USDA
United States Department of Agriculture

Effectiveness of Best Management Practices that Have Application to Forest Roads:
A LITERATURE SYNTHESIS

Pamela J. Edwards, Frederica Wood, and Robin L. Quinlivan

Forest Service
Northern Research Station

General Technical Report NRS-163
October 2016

ABSTRACT

Literature describing the effectiveness of best management practices (BMPs) applicable to forest roads is reviewed and synthesized. Effectiveness is considered from the perspective of protecting water quality and water resources. Both paved and unpaved forest roads are considered, but BMPs that involve substantial engineering are not considered. Some of the BMPs included are commonly used on roads; others are used less often. The synthesis focuses on quantitative BMP effectiveness and descriptions of processes or characteristics that influenced the effectiveness. Qualitative results and observations not supported by data are excluded. Most of the effectiveness results describe sediment losses and sediment delivery, but there is also some coverage of chemicals used as BMPs, such as dust palliatives and soil conditioners. Chapters and subheadings are based on how or where protection is provided, or type of BMP. The final chapter provides information on research needs and potential direction of BMP implementation in the future. Although there remains a great need to quantify BMP effectiveness more rigorously across more physiographic, topographic, climate, and soil conditions, the data provided in this synthesis give road and watershed managers and landowners a starting place for evaluating and selecting BMPs.

Cover photos: Inset photos, clockwise from top left: Compost filter sock in a roadside ditch (Composting Association of Vermont, used with permission); a forest road with a broad-based dip (U.S. Forest Service, San Dimas Technology and Development Center); erosion control blanket with slope interrupters on a road fillslope (Filtrexx International, used with permission); a newly constructed bridge at a forest stream crossing (Barb Ellis-Sugai, U.S. Forest Service). Background photo: Vista overlooking an eastern mixed forest (U.S. Forest Service, Southern Research Station via Bugwood.org).

Quality Assurance: This publication conforms to the Northern Research Station's Quality Assurance Implementation Plan which requires technical and policy review for all scientific publications produced or funded by the Station. The process included a blind technical review by at least two reviewers, who were selected by the Assistant Director for Research and unknown to the author. This review policy promotes the Forest Service guiding principles of using the best scientific knowledge, striving for quality and excellence, maintaining high ethical and professional standards, and being responsible and accountable for what we do.

The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

Manuscript received for publication January 2016

Published by:

U.S. FOREST SERVICE
11 CAMPUS BLVD., SUITE 200
NEWTOWN SQUARE, PA 19073-3294

October 2016

For additional copies, contact:

U.S. Forest Service
Publications Distribution
359 Main Road
Delaware, OH 43015-8640
Fax: 740-368-0152

www.nrs.fs.fed.us

Effectiveness of Best Management Practices that Have Application to Forest Roads:

A LITERATURE SYNTHESIS

Pamela J. Edwards, Frederica Wood, and Robin L. Quinlivan

PAMELA J. EDWARDS is a research hydrologist with the U.S. Forest Service, Northern Research Station in Parsons, WV; FREDERICA WOOD is an information technology specialist with the U.S. Forest Service, Northern Research Station in Parsons, WV; and ROBIN L. QUINLIVAN was formerly a biological science technician with the U.S. Forest Service, Northern Research Station in Parsons, WV.

TABLE OF CONTENTS

CHAPTER 1

Introduction

Every year, the U.S. Forest Service's San Dimas Technology and Development Center (SDTDC) sends out a Request for Proposals to agency field units to identify field personnel's priority needs that fall within SDTDC's mission areas. The need for a literature synthesis describing the effectiveness of road best management practices (BMPs) originated from such a request. Field personnel identified potential applications of a literature synthesis. One use would be to provide information that could be used for selecting appropriate BMPs and supporting mitigation strategies in environmental documents (for example, environmental impact statements and environmental analyses). Another would be to furnish material to support adaptive management for reducing problems associated with road construction and use.

Literature and associated data included in this synthesis come from a variety of sources; some of the references have been peer reviewed and some have not been. Journal papers; books; interim and final reports submitted to local, state, and federal agencies; graduate student theses; unpublished reports by research scientists, students, and companies; published state and federal documents; published and unpublished university documents; and information from Internet sites that we consider to be reputable (i.e., primarily federal, state, and university extension sites) are among the pieces of information that we reviewed. Data of unknown origin or data of obviously suspect quality found on the World Wide Web are not included. We have not attempted to rate or evaluate the quality of the data from the perspective of scientific rigor or statistical power. That analysis is outside the objectives and scope of this review, though the reader can pursue that objective if desired by returning to the originally cited sources. Instead, data are presented to provide the greatest breadth of information possible; this was deemed important because the amount of effectiveness data available for many BMPs is relatively limited (Anderson and Lockaby 2011a, Moore and Wondzell 2005). International System (metric) and English units are employed in this synthesis, and with few exceptions the units used are those in the citations. For readers' convenience, English, metric, and gradient conversions can be found in the Appendix on pages 170 and 171.

BMP effectiveness is at the heart of this report; however, descriptions of the BMP characteristics or processes that reduce pollutants are provided to augment explanations of BMP success or failure in specific situations and to provide the reader with a better understanding of the appropriate applications or limitations of the BMP. Rarely are only qualitative or observational results given from a reference. Where they are used in this synthesis, they were extracted from research studies that included data, and the lack of data supporting the observation is noted. Some reports (especially those by agencies in state departments of highway or transportation) included comments or observations by motor-grader operators or other equipment operators on BMP effectiveness. We did not include these observations because they are subjective and lack measured data or any other type of quantified BMP information to support them. We are not suggesting such observations have no merit, but because they are not otherwise scientifically supportable, they have been excluded. Along this same line, we emphasize that this synthesis **does not** simply describe or summarize BMPs that are used on or are applicable to roads (e.g., from state BMP manuals), because our intent was to avoid implying that broad acceptance and implementation of a BMP guarantee effectiveness or that its performance is well supported by research.

To maximize the amount of available effectiveness data presented in this literature synthesis, studies and results are not restricted to road research or road applications.

BMP information and effectiveness results that have application to roads but originate from a variety of other resources, particularly agriculture, have also been included. As such, readers should be aware that levels of effectiveness reported in nonroad applications may not represent the degree of effectiveness if the BMP were applied to roads.

BMPs are discussed within this document only from the context of their effectiveness for controlling nonpoint source pollution. This was the original context of the term "best management practices" within the Federal Water Pollution Control Act (Clean Water Act) of 1972 as amended, even though the term "BMP" has since been appropriated and applied to management considerations far beyond nonpoint source pollution (Aust and Blinn 2004).

This review does not consider passage of aquatic organisms. Although aquatic organism passage clearly has application to the Clean Water Act, it is not central to the theme of nonpoint source pollution **control**. Readers interested in aquatic passage are directed to annotated bibliographies by Anderson and Bryant (1980) and Moore et al. (1999). Road decommissioning (including "putting skid roads to bed") also is explicitly excluded because the focus of this review is road construction, presence, and use. A wide variety of actions can be taken to decommission a road (e.g., for the U.S. Forest Service, see U.S. National Archives and Records Administration 2015), so this topic is sufficiently broad to warrant its own literature review in another outlet. However, some effectiveness information provided for BMPs associated with road construction/use may be applicable to decommissioned roads.

Forest roads are constructed to a wide range of standards. For example, the U.S. Forest Service has the largest single ownership of roads in the United States: about 370,000 mi of system roads (Foltz 1999, Peters and Peters 2009), which can be maintained at one of five different levels (Ruiz 2005) (Table 1). Most of these roads are typical of what people envision when they think of forest roads: unsurfaced or graveled roads. But the quality of those varies from low-standard roads passable only by high-clearance four-wheel drive vehicles to high-standard roads that can be traveled comfortably at moderate speeds. Some forest roads are paved because they have high traffic volumes or high volumes of very heavy vehicles and thus require paving to protect the road surface from degrading. Most BMP studies included in this document involve lower-to-moderate standard roads, and skid roads when information is available. Because woods roads can include paved roads, however, there is some consideration of those roads (i.e., paving as a BMP) when effectiveness data were available. BMPs that require substantial design and engineering to implement and that are prescribed for very high volume roads (i.e., maintenance level 5 roads in Table 1) are beyond the scope of this synthesis.

The primary focus of BMP effectiveness in this synthesis is erosion and sediment control as sediment is the most common water pollutant associated with forest roads and forest operations (Stuart and Edwards 2006). Runoff and sediment also may carry other road- or traffic-derived pollutants (e.g., motor oil or other petroleum by-products), so other pollutants are discussed when applicable. For example, because road use can result in toxic metal contamination (Rogge et al. 1993), some BMPs have been evaluated for their effectiveness in reducing metal concentrations. Likewise, some chemicals that are used as BMPs (e.g., dust abatement chemicals, soil conditioners, deicing chemicals) have the potential to pollute nearby water bodies, so the pollution potential of such chemicals also is discussed. The effectiveness of BMPs at controlling nutrient losses, including phosphorus (P), which often is bound to mobile sediment, is not considered. Nutrients are excluded for two reasons: 1) Most nutrients are not pollutant concerns during road construction, use, or maintenance; and 2) in the case of P, focusing on sediment control typically is more informative in the context of BMP effectiveness—if sediment is controlled by a BMP, sediment-bound P usually is controlled.

