68.

Clinton, B.D., and J.M. Vose. 2003. Differences in surface water quality draining four road surface types in the southern Appalachians. Southern Journal of Applied Forestry 27(2):100-106.

Coe, D.B.R. 2006. Sediment Production and Delivery from Forest Roads in the Sierra Nevada, California. Master's thesis, Colorado State University.

Conrad, J.L., W.S. Ford, M.C. Groover, M.C. Bolding, and W.M. Aust. 2012. Virginia Tech forest road and bladed skid trail cost estimation method. Southern Journal of Applied Forestry 36(1):26-32.

Coulter, E.D., J. Sessions, and M.G. Wing. 2006. Scheduling forest road maintenance using the analytic hierarchy process and heuristics. Silva Fennica 40(1):43-160.

Coulter, E.D., J. Sessions, and M.G. Wing. 2006. Scheduling forest road maintenance using the analytic hierarchy process and heuristics. Silva Fennica 40(1):143-160.

Cristan R., W.M. Aust, M.C. Bolding, S.M. Barrett, and J.F. Munsell. forthcoming. National status of state developed and implemented forestry best management practices in the United States.

Cristan, R., W.M. Aust, M.C. Bolding, S.M. Barrett, J.F. Munsell, and E.B. Schilling. 2016a. Effectiveness of forestry best management practices in the United States: Literature review. Forest Ecology and Management 360:133-151.

Croke, J., S. Mockler, P. Fogarty, and I. Takken. 2005. Sediment concentration changes in runoff pathways from a forest road network and the resultant spatial pattern of catchment connectivity. Geomorphology 68(3):257-268.

Croke, J.C., and P.B. Hairsine. 2006. Sediment delivery in managed forests: A review. Environmental Reviews 14(1):59-87.

Dutton, A.L. 2000. Process-based Simulations of Near-surface Hydrologic Response for a Forested Upland Catchment: The Impact of a Road. Master's thesis, Stanford University.

Dymond, S.F., W.M. Aust, S.P. Prisley, M.H. Eisenbies, and J.M. Vose. 2014. Application of a distributed process-based hydrologic model to estimate the effects of forest road density on stormflows in the southern Appalachians. Forest Science 60(6):1213-1223.

Edwards, P.J., and K.W. Williard. 2010. Efficiencies of forestry best management practices for reducing sediment and nutrient losses in the eastern United States. Journal of Forestry 108(5):245-249.

Edwards, P.J., J.E. Schoonover, and K.W.J.Willard. 2015. Guiding principles for management of forested. agricultural, and urban watersheds. Journal of Contemporary Water Research and Education 154(4):60-84.

Foltz, R.B. 2012. A comparison of three erosion control mulches on decommissioned forest road corridors in the northern Rocky Mountains, United States. Journal of Soil and Water Conservation 67(6):536-544.

Forsyth, A.R., K.A. Bubb, and M.E. Cox. 2006. Runoff, sediment loss and water quality from forest roads in a southeast Queensland coastal plain Pinus plantation. Forest Ecology and Management 221(1):194-206.

Gee, G.W., and D. Or. 2002. Particle-size analysis. *In* Methods of soil analysis, Part 4, physical and mineralogical methods, 2nd ed., ed. A. Klute. Madison, WI: Soil Science Society of America, Inc.

Grace, J.M. III. 2002. Control of sediment export from the forest road prism. American Society of Agricultural Engineers 45(4):1127-1132.

Grace, J.M. III., B. Rummer, B.J. Stokes, and J. Wilhoit. 1998. Evaluation of erosion control techniques on forest roads. Transactions of the American Society of Agricultural Engineers 41(2):383-391.

Ice, G.G. 2004. History of innovative best management practice development and its role in addressing water quality limited waterbodies. Journal of Environmental Engineering 130(6):684-689.

Ice, G.G. 2011. Technical note: Assessing best management practices effectiveness at the watershed scale. Applied Engineering in Agriculture 27(6):925-931.

Ice, G.G., E.B. Schilling, and J. Vowell. 2010. Trends for forestry best management practices implementation. Journal of Forestry 108(6):267-273.

Jackson, C.R., J.K. Martin, D.S. Leigh, and L.T. West. 2005. A southeastern piedmont watershed sediment budget: Evidence for a multi-millennial agricultural legacy. Journal of Soil and Water Conservation 60(6):298-310.

JMP, Version 11. 2016. SAS Institute Inc., Cary, NC. 1989-2016.

Jones, J.I., J.F. Murphy, A.L. Collins, D.A. Sear, P.S. Naden, and P.D. Armitage. 2011. The impact of fine sediment on Macro-Invertebrates. River Research and Applications 28(8):1055-1071.