Within the text, we sometimes use wording such as "the BMP was effective." In these instances we simply mean that the BMP resulted in greater reduction of a pollutant (usually sediment) compared to no implementation of the BMP or compared to another BMP. However, such broad use of the concept of "effectiveness" fails to address the more essential question: What level of effectiveness is sufficient to label the BMP as effective? That question is complicated because it does not have a single answer. For some uses, BMP effectiveness may be

Table 1.—Abbreviated descriptions of U.S. Forest Service road maintenance levels from Forest Service Handbook 7709.59, Chapter 60, 62.32 (U.S. Forest Service 2009)

Road maintenance level[a]	Associated road characteristics
1	These are roads that have been placed in storage between intermittent uses. The period of storage must exceed 1 year. Basic custodial maintenance is performed to prevent damage to adjacent resources and to perpetuate the road for future resource management needs. Emphasis is normally given to maintaining drainage facilities and runoff patterns. Planned road deterioration may occur at this level. Appropriate traffic management strategies are "prohibit" and "eliminate" all traffic. Roads receiving level 1 maintenance may be of any type, class, or construction standard, and may be managed at any other maintenance level during the time they are open for traffic. However, while being maintained at level 1, they are closed to vehicular traffic but may be available and suitable for nonmotorized uses.
2	Assigned to roads open for use by high clearance vehicles. Passenger car traffic, user comfort, and user convenience are not considerations. Warning signs and traffic control devices are not provided with the exception that some signing, such as "No Traffic Signs" may be posted at intersections. Motorists should have no expectations of being alerted to potential hazards while driving these roads. Traffic is normally minor, usually consisting of one or a combination of administrative, permitted, dispersed recreation, or other specialized uses. Log haul may occur at this level. Appropriate traffic management strategies are either to: a. Discourage or prohibit passenger cars, or b. Accept or discourage high clearance vehicles.
3	Assigned to roads open and maintained for travel by a prudent driver in a standard passenger car. User comfort and convenience are not considered priorities. Warning signs and traffic control devices are provided to alert motorists of situations that may violate expectations. Roads in this maintenance level are typically low speed with single lanes and turnouts. Appropriate traffic management strategies are either "encourage" or "accept." "Discourage" or "prohibit" strategies may be employed for certain classes of vehicles or users.
4	Assigned to roads that provide a moderate degree of user comfort and convenience at moderate travel speeds. Most roads are double lane and aggregate surfaced. However, some roads may be single lane. Some roads may be paved and/or dust abated. The most appropriate traffic management strategy is "encourage." However, the "prohibit" strategy may apply to specific classes of vehicles or users at certain times.
5	Assigned to roads that provide a high degree of user comfort and convenience. These roads are normally double lane, paved facilities. Some may be aggregate surfaced and dust abated. The appropriate traffic management strategy is "encourage."

[a] See Ruiz (2005) for photographs of roads in each maintenance level.

interpreted as we did in this document—any reduction in a nonpoint source pollutant. In other circumstances, BMP effectiveness may need to meet some minimum threshold of nonpoint source pollutant reduction before it is considered effective. The cost-to-benefit ratio of the BMP may be another factor in determining if a BMP is sufficiently effective for implementation. Due to the subjective nature of and many ways for defining and interpreting effectiveness, we intentionally have made no further attempt to define effectiveness throughout the chapters. Just as the reader is left to evaluate the rigor of the research and quality of the data and studies presented, the reader also is responsible for further interpretation of the pollutant reduction values cited from these works. It is up to the reader to determine if the BMP is sufficiently effective to warrant implementation in the field or citation in written documents (e.g., environmental analyses).

In describing BMP effectiveness, the fundamental "unit" we have concentrated on is the individual BMP or a few bundled BMPs (i.e., several BMPs were grouped and effectiveness was reported for the group). Consequently, the focus is on studies where effectiveness of the individual or bundled practices was isolated and quantified. This approach thereby excludes many studies that conventionally are used to demonstrate overall BMP effectiveness on a watershed basis (e.g., Arthur et al. 1998, Brown 2010, Kochenderfer and Hornbeck 1999, Kochenderfer et al. 1997, Lynch and Corbett 1990, Reinhart et al. 1963). These types of studies depend upon the examination of water quality (e.g., turbidity, sediment concentrations, or total suspended solid loads) at the mouth of a watershed. Thus, in addition to not being able to ascribe quantifiable levels of effectiveness to specific BMPs, readers are advised to use caution in interpreting the results of these studies. Hillside and in-channel storage of eroded sediment as well as lags in sediment delivery to the mouth of the watershed may result in incorrect interpretations or overestimations of BMP effectiveness (Edwards 2003).

The approach of examining individual BMPs clearly is at odds with how BMPs are applied within a project or watershed. Typically, multiple BMPs are planned and applied, with some BMPs even providing overlapping or redundant protection (Stuart and Edwards 2006). For example, cross-drain spacing requirements and road graveling both contribute to controlling overland flow energy and sediment transport. The interdependency and redundancy of some BMPs created some challenges to deciding how this literature synthesis should be organized. Ultimately we decided to define chapters and subheadings based on categories of how or where protection is provided or on types of BMPs. In the end, we strived to condense information from almost 800 references into a state-of-the-science document that will be useful to a diversity of landowners and forest resource specialists, and for a variety of applications, both within and outside the U.S. Forest Service.

A forest road in West Virginia during autumn. (Photo by U.S. Forest Service, Northern Research Station.)

CHAPTER 2

Road Planning

Research has consistently shown that roads increase erosion and sedimentation more than any other practice associated with forest management (Megahan and King 2004). This is the reason that road planning is repeatedly noted as the single most important BMP (Grace 2002b, Kochenderfer 1970). Through proper planning, most deleterious effects to soil and water resources are thought to be avoidable, thereby reducing the need for additional BMPs to mitigate less-than-optimally-planned roads. Planning is believed to provide greater environmental protection (versus without planning) while simultaneously controlling costs of road construction, BMP implementation, and long-term maintenance.

Road planning really involves all phases of road construction, including the pre- and post-activities to ensure the road meets objectives such as duration and level of use. As such, every chapter in this document could be included under the heading Road Planning. As stated in the Introduction (Chapter 1), however, BMP topics have been artificially separated to simplify presentation of material. Consequently, in this chapter, road planning topics are restricted to a subset of issues that would be defined during the preconstruction period. These are: road location, road profiles (i.e., construction designs, such as cut-and-fill or full bench), on-road drainage techniques (including drainage accomplished by road surface geometry and drainage structures), and cross-drain spacing. Note that road drainage considerations in this chapter focus only on BMPs associated with drainage **on** the driving surface. The effectiveness of BMPs used to control the effects of water diverted off the road is covered in Chapter 7.

Road Location

To the degree possible, well-located roads simultaneously 1) avoid high-risk areas (Megahan and King 2004), 2) maximize the distance between the road and water bodies (Megahan and King 2004), 3) minimize the number of water body crossings (Egan 1999, Megahan and King 2004), 4) minimize the total area disturbed by roads (miles and width of road) (Megahan and King 2004), and 5) control road grades at acceptable levels (Haussman and Pruett 1978, Packer 1967). The first four of these location criteria are considered within this section. Road grades are discussed later in this chapter within the context of cross-drain spacing. Road location, with respect to streams and stream crossings, is discussed from the more general perspective of whether distance acts as an effective BMP for controlling sediment delivery. More in-depth discussion of the effects of distance on sediment delivery with respect to buffer lengths can be found in Chapters 5 and 7.

No single optimal road location exists within any parcel of land because the final location depends on the importance (qualitatively or quantitatively) implicitly or explicitly assigned to each of the road location criteria during road layout. But no matter how each variable is weighted, planning can benefit all of these variables. For example, early research showed planning skid road locations reduced per-acre lengths an average of 37 percent, and reduced the total area in skid roads by 40 percent compared to allowing layout at the time of logging. Planned skid road grades also were an average of 33 percent less (Mitchell and Trimble 1959, Trimble and Weitzman 1953).

From the perspective of road BMPs and the Clean Water Act of 1972, high-risk areas are primarily three types of areas susceptible to mass failures (specific regions, specific geology or soil, steep terrain) and water bodies (predominantly streams). Mass failures are a concern because they can deliver excessive amounts of sediment to downslope streams and rivers (DeGraff 1990, Larsen and Torres-Sánchez 1992, Megahan et al. 1978), which can degrade aquatic habitats (Beschta 1978, Cederholm et al. 1981, Eaglin and Hubert 1993, Harr and Nichols 1993) as well as water quality. Streams and other water bodies are critical because they are the objects of protection for the Clean Water Act, and because roads can act as channel extensions or provide direct conduits for sediment delivery. Furthermore, disregarding water bodies during road planning can make a road impassable periodically or even permanently.

Mass failures are not common everywhere. They tend to be more prevalent in specific regions, geologies, or soil types, and often in steep terrain with excess water (Anderson 1983, Beschta 1978, DeGraff 1990, Kingsbury et al. 1991, Maharaj 1993, Montgomery and Dietrich 1994). Duncan et al. (1987) showed the frequency of mass failures in parts of the Pacific Northwest increased exponentially with slope from 12 percent to ≥35 percent. Apparently, however, this relationship does not increase infinitely, as other characteristics of steeper slopes often make them less susceptible to landslides (Maharaj 1993, Megahan et al. 1978, Mehrotra et al. 1991). Not surprisingly, landslides also are often linked to the occurrence of large rain events or extremely wet periods (Anderson 1983, Chatterjea 1994, DeGraff 1990, Larsen and Torres-Sánchez 1992, Maharaj 1993, Moore et al. 1991, Scatena and Larsen 1991). In addition, many local naturally occurring variables contribute to landslide potential. The list of these is long, and an in-depth discussion of them is beyond the scope of this review. But some important factors are soil characteristics (Carrara et al. 1991, Maharaj 1993, Megahan et al. 1978, Sessions et al. 1987, Swanston 1974), lithology and bedrock characteristics (Carrara et al. 1991, DeGraff 1990, Maharaj 1993, Mehrotra et al. 1991, Swanston 1974), and hillside shape, including curvature and topographic convergence (Anderson 1983, Duncan et al. 1987, Montgomery and Dietrich 1994, Sidle et al. 1985). More uncommon events, such as seismic activity, also can trigger landslides (Brabb 1995, Moore et al. 1991).

The existence or construction of roads has been found to exacerbate landslide potential. Roads have more effect on landslide creation than does any other forest management activity (Megahan and King 2004, Moore et al. 1991). In Idaho, 88 percent of new winter and spring landslides surveyed over a 3-yr period (1974-1976) and 57 percent of new landslides inventoried after the winter of 1995–1996 were associated with the presence of roads (McClelland et al. 1997, Megahan et al. 1978). In an unpublished 1965 U.S. Forest Service report, Jensen and Cole reported 90 percent of landslides surveyed in parts of the South Fork of the Salmon River in Idaho were associated with roads (Megahan et al. 1978). An analysis in the Olympic National Forest in Washington state showed that 90 percent of sites with slope failures or high risk for slope failures had road-related factors (Lewis 1995), and Amaranthus et al. (1985) found that 60 percent of debris slides inventoried in the Klamath Mountains of Oregon were associated with roads. Debris avalanche erosion was 25 to 340 times greater where roads were present in the Pacific Northwest than in unroaded forests (Swanston and Swanson 1976). In the northern Rocky Mountain province in Idaho, Megahan et al. (1978) used a number of reconnaissance techniques and estimated that roads were associated with 58 percent of landslides, whereas only 3 percent of landslides occurred on undisturbed hillsides. Road cuts were twice as likely to cause landslides as road fills, but the latter were more likely to reach streams downslope. In highly fractured and weathered bedrock and soils, roads with steep road cuts were the most common sites of landslides in Jamaica; more than 50 percent of the landslides were associated with road cuts (Maharaj 1993). In contrast to many other studies, in western Oregon landslide frequency was not strongly tied to the presence of roads, but road-associated landslides contributed far more sediment to stream channels than did landslides at all other locations (Fredriksen 1970). Mass failures most commonly affected locations where roads intersected stream channels.

In some instances, areas of landslide activity also contained other hillside disturbances associated with the road, such as logging. In the northern Rocky Mountain province of Idaho, 30 percent more mass failures occurred in the presence of logging or fires with roads than from roads alone (Megahan et al. 1978). Landslide frequency was 3 to 26 times greater on hillsides with road building and logging in northwestern California, compared to nearby undisturbed forest land (Wolfe and Williams 1986).

Because of the influence of soil wetness on mass failures, road drainage control is critical to reducing slope failures (Megahan and King 2004). Poor drainage or the lack of drainage from roads concentrates flow (including intercepted interflow from upslope contributing areas), and has been responsible for hillslide failures after large rainfalls (Maharaj 1993, Sidle et al. 1985). Megahan et al. (1978) determined that lack of road drainage caused 27 percent of the landslides studied in Idaho. Dyrness (1967) reported that 4 of 47 slope failures in the Oregon Cascades, including the largest (in terms of material moved), were associated with blocked road drainage systems.

Relationships between mass failures and the presence of roads are illustrated by these and many other studies worldwide (for example, see Sidle et al. [1985] for a comprehensive review of mass failures and land use). Consequently, locating roads to avoid landslide-prone areas is a prudent BMP; however, the effectiveness of locating new roads outside these high-risk areas cannot be quantified directly. Thus, the existing known relationships, such as those described earlier, must be employed as surrogates of measurements of their effectiveness.

Locating roads to avoid landslide-prone areas can be achieved most successfully by identifying areas believed to be susceptible to mass failures based on local conditions and risk factors (Chatwin et al. 1994, Hammond et al. 1992, Larsen and Parks 1997, McClelland et al. 1997, Megahan et al. 1978, Montgomery and Dietrich 1994, Swanston 1974) and identifying the length (i.e., perpendicular to the contour) of the area typically influenced by mass wasting disturbance. This can be done with soil mapping, geotechnical investigations, and measurements of where roads have been constructed and landslides have occurred (Larsen and Parks 1997). Lower, middle, and upper landscape positions are all susceptible to mass failures (Amaranthus et al. 1985, Beschta 1978, Duncan et al. 1987, Fredriksen 1970, Megahan et al. 1978), so the analyses should include all slope positions. Roads outside of areas susceptible to mass failures still may trigger mass failure due to long lengths of influence, and alternatively, roads within the length of influence also may be affected by mass failures (e.g., covered by debris during failure) that other factors such as large precipitation events caused (Larsen and Parks 1997, Wemple et al. 2001).

Where roads cannot be avoided, the occurrence of mass failures can be reduced by controlling road width or length, or both dimensions. Widening an existing road in Malaysia resulted in a landslide during an intense rain event (Douglas 1967). Main access roads in Idaho had 3.4 mass failures per kilometer of road compared to narrower spur roads, which averaged only 0.8 mass failures per kilometer (Megahan et al. 1978). Sessions et al. (1987) found that using steeper roads with fewer miles across steep landslide-prone areas reduced landslide frequency as long as they were well maintained. This is because the shorter total road length resulted in two advantages: a greater proportion of the road system located on ridgetop areas, which are less susceptible to landslides, and smaller volumes of excavated material.

Although road-location BMPs for landslide-prone areas are focused primarily on entirely avoiding those areas, the objectives for reducing road impacts to streams in areas not prone to landslides are focused largely on maximizing distances between roads and streams, minimizing road length, and minimizing the number of stream crossings to the degree possible. There is broad acceptance that proper location of roads is critical to reducing stream sedimentation; roads are a major source of sediment and as much as 90 percent of sediment is attributable to roads (Megahan and King 2004, Packer and Christensen 1964). Much of the consensus about the importance of road location originates from the extensive amount of data that show the connection between the presence of roads and changes in watershed hydrology and sediment delivery, rather than from designed road location studies.

Differences in the degree of connectivity within different watersheds are due to topographic factors, road location and density, road drainage characteristics, and other conditions (Croke et al. 2005, Mockler and Croke 1999, Montgomery 1994, Skaugset and Allen 1998, Takken et al. 2008, Wemple et al. 1996), but road-to-stream connectivity is common. In the western Cascade Mountains, Wemple et al. (1996) found that 57 percent of the 350 km of road lengths surveyed were hydrologically connected to stream channels, so drainage density was effectively increased by 35 and 39 percent for two different basins. In Australia, Mockler and Croke (1999) estimated that hydrologic connections with the road system affected 44 percent of a stream network and 100 percent of the main channel of a 57-km^2 basin. Skaugset and Allen (1998) in Oregon and Bilby et al. (1989) in southwestern Washington state reported similar levels of road-to-stream connectivity: 31 to 39 percent and 34 percent, respectively. La Marche and Lettenmaier (2001) found 24 percent of ditch relief culverts in a 149-km^2 watershed in Washington were hydrologically connected to streams, and 33 percent of all culverts were stream crossing culverts that therefore were connected directly.

Because roads result in much greater sediment production than undisturbed hillsides, their hydrologic connection to streams consequently increases sediment delivery to levels much greater than would occur without roads (Mockler and Croke 1999). This is referred to as "sedimentological connectivity" (though it is not restricted to road-derived sediment) (Bracken and Croke 2007). However, the ubiquity of connectivity between roads and streams and consequent sediment delivery provide a more compelling argument for the effectiveness of minimizing road density (i.e., length per area) than for retaining maximum distance between roads and streams. If distance is important, evidence to support the latter should instead come from relationships between slope position and sediment delivery.

In the absence of roads, landscape position is not always related directly to hydrologic connectivity due to the complexities of hillslope, soil, and flowpath characteristics, but there is a greater probability of that connection if the transport distance is short (Bracken and Croke 2007). In the presence of roads, the potential for hydrologic and sedimentological connectivity is believed to increase as the distance between roads and streams becomes smaller (La Marche and Lettenmaier 2001), but illustrating that increasing the road-to-stream distance alone necessarily reduces hydrologic connectivity or sediment delivery is very difficult due to other factors such as slope and cover. This is because isolating the effectiveness of distance requires sediment reductions to occur solely from deposition on the hillside due to the increased opportunity for infiltration attributable to greater slope length. In reality many other natural hillside features and human-made structures reduce connectivity by slowing drainage or capturing sediment; these include natural obstructions, windrow filters, litter, vegetative cover, and designed erosion control structures (Burroughs and King 1989, Cook and King 1983, Ketcheson and Megahan 1996, King 1979, Megahan and King 2004, Packer 1967, Packer and Christensen 1964, Wasniewski 1994) and can be at least as important as available distance between roads and streams.

Another problem with demonstrating the effectiveness of maximizing distance between roads and streams is the high variability in the amounts of sediment that originate from different road segments, as most sediment comes from only a small minority of road segments (Croke et al. 2005, Luce and Black 1999, Takken et al. 2008). Many problematic road segments are the result of inadequately spaced drainage features that allow concentrated flow from cross drains to reach streams (Skaugset and Allen 1998, Takken et al. 2008, Wemple et al. 1996). This is the principal source of connectivity aside from the direct connection at stream crossings. Skaugset and Allen (1998) found that 10 percent of drainage locations on roads that delivered sediment to streams (i.e., non-crossings) were from random, non-engineered points along roads in Oregon, whereas about twice that many origination points (19 percent) were cross drains. Wemple et al. (1996) reported that gullies which acted as stream channel extensions occurred below 25 percent of the cross drains surveyed. In a study by Croke et al. (2005), cross-drain culverts were 10 percent of all drain types surveyed, but 90.5 percent of them connected directly to streams by gullies. Because gullies can transport runoff and associated sediment almost unimpeded very long distances once they become established, even very long hillside distances between roads and water bodies can be rendered ineffective in preventing sediment delivery.

Even with acknowledging the difficulties of isolating hillside distance as an important variable in controlling sediment transport, it seems intuitive that decreasing the distance between roads and streams should increase the potential for road to stream connectivity. Some evidence supports high risk of sediment delivery from valley bottom roads that are close to parallel streams (Takken et al. 2008). In watersheds throughout several geographic regions in western Oregon, Skaugset and Allen (1998) reported that roads in lower valley segments constituted only 11 percent of the roads surveyed, but 59 percent of those delivered, or possibly delivered, sediment directly to streams. In these situations, advantages that distance could have provided are largely unavailable;

the flexibility and choices of other suitable mitigation strategies (e.g., erosion control structures between roads and water bodies) become much more limited, making sediment delivery very difficult to control (Swift 1985). It is also almost impossible to add enough cross drains on valley bottom roads to reduce runoff volumes enough to eliminate hydrologic and sedimentologic connectivity. Takken et al. (2008) calculated that cross-drain spacing in some situations on roads in valley bottom segments would have to be no more than 5 m, which is infeasible.

The relationship between distance and sediment delivery is not so clear for midslope and ridgetop positions. Skaugset and Allen (1998) rated about 30 percent of both midslope and ridgetop road segments as delivering or possibly delivering sediment to streams, though the total number of midslope road segments surveyed was much greater than the number of ridgetop segments (~2,450 versus 500, respectively). Midslope connectivity may be more common than often anticipated because these roads are most likely to intercept subsurface flow in cutbanks (Jones 2000b, Wemple et al. 2001), so more water must be handled by cross drains on midslope roads. Cutbanks can intercept subsurface flow when the depth of the cutslope extends below the permanent water table (O'Loughlin 1975, Parizek 1971), when there are transient (e.g., seasonal) water tables that rise above the base of cutslopes, or when discontinuous saturated zones (e.g., springs) exist above the water table (Dutton 2000, Tague and Band 2001, Wemple and Jones 2003). The locations where cutslopes will intercept subsurface flow is difficult to predict and plan for during road construction (Toman 2004). Experience in the physiographic region may be of limited usefulness in anticipating where problems might occur because seemingly similar road segments may not have similar potential for or amounts of cutbank interception (Toman 2004). Midslope roads contribute additional challenges during road location because they also tend to have more stream crossings than upper slope or lower slope roads due to the high drainage densities in midslopes (Takken et al. 2008). These provide a direct connection of runoff and sediment to streams (Weaver and Hagans 2004).