Jordan, A., and L. Martinez-Zavala. 2008. Soil loss and runoff rates on unpaved forest roads in southern Spain after simulated rainfall. Forest Ecology and Management. 255:913-919.

Kraebel, C.J. 1936. Erosion control on mountain roads. US Department of Agriculture Circ. 380. [Old series]. 44 pp.

Lakel, W.A., W.M. Aust, M.C. Bolding, C.A. Dolloff, P. Keyser, and R. Feldt. 2010. Sediment trapping by streamside management zones of various widths after forest harvest and site preparation. Forest Science 56(6):541-551.

Lane, P.N.J., and G.J. Sheridan. 2002. Impact of an unsealed forest road stream crossing: water quality and sediment sources. Hydrological Processes 16(13):2599-2612.

Lang, A.J., M.C. Bolding, and W.M. Aust. 2015b. Sediment reduction and cost of operational ditch best management practices. 38th Council on Forest Engineering Annual Meeting, Lexington, KY.

Lang, A.J., W.M. Aust, M.C. Bolding, and E.B. Schilling. In review. Road attributes at forest haul road stream crossing approaches influence potential sedimentation. Forest Science.

Lang, A.J., W.M. Aust, M.C. Bolding, S.M. Barrett, K.J. McGuire, and W.A. Lakel. 2015a. Streamside management zones compromised by stream crossings, legacy gullies, and over-harvest in the piedmont. Journal of the American Water Resources Association 51(4):1153-1164.

Litschert, S.E., and L.H. MacDonald. 2009. Frequency and characteristics of sediment delivery pathways from forest harvest units to streams. Forest Ecology and Management 259(2):143-150.

Luce, C.H., and B.C. Wemple. 2001. Introduction to special issue on hydrologic and geomorphic effects of forest roads. Earth Surface Processes and Landforms 26:111-113.

Luce, C.H., and T.A. Black. 1999. Sediment production from forest roads in western Oregon. Water Resources Research 35(8):2561-2570.

Martin, D. 2009. Forest road runoff disconnection survey for private timberlands in Washington. Olympia, WA: Washington Forest Protection Association.

McBroom, M.W., R.S. Beasley, M. Chang, and G.G. Ice. 2008. Storm runoff and sediment losses from forest clearcutting and stand re-establishment with best management practices in East Texas, USA. Hydrological Processes 22:1509-1522.

McKee, S.E., L.A. Shenk, M.C. Bolding, and W.M. Aust. 2012. Stream crossing methods, cost, and best management practices for Virginia loggers. Southern Journal of Applied Forestry 36(1):33-37.

Megahan, W.F. 1974. Erosion over time on severely disturbed granitic soils: A model. Research Paper INT-156. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 15 p.

Megahan, W.F. 1983. Hydrologic effects of clearcutting and wildfire on steep granitic slopes in Idaho. Water Resources Research 19(3):811-819.

Megahan, W.F., M. Wilson, and S.B. Monsen. 2001. Sediment production from granitic cutslopes on forest roads in Idaho, USA. Earth Surface Processes and Landforms 26(2):153-163.

Moll, J., R. Copstead, and D.K. Johansen. 1997. Traveled way surface shape. San DimasTechnology and Development Center, United States Forest Service. Record 9777-1808. 10 Pp.

Morris, B.C. 2014. Effectiveness, Cost, and Implications of Forest Haul Road Stream Crossing Structures and Best Management Practices. PhD dissertation, Virginia Tech.

Motha, J.A., P.J. Wallbrink, P.B. Hairsine, and R.B. Grayson. 2003. Determining the sources of suspended sediment in a forested catchment in southeastern Australia. Water Resources Research 39(3):1056.

National Oceanic and Atmospheric Administration (NOAA). 2015. 1981-2010 U.S. Climate Normals. Retrieved from https://www.ncdc.noaa.gov/data-access/land-based-station-data/land-based-datasets/climate-normals/1981-2010-normals-data.

Neary, D.G., W.T. Swank, and H. Riekerk. 1988. An overview of nonpoint source pollution in the southern United States. Pp. 1-7. *In:* D.D. Hook and R. Lea (eds.) Proceedings of a symposium: The forested wetlands of the southern United States. Orlando, FL, July 12-15. U.S. Department of Agriculture Forest Service General Technical Report No. SE-50. 168 pp.

Nolan, L., W.M. Aust, S.M. Barrett, M.C. Bolding, K.R. Brown, and K.J. McGuire. 2015. Estimating costs and effectiveness of upgrades in forestry best management practices for stream crossings. Water 7(12):6946-6966.

North Carolina Forest Service (NC FS). 2014. A Guide for Forest Access Road Construction and Maintenance in the Southern Appalachian Mountains. Publication number WQ-02-14.

Patric, J.H. 1976. Soil erosion in the eastern forest. Journal of Forestry 74(10):671-677.

Provencher, Y. 1995. Optimizing road maintenance intervals. *In* Proceedings of the Sixth International Conference on Low-Volume Roads, Volume I, Minneapolis, Minnesota, June 25-29, 1995.

Reid, L.M., and T. Dunne. 1984. Sediment production from forest road surfaces. Water Resources Research 20(11):1753-1761.

Robichaud, P.R., and P.R. Brown. 2002. Silt fences: An economical technique for measuring hillslope soil erosion. General Technical Publication *RMRS-GTR-94*. Washington, D.C.: USDA Forest Service.

Saleh, A. 1993. Soil roughness measurement: Chain method. Journal of Soil and Water Conservation 48(6):527-529.

Sawyers, B.C., M.C. Bolding, W.M. Aust, and W.A. Lakel. 2012. Effectiveness and implementation costs of overland skid trail closure techniques in the Virginia Piedmont. Journal of Soil and Water Conservation 67(4):300-310.

Shepard, J.P. 2006. Water quality protection in bioenergy production: the US system of forestry best management practices. Biomass and Bioenergy 30(4):378-384.

Sheridan, G.J., and P.J. Noske. 2007. Catchment-scale contribution of forest roads to stream exports of sediment, phosphorus and nitrogen. Hydrological Processes 21(23):3107-3122.

Swank, W.T., J.M. Vose, and K.J. Elliott. 2001. Long-term hydrologic and water quality responses following commercial clearcutting of mixed hardwoods on a southern Appalachian catchment. Forest Ecology and Management 143(1):163-178.

Swift, L.W., Jr. 1984. Soil losses from roadbeds and cut and fill slopes in the southern Appalachian mountains. Southern Journal of Applied Forestry. 8(4):209-216.

Swift, L.W., Jr. 1985. Forest road design to minimize erosion in the southern Appalachians. *In* Proceedings of Forestry and Water Quality: A Mid-South Symposium. Monticello, AR, 141-151.

Toman, E.M. 2004. Forest Road Hydrology: The Influence of Forest Roads on Stream Flow at Stream Crossings. Master's thesis, Oregon State University.

Toman, E.M., and A.E. Skaugset. 2011. Reducing sediment production from forest roads during wet-weather hauling. Transportation Research Record: Journal of the Transportation Research Board 2203: 13-19.

Trimble, G.R. and R.S. Sartz. 1957. How far from a stream should a logging road be located? Journal of Forestry 55(5):339-341.

United States Department of Agriculture, National Resources Conservation Service (USDA NRCS). 2016. Soil Survey. Retrieved from Soil Survey Staff http://websoilsurvey.nrcs.usda.gov/.

United States Forest Service (USFS). 2009. Cost estimating guide for road construction. Intermountain, Southwestern, and Rocky Mountain Regions. 155 p. Available online at www.fs.fed.us/r1/projects/costguide_index.shtml.

Vinson, J.A. 2016. Soil erosion and modeling following closure best management practices for bladed skid trails in the Ridge and Valley region. Master's thesis. Virginia Tech.

Virginia Department of Forestry (VDOF). 2011. Virgina's Forestry Best Management Practices for Water Quality: Technical Manual. Fifth ed. Charlottesville, Va.: Virginia Department of Forestry.

Wade, C., M.C. Bolding, W.M. Aust, W.A. Lakel, and E.B. Schilling. 2012. Comparing sediment trap data with the USLE-Forest, RUSLE2, and WEPP-Road erosion models for evaluation of bladed skid trail BMPs. Transactions of the American Society of Agricultural and Biological Engineers 55(2):403-414.

Walbridge, T.A. 1997. The Location of Forest Roads. Blacksburg, Virginia: Walbridge and Associates. p. 91.

Wear, L.R., W.M. Aust, M.C. Bolding, B.D. Strahm, and C.A. Dolloff. 2013. Effectiveness of best management practices for sediment reduction at operational forest stream crossings. Forest Ecology and Management 289:551-561.

Wemple, B.C., and J.A. Jones. 2003. Runoff production on forest roads in a steep, mountain catchment. Water Resources Research 39(8):1-17.

Wemple, B.C., J.A. Jones, and G.E. Grant. 1996. Channel network extension by logging roads in two basins, Western Cascades, Oregon. Water Resources Bulletin 32(6):1195-1207.