Locating roads in midslope and upslope positions may be somewhat more effective at controlling sediment than locating roads in valley bottoms immediately adjacent to streams, but the degree of protection often may be overassumed, particularly for midslope locations. Concentrated road drainage from midslope roads can extend to streams, even for streams that are relatively far away (Croke and Mochler 2001, Ketcheson and Megahan 1996, Wemple et al. 1996). The length of such connectivity that is possible may be underestimated due to misconceptions about the objectives of road drainage BMPs. Most BMP cross-drain spacing recommendations in the United States are designed to control the energy of water at volumes small enough to control erosion and damage on the road surface (see Cross-Drain Spacing section). Maintaining concentrated road runoff at volumes that do not create hydrologic or sedimentologic connectivity once the water is diverted onto the hillside is not an objective of many recommendations, and the spacings required to meet these two different objectives are probably not equivalent in most instances (Edwards and Evans 2004). Therefore, control of hillside connectivity becomes dependent in part upon hillside erosion control, and also in part on limiting the volumes of water delivered at any one point from the road (described in Chapter 7).

Maintaining distance between roads and streams is probably most effective as a BMP when erosion or sediment transport is associated with dispersed road runoff, or with fillslope erosion from rainfall or dry ravel. In these situations, infiltration and sediment deposition can be achieved relatively easily in short distances because interrill and dry ravel erosion (versus rill and gully erosion) dominate sediment transport (Ketcheson and Megahan 1996).

The presence of stream crossings can strongly influence water quality and stream health because crossings directly contribute runoff and sediment (Kruetzweiser and Capell 2001, Lane and Sheridan 2002, Swift 1988, Weaver and Hagans 2004; also see Chapter 5), suggesting that planning to control the number of crossings can be an effective BMP. Croke et al. (2005) modeled runoff and sediment delivery from a variety of dispersive pathways to streams in a catchment in New South Wales, Australia, and found that the point with the greatest inputs of both was a stream crossing; other individual sources, such as cross drains, yielded substantially less sediment by dispersed pathways. Eaglin and Huber (1993) provide additional evidence of the effects of stream crossing density on sediment delivery. Stream embeddedness and the amount of fine sediment in streams were both significantly and directly related to stream crossing density, and the amount of cobble substrate was significantly and indirectly related to crossing density. Bill (2005) also found stream crossing construction increased the percentage of fines downstream of the crossings and when compared to a stream in an unroaded watershed followed through several years of monitoring.

Road Profiles

The profile of a road refers to the general shape of the road prism and the driving surface. Most forest roads are constructed as cut-and-fill roads. A cut is made into the hillside and the removed material is side cast downslope (Fig. 1). About half of the road driving surface is built on residual soils in the cut portion and half is built on the side-cast fill material. The area from the top of the cutslope above the road to the bottom of the fillslope below the road constitutes the road prism.

Beyond the sediment associated with road driving surfaces (see Chapters 3 and 4) and with landslides or sediment delivery due to the location of roads, fillslopes and cutbanks of cut-and-fill roads have their own unique set of challenges with respect to erosion and sediment control. The design of cut-and-fill roads alters the surface and subsurface hydrology and oversteepens both cutbanks and fillslopes compared to the original hillside slope, though fillslopes tend not to be as steep as cutbanks (Burroughs and King 1989). Additionally, fillslopes generally are composed of unconsolidated material (Edwards and Evans 2004, Megahan and King 2004, Rothwell 1978). Together, these attributes make

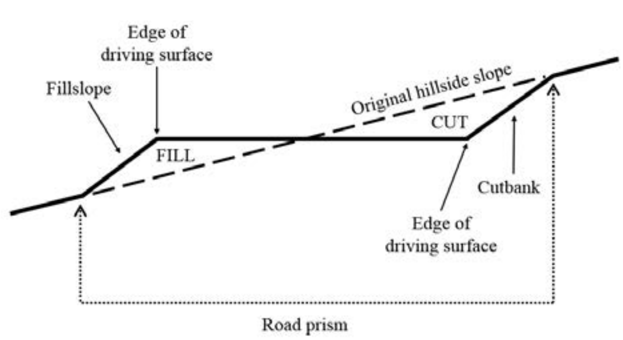

Figure 1.—Schematic of a cut-and-fill road. Soil removed from the cut is used to construct the fill. The road driving surface is composed of both cut and fill areas.

these areas susceptible to erosion until their soils become stabilized, usually through revegetation (see Chapter 6).

Through the life of the road, sediment contributions are generally much less from fillslopes and cutbanks than from an active driving surface (Croke et al. 2006, Reid and Dunne 1984, Swift 1984b). But fillslope and cutbank contributions can be substantial during construction, and they can be chronic in the long term if slope stabilization is not fully successful and concentrated flow from road surfaces is not controlled. Because fillslopes are on the downhill side of road prisms, they can deliver more sediment during construction than cutbanks. Stedman (2008) measured 1,178 kg of soil that was mechanically delivered to a stream from fillslope construction in the approaches[1] (a total of 152 m of length) of three stream crossings in West Virginia.

Even where road fills do not encroach on streams during construction, initial losses from fillslopes are elevated. This is because fillslopes are composed of unconsolidated material that is susceptible to water erosion and dry ravel (King 1984, Megahan 1974a). Conversely, the unconsolidated soil also provides high infiltration capacities (at least for nonconcentrated flow), reducing the potential for runoff and allowing rapid revegetation (Arnáez et al. 2004, Edwards and Evans 2004, Jordán-López et al. 2009). As a result, erosion rates on fillslopes typically decline relatively quickly once vegetation becomes established (Stedman 2008).

The primary situation in which fillslopes result in greater losses of soil than cutbanks after revegetation has become established is associated with fillslope failures. These can be large mass failures or small slides or slumps (Arnáez and Larrea 1995, Pitts 1992). As with other types of mass failures, fillslope failures commonly are related to poor road drainage control (Arnáez and Larrea 1995). Lewis (1995) reported that 70 percent of slope failures or locations where the risk for slope failure was high were on fillslopes, some of which were fillslopes where road drainage had failed and was destabilizing fills.

[1]Approaches are defined as the length of road or ditch line from which water would drain directly to the crossing. The outer boundaries of an approach are usually definable by road-surface drainage features or grade changes on the road surface and ditch line.

Cutbanks can lose substantial amounts of soil during construction. Cerdà (2007) found soil erosion from cutbanks during construction was 30 times greater than previously constructed cutbanks that had about 35- to 55-percent vegetative cover. However, cutbanks tend to be more of a concern in the longer term than fillslopes because sediment production is more chronic due to gravity sloughing, undercutting, disturbance of cutbank vegetation with mowing, and bank failures. Several similarly designed experiments from existing cutbanks and fillslopes in the Mediterranean region of southern Spain illustrate this tendency. Using simulated rain events on 12 cutbanks (1 to 4 m high) and 12 fillslopes, Jordán-López et al. (2009) found cutslopes resulted in 18 times more total average soil loss (486.7 g m^{-2}) than fillslopes (27.2 g m^{-2}). On average, the cutbanks were 50 percent steeper (40-percent grade) than fillslopes (29-percent grade), but soil loss on the fillslopes was statistically ($p < 0.05$) explained only by soil texture and not by other soil cover or slope variables.

Arnáez et al. (2004) measured average total soil losses from simulated rain events on 12 cutbanks and 6 fillslopes of 160.7 g m^{-2} and 10.5 g m^{-2}, respectively. In their study, cutbank slope, which ranged from 60 to 120 percent, was significant (and positive) in explaining soil loss, and coarse fragment cover was significant (and positive) for the fillslopes. Rather than protecting the soil from erosion, increasing gravel content on the fillslope soil surface concentrated water and runoff locations between gravel particles; this is similar to processes noted for stone mulch (Poesen and Ingelmo-Sanchez 1992; also see Chapter 6). Jordán and Martínez-Zavala (2008) measured an average of 106 g m^{-2} from 10 cutslopes and 17 g m^{-2} from 10 fillslopes from simulated rain events, but neither slope gradient, rock cover, nor plant cover was statistically important in explaining the soil losses from either type of slope. Arnáez and Larrea (1995) used erosion pins and also found cutslope erosion exceeded fillslope erosion.

Although these Mediterranean studies do not consistently show cutslope gradient to be important in explaining erosion, it often is. For example, 0.75:1 (horizontal:vertical) slopes resulted in sediment reductions of just 32 to 47 percent over a 3-yr period after mulch, seed, and fertilizer treatments, compared to the 90-percent reductions estimated for more gentle slopes of 1.25:1 (Burroughs and King 1989). Diseker and Sheridan (1971) found steepness to be important in predicting roadside sediment yield in Georgia. For granitic cutslopes in Idaho, Megahan et al. (2001) reported that cutslope gradient was the most important variable when predicting cutslope sediment yield. Results from a road cutslope study by Odemerho (1986) in Nigeria showed a curvilinear relationship between cutslope gradient and sediment yield. Soil losses increased from about 35 tonne ha^{-1} yr^{-1} to 125 tonne ha^{-1} yr^{-1} as cutslope gradient increased from 2 percent to 7 percent, then decreased to less than 20 tonne ha^{-1} yr^{-1} as cutslope gradient continued to increase to approximately 50 percent.

Steep cutbanks are difficult to stabilize with vegetation and to keep vegetated for several reasons. Infiltration often is poor on cutbanks because of crusts that form due to erosion and low organic matter content in cutbank soils (Cerdà 2007). Runoff can wash seed, poorly established vegetation, and even mulch from cutbanks (Bochet and García-Fayos 2004, Buchanan et al. 2002, Burroughs and King 1989, Meyer et al. 1972) (also see Chapter 6). Freeze–thaw processes, frost heaving, and ground ice contribute by uprooting shallowly rooted vegetation and loosening soil, which continues the cycle of erosion (Arnáez and Larrea 1995). Intercepted subsurface flow can destabilize areas on the cutbank near where water comes to the surface and saturates the soil. Soil that has eroded and blocked drainage at the base of the cutbank can further saturate soil and contribute to destabilization of the cutbank (Arnáez and Larrea 1995). Cutslopes also are susceptible to small failures through undercutting or removal of the toe of the slope during ditch or road maintenance (Chatwin et al. 1994, Yee 1976).