Wood, P.J., and P.D. Armitage. 1997. Biological effects of fine sediment in the lotic environment. Environmental Management 21(2):203-217.

Yoho, N.S. 1980. Forest management and sediment production in the south: A review. Southern Journal of Applied Forestry 4(1):27-36.

Zar, J.H. 2010. Biostatistical Analysis. Upper Saddle River, NJ: Pearson-Prentice Hall. pp. 960.

Ziegler, A.D., R.A. Sutherland, and T.W. Giambelluca. 2000. Partitioning total erosion on unpaved roads into splash and hydraulic components: The roles of interstorm surface preparation and dynamic erodibility. Water Resources Research 36(9)2787-2791.

Ziegler, A.D., R.A. Sutherland, and T.W. Giambelluca. 2001. Special issue interstorm surface preparation and sediment detachment by vehicle traffic on unpaved mountain roads. Earth Surface Processes and Landforms 26:235-250.

5.0 Conclusions

The overall purposes of this research were to quantify and compare sediment delivery and forestry BMP effectiveness from haul road stream crossing approaches and ditches. The three studies in this dissertation provide a quantitative assessment of sediment production and sediment delivery from forest haul roads in the Virginia Piedmont and Ridge and Valley regions. Water quality and economic efficiency are foci during pre-harvest planning and operations and state agencies continue to improve and modify forestry BMP recommendations for forest operations based on applicable research findings. Additionally, state agencies are interested in the use of soil erosion models for estimations and rankings of water quality impacts following forest operations. Erosion models may be useful tools in ranking erosion prone areas, which can help direct expenditures from limited budgets, and such estimates are needed to evaluate erosion control techniques across broad spatial scales. However, models need to be tested in representative situations for which they might be applied. Therefore, sediment production rates were measured and modeled to evaluate and compare road and ditch segments near stream crossings with various ranges of road attributes, BMPs, and management objectives.

Sediment mass delivered from the 37 haul road stream crossing approaches ranged from <0.1 to 2719.1 kg for the one year study. Piedmont sites delivered significantly more sediment than sites in the Ridge and Valley region ($P \leq 0.0252$). The differences between the two regions were likely functions of management objectives. Piedmont sites were managed intensively for short-rotation pine, while Ridge and Valley sites were managed extensively for long-rotation hardwoods. Consequently, road width, bare soil level, and distance to the nearest water control structure were significantly different between regions ($P \leq 0.1000$). Collectively, five approaches accounted for 82% of the total sediment mass trapped. Most sediment delivery rates greater than 10 Mg ha^{-1} yr^{-1} were associated with Low quality roads (i.e., inadequately spaced, constructed, or maintained water control structures, inadequately stabilized surfaces, or unrestricted road access) with unfavorable road characteristics (i.e., steep and/or excessive road prism dimensions and poor road location and drainage). Low quality roads produced more sediment than Standard or High quality roads ($P \leq 0.0017$). Nearly 90% (26 of 30) of Standard or High road quality ranked approaches generated less than 0.10 Mg of sediment or less than 10 Mg ha^{-1} yr^{-1}. Among approaches with less than 0.1 Mg of trapped sediment, road gradients ranged from 1% to 13%, bare soil ranged from 2% to 94%, and distances to nearest water control structures ranged from 8.2 to 427.0 m. Such a wide spectrum of road attributes with relatively low sediment delivery rates indicate that contemporary BMPs can minimize problematic road attributes and reduce erosion and sediment delivery. Additionally, these results support existing state BMP recommendations for water control on stream crossing approaches and site-specific implementation. In general, percent bare soil exceeding 50% and improperly installed or maintained water control structures tended to yield greater amounts of sediment. Currently, the VDOF BMP manual recommends that approaches should be stabilized with rock, corduroy, brush, or other non-erodible surface for minimum distance of 15.2 m (50 ft). This research study showed that approaches with less than 50% bare soil and proper management of surface runoff with effective water control structures could often reduce sediment delivery to streams. Similar recommendations could be added to the VDOF BMP manual in order to optimize economic efficiency.

Soil erosion models encompass our understanding of the physical soil erosion process, but numerous information gaps confound our ability to accurately assess the impact of forestry BMPs on stream sediment. Model performance must be evaluated for a range of operational situations and site conditions in order to partially confirm the utility of soil erosion models for forest management. In the second study, three erosion models, USLE-forest, RUSLE2, and WEPP were compared to trapped sediment data from the 37 forest haul road stream crossing approaches in the first study. The second study assessed model performance from five variations of the three erosion models that have been used in previous forest operations research, USLE-roadway, USLE-soil survey, RUSLE2, WEPP-default, and WEPP-modified. The results suggest that these soil erosion models could estimate erosion and sediment delivery within 5 Mg ha^{-1} yr^{-1} for most approaches with erosion rates less than 11.2 Mg ha^{-1} yr^{-1}, while model estimates varied widely for approaches that eroded above 11.2 Mg ha^{-1} yr^{-1}. Based on the results from the 12 evaluations of model performance, the WEPP variation with numerous parameter modifications (WEPP-modified) consistently performed greater compared to all other model variations tested. However, WEPP-modified estimates above 11.2 Mg ha^{-1} yr^{-1} clearly needed additional improvement. WEPP requires input parameters that are not readily available and easily measured, yet modifications to input parameters are necessary fit the site conditions. WEPP estimates with default parameters (WEPP-default) ranked worst in 75% of model performance metrics. These results indicate that the WEPP model has limited utility as a forest management tool because of the time, effort, modeling expertise, and uncertainty associated with model results. However, WEPP is well suited for researchers and government agencies that have the capability to measure extensive parameter data and conduct sensitivity analyses. Although the USLE-forest can be applied easily in forest management, caution should be exercised when interpreting its results. State agencies are interested in the use of models for estimation of road and ditch BMP effectiveness and sediment delivery. However, results from the study suggest that additional field evaluations and improvement of soil erosion models are needed for stream crossings. Furthermore, sensitivity analyses are needed to enhance our understanding of uncertainty of input parameters, model parameterization, and model structure for forest roads within the WEPP and USLE models.

The third study was conducted to evaluate the sediment control effectiveness of four commonly recommended ditch BMPs on forest haul road stream crossing approaches. A secondary objective was to quantify and compare costs associated with different BMPs to sediment reduction. Sixty ditch segments near stream crossings were reconstructed using a bulldozer and tractor and five ditch BMP treatments were applied within a completely randomized design. Ditch treatments were bare (Bare), grass seed with lime fertilizer (Seed), grass seed with lime fertilizer and erosion control mat (Mat), rock check dams (Dam), and completely rocked (Rock). Mat treatments had significantly lower erosion rates than Bare and Dam, while Rock and Seed produced intermediate levels. Findings of this study suggest Mat, Seed, and Rock ditch BMPs were effective at reducing erosion, but Mat was most effective directly following construction because Mat provided immediate soil protection measures. The Seed treatment is a viable alternative if funds are limited, but should be applied in areas where grass seed has a high likelihood of germination. When planning which erosion control techniques to utilize, it is important to remember that a variety BMPs can be used to achieve similar levels of water quality protection and can be selected to fit site specific circumstances. The most effective ditch BMP treatments will be those selected by a trained professional having adequate site and operation knowledge. Percent bare soil values were significant predictors of ditch erosion and explained 50% total variation in erosion rates. Any BMP that can reduce bare soil can provide reduction in erosion and even natural site condition, including litterfall and invasive vegetation, which can provide erosion control. However, ditch BMPs cannot totally mitigate inadequate water control structures. A ditched road segment that handles an oversupply of runoff, is too steep, or has unfavorable conditions that generate erosion cannot totally be ameliorated by the use of ditch BMPs alone. In contrast, ditch segments with favorable characteristics such as low grades and suitable water runoff loads may not require ditch BMPs beyond bare treatments. When BMP costs are included into the evaluation, Seed and Mat treatments clearly have advantages over the more expensive Dam and Rock BMP treatments.

Overall, forest roads and stream crossings have the potential to be major contributors of sediment in forested watersheds when roads are not designed well or when BMPs are not properly implemented. Forestry BMPs reduce stormwater runoff velocity and volume from forest roads, but can have varying levels of effectiveness due to site-specific conditions. Operational field studies provide valuable information regarding erosion and sediment delivery rates, which helps guide BMP recommendations and subsequently enhances water quality protection. The research results presented in this dissertation can be used to evaluate the efficacy of BMPs and demonstrates the capability of forest managers to assess and select BMPs to fit site-specific conditions. Soil erosion models may be useful tools in ranking erosion prone stream crossing approaches in the future, but require further research and development. Such efforts are needed to evaluate erosion control techniques across broad spatial scales and help focus limited management budgets. Additionally, this field experiment data may assist policymakers in their assessment of the need for policy changes. Results from these studies suggest that BMPs are effective when properly implemented. In summation water quality protection is the result of proper location of roads with attention to stream crossing design and construction techniques, coupled with adequate implementation of appropriate BMPs, and followed with road maintenance monitoring.