Even after 10 yr, Cerdà (2007) found cutbanks at two locations in Spain generally had <50-percent vegetative cover. Bold (2007) also found revegetation was slow to occur on cutbanks with a mean slope of 96 percent in the moist climate of West Virginia. About 2 yr were required for moderate levels of vegetation to become established after cutbank construction and seeding, and after 4 yr the percent cover for cutbank sections with southern aspects averaged only 36 percent compared to 64 to 82 percent for other aspects. In eastern Spain, Bochet and García-Fayos (2004) reported almost no vegetation cover on road cutslopes greater than 100 percent, and <10-percent cover on slopes less than 100 percent, for 6- to 8-yr-old roads; where vegetation did develop, cover was about twice as thick on north-facing versus south-facing cutbanks.

Several types of practices have been developed as BMPs to reduce some of the erosion and sedimentation problems associated with the fillslopes and cutbanks of cut-and-fill roads. These BMPs include using full bench construction and end hauling in place of cut-and-fill roads (at least in high-risk locations), compacting fillslopes, and terracing cutbanks (all of which are described in the following paragraphs) (Burroughs et al. 1976, Cameron and Henderson 1979, Chatwin et al. 1994, Gwynne 1950, Megahan and King 2004, Megahan et al. 2001, Stedman 2008) as well as a myriad of techniques to control drainage on fillslopes (Bethlahmy and Kidd 1966, Burroughs and King 1989, Cook and King 1983, Dudeck et al. 1970, Swift 1985) that are covered in Chapter 7.

Full bench construction involves constructing the driving surface fully from residual soil material so no material is side cast and no fillslope is created (Fig. 2). All of the excavated soil is used to supplement material where needed, including on the road surface and turnouts, and unneeded excess soil is stored in a location safe from sediment transport or taken offsite (i.e., end hauling) (Cameron and Henderson 1979). Thus, the primary advantage of full bench construction is that it eliminates fillslopes and problems associated with fillslope erosion and failure. A disadvantage of this alternative is that building a road of the same driving-surface width requires excavation farther into the upslope hillside, which typically results in a higher cutbank.

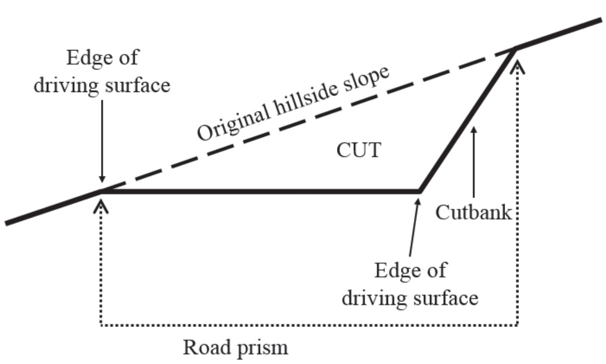

Figure 2.—Schematic of a full bench construction road. The road driving surface is entirely within the cut area since a fillslope is not developed in full bench construction.

Given that large mechanical additions of sediment to streams can result from cut-and-fill roads during construction where roads approach streams (Stedman 2008), and fillslopes are relatively common sources of hillside failures (Arnáez and Larrea 1995, Lewis 1995, Pitts 1992), full bench construction might be considered to be an effective means of reducing erosion and sedimentation in some situations. However, few studies have examined its effectiveness, so there is little direct evidence to support it as a BMP, to illustrate the improvement in effectiveness compared to cut-and-fill-roads, or to identify those situations where it is most effective. An analysis by Sessions et al. (1987) of landslide occurrence compared old construction techniques (late 1960s and early 1970s) using moderately sloped cut-and-fill roads to more modern construction techniques of steeper roads that included a subset of full bench roads (37 percent of new roads were full bench; the remaining were cut-and-fill). They found lower landslide occurrence in full bench roads, but attributed the reduction to the steepness of the roads and not to full bench construction, because steep roads, regardless of construction techniques, could reach ridgetop locations more quickly with shorter road lengths. They suggested construction of steeper cut-and-fill roads with adequate maintenance as an alternative to full bench construction to reduce landslide occurrence.

Whether roads are constructed using cut-and-fill or full bench techniques, cutbanks are created; therefore associated problems must be considered during road planning. In some locations, cutslope height has been associated with intercepted flow (Wemple and Jones 2003); in contrast, La Marche and Lettenmaier (2001) found no relationship between the two. Thus, designing roads to reduce cutbank heights to the degree possible will not guarantee that subsurface flow is not intercepted. Wemple and Jones (2003) found that road segments with the largest cutbank heights tended to respond rapidly to precipitation, resulting in higher unit area peak runoff, and hence, more energy for sediment production (Piehl et al. 1988). Skaugset and Allen (1998) reported that 72 percent of midslope road segments with cutbank heights more than 5 ft were rated as delivering or possibly delivering sediment to streams whereas 60 percent with cutslope heights less than 5 ft had those same ratings.

Much less information is available about other techniques that have been recommended as BMPs to reduce fillslope or cutbank erosion. There is little information in the literature to suggest that fillslope compaction reduces erosion. Instead, compaction may increase erosion. Fillslopes in Idaho were rolled and compacted, and runoff and erosion were compared to uncompacted fillslopes (Boise State University 1984). Compaction reduced infiltration rates and increased runoff and erosion. Sediment yields were 107 to 532 percent greater than for uncompacted fillslopes and averaged 282 percent more.

Similarly there is little information about or support for the effectiveness of terracing cutbanks. Terracing is rarely performed on forest road cutbanks, probably because on forest roads the cutbanks are usually not very tall. Creating terraces on short cutbanks with heavy equipment is difficult, especially in rocky soils. Terraces are more commonly used where cutbanks are relatively tall, and therefore vertical or nearly vertical walls would be unstable, such as along highways or U.S. Forest Service maintenance level 5 roadways. Gwynne (1950) described the appearance of 15-ft-high and 15-ft-wide step terraces that had been created in loess cutbanks that were up to 80 ft tall along an Iowa state highway. A year after construction, some of the risers had started to break apart. North-facing terraces underwent far more damage than south-facing terraces. In some instances so much of the risers had broken off that the cutbank slope had become an almost continuous slope with rills forming. The combination of spalling and water erosion was expected to eventually return the cutbanks to approximately 1:1 slopes. Megahan et al. (2001) found no significant differences in erosion rates of cutslopes treated with hydroseeding and mulch compared to cutslopes that were terraced, hydroseeded, and mulched. Rather than terracing or sloping cutbanks, Swift (1985) recommended building vertical cutbanks on roads that do not have ditch lines and letting them slump and settle to their natural angle of repose, as this technique can reduce both the width needed for the road prism (less distance for laying the slope back) and the size of fills because less material is excavated to create the cutbank. Swift (1985) did not report how much soil was eroded in this process when soil loosened during slumping. This technique is not possible with high vertical banks because too much of the driving surface would be lost to accumulation of sloughing cutbank material, similar to that reported by Gwynne (1950).

The driving surface is the last major part of the road profile. Road shape is used to aid in road drainage to reduce the amount of erosion and sediment transport that occurs along with the amount of water discharged from a road's surface. Road drainage can be achieved with a variety of techniques. These include exploiting natural drainage attributable to road location or hillside shape (e.g., natural outsloping on the noses of ridges) and shaping the road surface (outsloping, insloping, crowning) or installing surface drainage structures (e.g., broad-bassed dips, open-top culverts, water deflection devices).

Outsloping has long been considered a simple and affordable technique to transversely drain water from along much of the road surface so that it cannot accumulate sufficiently to cause erosion or contribute to rutting (Swift 1985, Trimble and Weitzman 1953). An outsloped road is constructed so that the entire driving-surface road width is sloped away from the cutbank at about a 2- to 4-percent grade (Moll et al. 1997). This angle is considered to be sufficient to disperse the water and be safe enough to keep vehicles on the road at allowable speeds, particularly in wet or winter weather. However, slippage off outsloped roads has occurred under wet or icy conditions (Hafterson 1973). Analogously, insloping is designed to transfer all water toward the cutbank at a 2- to 4-percent grade (Moll et al. 1997) and transfer it into a ditch at the toe of the cutbank. The ditch parallels the road and then transfers drainage water (both road drainage and intercepted subsurface flow from upslope) under the road through cross-drain or relief culverts. Crowning includes both of these designs so that half of the drainage water is diverted to the outside of the road and half to the inside of the road. Taking advantage of both insloping and outsloping further reduces the risk of the vehicle sliding to the inside or outside of the road under wet or icy conditions. Crowning is probably the most common road surface shape used on roads (Skaugset and Allen 1998).

There are many claims in the literature about the benefits of each of these types of drainage in reducing erosion on the road surface because water is removed in small quantities, a primary tenet of controlling road erosion

(Croke and Hairsine 2006, Packer 1967). Unfortunately, no field data show that one road shape is preferable to another, or suggest in which situations each is most applicable (Elliot et al. 1999). Nor is there published research to indicate that any of these techniques generally is effective at draining water from roads in the way intended.

Each of these road surface drainage techniques works only if rainwater or snowmelt is effectively dispersed transversely from the road surface over the entire road length (or most of it). However, it is easy for the road surface to become deformed so that transverse drainage is inhibited, and instead, water runs down the driving surface, draining off only sporadically along the road length (Swift 1985). Road deformation and the loss of transverse (inslope or outslope) drainage can occur due to a number of processes. Longitudinal depressions are created in wheel tracks by vehicle loads being concentrated and translated to the road surface by the tires. Wheel tracks develop on all types of roads, including paved roads (Aycock 2009, Moll et al. 1997), where they contribute substantially to hydroplaning (Aycock 2009) and can concentrate flow meant to be dispersed or controlled in ditches. Unpaved roads are particularly susceptible to wheel track development when subjected to heavy equipment such as loaded log trucks. Wheel tracks can become ruts with continued heavy vehicle use during wet weather as the tracks retain water (Elliot et al. 2009). Berms also can form on the inside and outside edges of the driving surface (Swift 1985) and further exacerbate poor functioning of insloped, outsloped, and crowned roads.

Road Drainage Structures

Relief or cross-drain culverts that cross under the road from a ditch line adjacent to the road are the most common way to move water from roads (Piehl et al. 1988). Culverts are an efficient way to transfer drainage onto the hillslope below the road. However, several attributes can influence their effectiveness. Culverts are susceptible to plugging if cutslopes or hillsides fail and deposit soil or debris in the ditch line, or if they are not designed with sufficient slope to flush deposited sediment or organic debris, such as leaves and woody material. Plugging can be exacerbated by mechanical ditch cleaning because the toe of the slope is disturbed, which can increase cutslope erosion or failure and provide a source of sediment. Ditch line cleaning also can disturb the cutbank proper, and remove established vegetation (Luce and Black 2001), which may increase erosion and culvert plugging. In Oregon, Skaugset and Allen (1998) found that 54 percent of cross-drain culverts surveyed were obstructed to some degree, and 60 percent of obstructed cross-drain culverts were blocked by sediment. Piehl et al. (1988) found that 48 percent of ditch relief culverts surveyed had inlets blocked by sediment (24 percent), physical damage (17 percent), or both. Careless road and ditch maintenance also can contribute to culvert failure by partially or fully crushing the inlet of relief culverts with heavy equipment. One-quarter of the relief culverts that had problems reported by Skaugset and Allen (1998) had crushed inlets.

Relief culverts are constructed from a variety of materials, but can have two basic designs: smooth-bored or corrugated. BMP manuals typically recommend corrugated culverts over smooth-bored culverts because the former provides roughness to the inflow, which reduces velocity and erosive potential of discharged water (on the outlet side). In a study of culvert hydraulics related to fish passage, Barber and Downs (1995) measured maximum pipe centerline velocities for a range of discharges and slopes for corrugated and smooth pipes of the same diameter. The smooth pipe produced maximum velocities that were 16 percent higher on average.

Incorrect cross-drain culvert installation and sizing are believed to result in increased sediment production. Improperly sized culverts can plug more easily, leading to culvert overtopping and road surface erosion. Culverts installed too deep at the inlet or at an incorrect gradient can cause sediment deposition, as can culverts installed more perpendicular (less skewed) to the road. Hanging culvert outlets that extend beyond the edge of the fillslope can cause increased scour and gullying. Guidelines for proper cross-drain culvert sizing and installation are widely available (e.g., Johansen et al. 1997, Keller and Sherar 2003, Kramer 2001, Rothwell 1978) and generally are based on federal design standards (Normann et al. 1985). However, research documenting the effectiveness of the guidelines is generally lacking in the literature. A survey by Piehl et al. (1988) of ditch-relief culverts in Oregon found that even though culvert skew angles averaged half the recommended value of 30º, skew angle was not correlated with sediment accumulation at the culvert inlet.

Ditches are inherently a component of cross-drain culverts. The amount of sediment they provide can vary widely. Croke et al. (2006) found that road ditches, when isolated from the road surface and cutslope, contributed 17 to 45 percent of the total sediment yield from cut-and-fill insloping road segments in Australia. Ditch lines can become vegetated because they are moist and accumulate unconsolidated soil. But vegetation is not always effective at controlling ditch erosion, because grasses and herbaceous vegetation may be uprooted and scoured out when subjected to high amounts of energy from concentrated flow (Barrett et al. 1998b).

Ditch hardening with rock is presented commonly as a BMP to reduce ditch erosion (e.g., Berkshire Regional Planning Commission 2001, Elliot and Tysdal 1999), but only a few studies in the literature show how much of a reduction can be achieved with rock lining. Burroughs et al. (1984b) compared sediment production from cutslopes and rocked or unrocked ditches in Idaho. A 4-inch layer of 1.5-inch-diameter clean gneiss rock in ditch segments reduced sediment by 2.3 times compared to unrocked ditch segments. The characteristics of rock (e.g., size, compaction versus loose placement) that provide the most benefit are unknown. These characteristics may be very different from those of rock best suited for driving surfaces due to the amount of water that is intentionally concentrated in ditches.

A possible alternative to ditch hardening with rock is the use of soil conditioners for erosion control (also see Chapter 6). Soil conditioners have been shown to reduce erosion from irrigated agricultural furrows (Lentz et al. 1992; Zhang and Miller 1996a, 1996b), which may have some similarity with bare soil in roadside ditches. But there are no studies in which soil conditioners have been used specifically for erosion control in forest road ditch lines. Although soil conditioners are ideally suited for erosion control by concentrated flow, they are applicable only for the short term (e.g., during vegetation establishment). If long-term erosion control in ditches is possible with soil conditioners, it would be feasible only with repeated treatments.

Broad-based dip installation is another cross-drain technique used commonly on haul roads and other roads of comparable standard to overcome some of the problems associated with maintaining proper road surface shape for cross-drain culverts or for outsloping (Swift 1985). A broad-based dip (Fig. 3) is built as a gentle roll in the road centerline profile, which is carefully outsloped (Haussman and Pruett 1978, Swift 1988). The dip captures road surface drainage and directs it down the fillslope. Dip geometry—length and depth—must be designed to accommodate the traffic levels and vehicle types expected to use the road (Copstead et al. 1998, Hafterson 1973). The road surface contributing drainage to a broad-based dip is usually relatively flat (i.e., neither insloped nor outsloped) or slightly outsloped (Cook and Hewlett 1979). Broad-based dips are best suited for road segments that are no more than 8- to 10-percent grade (Cook and Hewlett 1979, Kochenderfer 1970, Swift 1985). On steeper roads, dips are difficult to build for effective drainage control (Kochenderfer 1995); dips on steeply sloped roads can become overtopped by water during large events so they become ineffective and create the potential for road gullying. If dips are deep enough to contain and drain water on steep roads, they are very difficult to drive over without vehicles bottoming out or high centering (Hafterson 1973).

Wheel ruts can form along the contributing road length of broad-based dips, but the dip itself theoretically compensates for that because water is meant to be turned off the road only at the dips. This suggests that

Figure 3.—A broad-based dip on a forest haul road. From Keller and Sherar (2003).

roads with broad-based dips may be more effective at reducing the occurrence of rill or gully formation than insloped, outsloped, or crowned roads, where wheel tracks may confine drainage for potentially long distances. However, there are no data that compare these road drainage techniques, so broad-based dips cannot be shown to be superior at controlling on-road erosion compared to other drainage techniques. Nor should this assumption be accepted widely without supporting data because much can happen to retard the effectiveness of dips. Ruts can continue through the entire dip so that the top of the dip (at least in the tracks) deteriorates. Water is not diverted across the road but instead remains trapped within the tracks, bypasses the dip, and continues running down the road. Hafterson (1973) showed that 3-inch ruts with 3-inch ridges on the sides of the wheel tracks were sufficient to prohibit lateral drainage and cause a 6-inch-deep dip to fail. An unobstructed downward outsloped angle in the base of the dip (Fig. 3) also must be maintained for dips to function properly. The accumulation of sediment can turn the dip into a mud hole and eventually fill the dip base enough to allow water to run down the road instead of drain off it (Swift 1985).

Drainage obstruction by grass at the outlet of a well-functioning dip can trap sufficient sediment to block the dip outlet (Swift 1985); the effectiveness of one BMP thereby can negate the effectiveness of another. A study by Bold et al. (2007) of 130 broad-based dips on haul roads of multiple ages on the Monongahela National Forest (MNF) in West Virginia reported that only 58 percent were outsloped to MNF specifications (2- to 5-percent outslope), and all dips had some obstruction to outward drainage by berms on each side of the wheel tracks. The dips were not fully dysfunctional in that the differences in elevation were not generally enough to result in overtopping the lower dip boundary. But they were not fully effective either, because most of the dips retained water or the base of the dip remained moist even between storm events. An earlier survey of dips on MNF roads by Eck and Morgan (1987) showed similar results with 27.5 percent of 255 dips categorized as drainage failures due to fillslope erosion, rutting, siltation, or ponding.

Road maintenance is required to restore dip function. However, restoration requires that road maintenance affect the source of the problem, which is the subsoil from which the roadbed was constructed. Swift (1985) found that small dozers or front-end loaders were better at maintaining road dips compared to motor-graders, which tended to fill in the dips.

Open-top culverts are alternatives to broad-based dips on steep road segments (Kochenderfer 1995). A variety of open-top culvert designs and materials have been developed and used (Copstead et al. 1998, Haussman and Pruett 1978, Kochenderfer 1995). If effective, they too provide some of the advantages of broad-based dips in that they can overcome some of the difficulties of water draining down wheel ruts by intercepting it at regular intervals. However, there are no studies showing the effectiveness of open-top culverts or illustrating the situations where they are best suited. But because open-top culverts require regular maintenance to keep them clean and operational (Hafterson 1973, Swift 1985) and erosion increases with increased traffic (Reid and Dunne 1984), the frequency of required maintenance rises substantially as traffic increases. Consequently, this type of drainage structure is appropriate only for very low volume roads.

The "proof" that cross drains are effective BMPs comes primarily from observations and limited measurements of physical road characteristics (e.g., road rutting, washouts) comparing roads with cross drains and those without cross drains. Long-term utility of drained roads also provides evidence of road drainage BMPs. Properly functioning cross-drain structures can maintain roads in conditions that are usable in the long term, though other protection on the road surface, such as surfacing, contributes substantially to reducing soil losses (see Chapter 3). Therefore, it is often difficult to separate the effectiveness of drainage BMPs from other road surface BMPs.

Additionally, there are no studies comparing the effectiveness of different types of cross drains to each other in situations where multiple types might be applicable. This also applies to less common water deflection devices, such as water bars (Copstead et al. 1998) or used tires or rubber belting (Wiest 1998), that are used on low standard roads or skid roads. Thus, no data-based prescriptions for specific types of drainage can be made for specific site conditions at this time. Instead, current recommendations are based on the limitations or capabilities of the drainage features themselves relative to the physical site conditions (e.g., whether the road grade is suitable for a technique), the standard to which the road will be built (e.g., traffic levels, type and speed of traffic expected), and economic considerations. Even though the applications of techniques that will physically match the situation provide some degree of potential for erosion control, this is far from the desired outcome: the ability to choose the technique that provides the best degree of erosion control for the specific conditions.

Cross-Drain Spacing

Roads other than outsloped roads require features or structures, such as relief culverts, broad-based dips, and water bars, to ensure that drainage occurs. The retention of water on the road or the delivery of water from the road is determined by cross-drain spacing (Bracken and Croke 2007). The spacing and location of cross drains are critical as water concentrated on the road surface provides the energy for erosion and sediment transport. Road surfaces concentrate water because they are highly compacted with low infiltration rates (Reid and Dunne 1984), leading to rapid generation of overland flow. In terms of stream power, the erosion potential of overland flow routed to cross drains is proportional to the product of road surface slope and discharge (Croke and Hairsine 2006). For roads studied in the Oregon Coast Range, sediment production was correlated to the product of road segment length and the square of the slope, showing slope was most important (Luce and Black 1999). Thus, longer and steeper gradients result in surface runoff with greater velocity and energy (i.e., shear strength) and greater potential for erosion than low-gradient roads. Increased spacing between cross drains on low-gradient roads has much less effect on erosion than it does on high-gradient roads (Luce and Black 1999) because kinetic energy increases with the square of velocity, which is tied to gradient.

Road surfacing in the form of gravel or other coarse fragments can substantially decrease the potential for erosion by providing greater roughness and tortuosity than would otherwise occur without the surfacing (see Chapter 3). However, even when surfacing materials are present on unpaved roads, rill and gully formation can result if sufficient water concentrates on the surface (Packer 1967). This tendency has led to the development of BMPs that focus on spacing drainage control features based on road gradient. Following these recommendations for the spacing of road drainage features is considered crucial to limiting overland flow and sediment delivery (Croke and Hairsine 2006).

The earliest study of and recommendations for cross-drain spacings in the United States were by Trimble and Weitzman (1953) on the Fernow Experimental Forest in the central Appalachian Mountains. Their recommendations were based on spacings that would maintain maximum allowable erosion of no more than 0.6 inch on skid roads after they had been closed (Table 2). These recommendations were calculated by using road gradient, road length (segment lengths between adjacent surface drainage features), and measured erosion data for skid roads located on benches overlaying limestone-derived soil following road closure. Forty-one profile stations were established on the skid roads, with 13 falling into slope gradients between 0 and 20 percent and road lengths from 0 to 132 ft. Twenty-three profile stations were between 21-and 40-percent grade with lengths from 0 to 132 ft. The remaining five were between 2- and 40-percent slope and 133 to 264 ft long. The authors used two approaches to develop recommendations; one was an "alignment table," where depth of erosion associated with pairs of slope gradients and road segment lengths

Table 2.—Recommendations for skid road water-bar spacing for the central Appalachian Mountains (from Trimble and Weitzman 1953)

Skid road grade	Distance between water bars	
percent	*feet*	*meters*
2	250	76
5	135	40
10	80	24
15	60	18
20	45	14
25	40	12
30	35	11
40	30	9

was identified and recommendations were based on those relationships. Because of limited data on the low and high ends of slope/distance pairs, they also used a modification of Manning's formula to calculate values and develop recommendations that included spacing recommendations for slope values beyond those used for the initial method. Although Trimble and Weitzman (1953) noted the importance of other factors (e.g., climate, soil organic matter, soil stoniness, and soil series-dependent characteristics) in affecting water bar spacing, the formula-derived spacings (Table 2) became the standard recommendation that these authors and others (e.g., Haussman and Pruett 1978, Weitzman 1952) presented in other general forest management publications.

Kidd (1963) also developed spacing recommendations for controlling erosion on skid roads (Table 3). His recommendations for western Idaho are based on amounts of erosion associated with cross drains derived from a combination of information in an unpublished 1954 report by Packer; spacings commonly used by the Boise National Forest at that time; and visual, qualitative scores of erosion from 569 skid trail segments on granitic and basaltic soils in Idaho surveyed for Kidd's (1963) study. Kidd found that erosion differed not only between the soils but also between ravine and hillside positions, so his recommendations were separated by those variables, as well as by skid road gradient.

Arnold (1957) developed maximum lateral drainage culvert spacing recommendations for Pacific Northwest truck roads based on soil erosion class, road grade, and rainfall intensities. Soils were assigned to 1 of 10 erosion classes based on the unified soil classification system used in engineering, and soil texture and parent material. Recommendations were made for road grades from 2 to 18 percent and rainfall intensities of 1 to 2 inches h^{-1}. Factors were included for the adjustment of the spacings for higher and lower rainfall intensities (e.g., spacings should be divided by 1.75 for rainfall intensities of 3 to 4 inches h^{-1}). However, no information was provided about how the spacing values were derived, and the recommended culvert spacings were much longer than those developed later. For a 10-percent road grade, spacings ranged from 180 to 845 ft, whereas spacings ranged from 120 to 565 ft on a road with a 15-percent grade.

Table 3.—Recommendations for skid trail control-structure spacing for soils and topographic locations found in west-central Idaho (from Kidd 1963)

	SKID TRAIL SPACING			
	Granite-derived soil		Basalt-derived soil	
Slope	Sidehill	Ravine	Sidehill	Ravine
percent	----------------------- *feet* -----------------------			
10	65	50	90	80
20	50	35	70	65
30	40	25	60	50
40	30	20	50	40
50	20	15	40	35
60	15	10	25	20
70	10	10	15	15

Packer and Christensen (1964) and Packer (1967) employed by far the largest quantitative dataset of those in the United States to develop cross-drain spacing guidelines for haul roads in the Rocky Mountain region. They gathered data from 720 study sites, with 120 sites in each of 6 major soil groups (Table 4). These were further stratified into 60 sites with southern aspects and 60 sites with northern aspects. Data from 20 sites in each of 3 slope positions (upper, middle, and lower one-third of slopes), and 5 study sites within each of those slope positions from each of 4 road gradient classes (0 to 3 percent, 4 to 7 percent, 8 to 11 percent, and 12 to 15 percent) on each aspect were used. They determined the distances that water could flow for various combinations of these variables between consecutive drainage structures before 1-inch-deep rills formed. The 1-inch-deep rill values were determined to be the depth beyond which the road surface would rapidly deteriorate (Packer 1967). From these strata of data, tables of cross-drain spacings were developed from regression equations for each topographic position, aspect, and upslope gradient of the hillside (Table 4). Roads on south-facing slopes are likely to erode more than on north-facing slopes due to earlier spring snowmelt, so they generally require closer cross-drain spacings (Packer 1967).

Table 4.—Cross-drain spacing recommendations for secondary logging roads in the upper topographic position[a] of north-facing slopes[b] having a gradient of 80 percent[c] (from Packer 1967)

Road grade	CROSS-DRAIN SPACING					
	Hard sediment	Basalt	Granite	Glacial silt	Andesite	Loess
percent	------------ *feet* ------------					
2	167	154	137	135	105	95
4	152	139	122	120	90	80
6	144	131	114	112	82	72
8	137	124	107	105	75	65
10	128	115	98	96	66	57
12	119	106	89	87	57	48
14	108	95	78	76	46	37

[a] On middle topographic position, reduce spacings 18 ft; on lower topographic position, reduce spacings 36 ft.

[b] On south aspects, reduce spacings 15 ft.

[c] For each 10-percent decrease in slope steepness below 80 percent, reduce spacings 5 ft.

Swift (1985) presented an equation for broad-based dip spacing on forest access roads:

Spacing in feet = 320 - (20 × road segment gradient in percent)

But this equation is based only on visual observations of functioning structures. Similarly, Rothwell (1978) stated that for Alberta, Canada, the distance in meters between water bars on skid roads should be calculated as:

350/ the road grade (presumably in percent)

There is no citation to identify what data were used to develop this method, however. In some instances spacings have been designed arbitrarily, and stated as such (e.g., Haupt and Kidd 1965).

As mentioned previously in this chapter, cross-drain spacings have been designed primarily to control erosion on the road surface (Croke and Mockler 2001, Edwards and Evans 2004). However, a few spacing studies have recognized the importance of considering the cross-drain spacing to control sediment delivery or erosion downslope of the cross-drain outfall. Croke and Mockler (2001) developed a table of cross-drain spacing recommendations specifically for Cuttagee Creek catchment, New South Wales, Australia, based on contributing road length and upslope contributing area. They surveyed road and road-to-stream linkage characteristics for 224 cross-drain structures. Discriminant function analyses were used to first determine the threshold value for contributing road length (distance between cross drains) that best predicted presence or absence of channels or gullies at drain outlets for a range of hillside gradients onto which runoff was cast. The threshold values for contributing length and upslope area were used to calculate recommended cross-drain spacings (for a range of road slopes and drain discharge hillslope gradients) which would minimize channel or gully initiation at drain outlets.

One of the most recent studies of cross-drain positioning was by Damian (2003). He focused on the location of cross drains that are nearest to stream crossings to reduce connectivity between roads and streams. Road segments that contain stream crossings have the greatest connectivity with streams, and thus, the greatest potential for sediment delivery, so their management is critical (Skaugset and Allen 1998). Conventionally, recommendations for those cross drains have been to position them close to crossings. This location was intended to minimize the amount of water and sediment delivered directly from the road and ditch line to the stream (i.e., the drainage and associated sediment originating between the nearest cross drain and the stream crossing). In some states, placement within 50 to 100 ft of crossings is a recommended practice (Washington Forest Practices Board Manual 2000). Damian (2003) used a modeling approach to optimize cross-drain location and found that the last cross drain in most cases should be positioned between 100 and 200 ft from the stream. Closer positions did not allow effective filtration of sediment because the buffer strip was too short. Consequently, there was a much greater net delivery of sediment to streams when the nearest cross drains were positioned close to streams compared to when they were moved farther away. For two watersheds in Washington state with 28 and 39 stream crossings, the reduction in sediment delivery by repositioning cross drains through modeling was calculated to be about 75 percent in both watersheds. However, no cross drains were actually relocated in the field, so no data were collected to measure and verify any reduction in sediment inputs.

The lack of independent data to verify cross-drain spacing recommendations more broadly within a region or across regions is a limitation of all of these studies. Studies which were designed as more rigorous experiments (e.g., Packer and Christensen 1964) may provide more robust data that are likely to be applicable elsewhere in the region compared to studies with little experimental design or those that were based primarily on observations or measurements from only a few road segments. However, no studies are available to verify the applicability of these studies within or across regions, even though many of these recommendations, or slight variations, are used fairly widely. For example, Trimble and Weitzman's (1953) recommendations became the foundation for many eastern states' skid road BMPs (Elliot et al. 2014) and some western states' (e.g., Colorado State Forest Service 2010), even though the East and the West have very different physiographic, soil, and climate conditions. Additionally, cross-drain spacings defined in each study were developed for a specific type of road (e.g., skid roads or haul roads), but the recommendations are also commonly applied to other types of road systems (Copstead et al. 1998).

The more intensive studies of cross-drain spacing (e.g., Packer 1967, Packer and Christensen 1964) illustrate that cross-drain spacing is vital to controlling erosion and that cross-drain spacing should decrease as road gradients increase. But the lack of studies specifically comparing erosion resulting from cross drains installed at a variety of replicated spacings in similar conditions and the lack of validation of the recommended spacings within regions, across regions, or across road types make it impossible to quantify the effectiveness of current recommended spacings. Nor is it possible to determine if spacings closer than those currently used are warranted, and in what conditions they should be implemented. Modeled data by Damian (2003) suggest that improvements on cross-drain spacing on forest roads are possible, at least in some locations, but these results also have not been field verified.

CHAPTER 3

Protecting the Driving Surface

This chapter includes information about the two types of BMPs most commonly applied to the roadbed: surfacing to control water-driven erosion and dust suppressants to control primarily wind- and vehicle-caused erosion.

The driving surface is inherently different from the rest of the road prism. It is purposely compacted during construction, and it is repeatedly disturbed by traffic. Consequently, it is highly susceptible to loss of soil and geologic materials due to vehicle use, especially during wet or extremely dry weather. So while surfacing and dust control techniques used to protect the road surface have value at all times, they are particularly aimed at these more extreme conditions.

Road Surfacing

Roads can be surfaced with a variety of materials, including asphalt or concrete pavement, quarried washed rock or stone, crushed rock, and gravel and sand materials (Beaty and Raymond 1995, Bolander and Yamada 1999, Dawson and Kolisoja 2006). Pavement is the most common type of road surfacing material (Succarieh 1992), but its use is typically restricted to moderate- and high-use roads due to its cost (Bolander and Yamada 1999, Sanders and Addo 1993). The primary reason that roads are paved is to provide a durable, smooth driving surface. Sediment control is obviously a benefit of paving because the surface is fully covered by a thick and relatively impermeable material, but few studies actually have quantified paving's effectiveness at controlling water-driven erosion. Reid and Dunne (1984) developed sediment rating curves and unit hydrographs for different types of surfaced road segments in Washington state and calculated that paved road segments yielded only 0.4 percent of the sediment that was generated from a heavily used gravel road. Clinton and Vose (2003) found much lower sediment concentrations in runoff from a paved asphalt road than from the dirt or gravel road segments to which it was compared. However, because of differences in frequency of maintenance and maintenance procedures applied to the three road treatments, quantifying the reductions in sediment specifically due to paving could not be done cleanly. Even with the general lack of data showing reductions in sediment losses due to paving, it is commonly recommended as a technique for reducing erosion and sedimentation (e.g., MacDonald et al. 1997).

Runoff from asphalt-paved roads has the potential to affect water quality in other ways. Petroleum hydrocarbons can be washed or leached from pavement and move into terrestrial or aquatic ecosystems. Hydrocarbons also can result from tire abrasion, exhaust, and lubricating oils from vehicles on the roads (Ngabe et al. 2000). In the aforementioned study by Clinton and Vose (2003), measurable levels of total petroleum hydrocarbons (TPH) were found in the runoff from the 2-yr-old paved road (<0.5 ppm), but the concentrations were well below those permitted in sediment. No federal water quality standards exist for TPH because these hydrocarbons include complex aliphatic and aromatic hydrocarbons that are not well understood chemically (Todd et al. 1999). Clinton and Vose (2003) did not attempt to determine the concentrations of the individual compounds in the TPH in the runoff. Their study is the only one found in the literature review that examined runoff chemistry from paved roads in predominantly forested watersheds. Hydrocarbon data were found for urban areas but were measured in samples collected from water bodies (e.g., Hunter et al. 1979, Latimer et al. 1990, Whipple and Hunter 1979) rather than from roadway drainage. Thus, those data would not be applicable for assessing petroleum hydrocarbon levels on paved roads in most forested watersheds or even in surface waters in forested watersheds (Sanger et al. 1999).

Although paving may be the preferred method to surface roads for long-term protection, many miles of rural roads exist for which paving cannot be justified. Nearly all of the road miles managed by the U.S. Forest Service are unpaved roads (Foltz and Truebe 1995), and 53 percent of all roads in the United States are unpaved (Koch and Ksaibati 2010). In these types of situations, other surfacing materials, such as quarried washed rock or stone and crushed rock or alluvial gravel, may be applied to provide strength and protection against soil displacement from the driving surface. Raindrop impact and consequent soil displacement are eliminated if the surface is completely covered (Quinton et al. 1997, Stuart and Edwards 2006). Additionally, pores between large surfacing aggregate allow drainage and movement of water from the road. Road runoff is slowed, and its energy for detachment and entrainment of particles is reduced by the tortuosity and friction resulting from water moving among the coarse fragments (Wisconsin Transportation Information Center [WTIC] 1997). However, large aggregates that do not have sufficient fines to help bind them together are susceptible to being kicked off the road by traffic or washed off the road, depending upon the aggregate and overland flow conditions present (Sanders and Addo 1993).

Smaller aggregates create a smoother driving surface and help to hold larger particles together (Bennett 1994, Bolander and Yamada 1999), but their presence retards internal drainage. In graded materials, fines should be present in mixtures that minimize water infiltration (Foltz and Truebe 1995) so that water runs off the surface as dispersed overland flow. If the percentage of fines is too small, water will be retained in the existing fines during wet weather due to capillary forces. The road then will perform poorly due to the development of ruts and susceptibility to surface failure because of freeze/thaw processes (Dawson and Kolisoja 2006). Conversely, during dry periods, excessive levels of fines are susceptible to being detached from the road surface by wheel contact (Bolander and Yamada 1999). As fines are lost, larger aggregates loosen and begin to abrade the surface materials as the aggregates move under traffic. This then leads to displacement of particles and road raveling (Sanders and Addo 1993).

There is little published research concerning the effectiveness of specific aggregate types (e.g., clean gravel, various graded mixtures) on controlling erosion on unpaved, low-volume roads. The lack of data about relationships between road surfacing and erosion is somewhat unexpected given the relatively easy nature of designing and implementing such studies. Based on the review of literature, the most commonly cited papers concerning the effectiveness of road surfacing involve three studies: in the Idaho Batholith (Burroughs et al. 1984a, 1984b, 1984c); in the central Appalachian Mountains of West Virginia (Kochenderfer et al. 1997, Kochenderfer and Helvey 1987); and in the southern Appalachians of North Carolina (Swift 1984a). Each of these studies showed that adequate surfacing results in substantial reductions of soil losses from graveled roads (Tables 5 and 6).

Using four simulated rain events, Burroughs et al. (1984a, 1984b) provide a straightforward illustration of the effectiveness of a single type of gravel with a single application thickness. A fresh application of 10-cm-deep hard crusher run (38-mm diameter and smaller) gneissic rock reduced erosion by 77 percent compared to a segment with no surfacing material. Scheetz and Bloser (2008) and Bloser and Scheetz (2012) reported similar results during simulated rain events on Pennsylvania roads. They compared erosion before and after application of Driving Surface Aggregate (DSA), a specific gradation of crushed stone surfacing material designed to establish a well-packed driving surface (Penn State Center for Dirt and Gravel Roads 2014). The DSA surfacing treatment reduced sediment losses on four road segments by 67 to 95 percent compared to the original native surfacing (Scheetz and Bloser 2008). In a subsequent study, Bloser and Scheetz (2012) found that a year after applying pit run aggregate (locally obtained material from "borrow pits" that is variable in composition and quality) to two existing native-surface roads, sediment losses were reduced by an average of 39 and 64 percent during 30-min simulated rain events. These sediment losses were similar to average losses on other roads surfaced with pit run aggregate that they measured in the region. Analogous treatments of DSA on two road segments reduced soil losses by an average of 67 and 95 percent during identical simulated rain events. DSA produced one-tenth of the sediment that the pit run segments produced, but sediment losses from the road segments slated for pit-run treatment were about three to four times as large as the DSA-treated segments prior to pit-run additions. In the southwestern Virginia Piedmont, Brown et al. (2013) monitored sediment delivery to streams through road approaches from newly reopened roads with bare soil surfaces and from legacy graveled roads, though no information was given about the gravel characteristics. During the year of study, sediment delivery from the five newly bladed bare-soil road segments averaged 98 Mg ha^{-1} yr^{-1} compared to 13 Mg ha^{-1} yr^{-1} from the four graveled road segments.

Kochenderfer et al. (1984) and Kochenderfer and Helvey (1987) compared the effectiveness of crusher run and clean gravel over a much longer timeframe with natural rain events and with vehicle traffic. Applications of 6-inch-deep layers of 3-inch-diameter crusher run and clean limestone gravel to replicated sections of a newly constructed haul road resulted in an average of 4.5 to 9 times less erosion than occurred on native soil, and the clean gravel was about twice as effective as the crusher run material in reducing soil losses (Table 5). Sediment yields from both aggregate surfaces were statistically less than from the dirt road sections. Because cutbank erosion contributed to the driving surface measurements in this study, the effectiveness of the road surfacing may be better than the data suggest, particularly for the crusher run treatment, which had a small cutbank slump in one road section that contributed to soil losses (Kochenderfer and Helvey 1987).

One reason that crusher run surfacing can result in more erosion than clean gravel is that the fines within the aggregate matrix contribute to loss of mineral material. Fines in crusher run material are susceptible to displacement and kick-off from the road surface by traffic, and water can displace and transport them fairly easily, particularly during intense rain events if the surface can be infiltrated (Swift 1984a). Thus, the loosened fines contribute to the overall losses (Bilby et al. 1989, Kochenderfer and Helvey 1987, Swift 1984a, Toman and Skaugset 2011, Ziegler et al. 2001). Toman and Skaugset (2011) found that aggregate with the largest percentage of fines less than 0.6-mm diameter produced the highest amount of sediment of any of the surfacing treatments they examined. Sediment loss was higher than that from two other sites with the same treatments, but percentages of fines less than 0.6-mm diameter were lower.

The quality of surfacing material also can influence sediment production. Aggregate that is weak and friable under loads or that cannot resist breakdown in the presence of water makes poor-quality surfacing material

Table 5.—Sediment yield by surfacing treatment

Treatment	Average or total sediment yield	Reference
Native untreated soil	54.5 kg 100 m^{-2}	Burroughs et al. (1984a), Burroughs et al. (1984b)
Rolled native surface	67.1 kg 100 m^{-2}	
Rolled gravel surface with 10 cm of hard crushed gneissic rock	11.7 kg 100 m^{-2}	
Unsurfaced, bare soil	44 ton ac^{-1} * 47.2 ton ac^{-1} **	Kochenderfer et al. (1984), Kochenderfer and Helvey (1987)
3-inch crusher run on new road, 6-inch uncompacted depth	10.1 ton ac^{-1} **	
3-inch clean gravel on new road, 6-inch uncompacted depth	5 ton ac^{-1} * 5.7 ton ac^{-1} **	
1-inch crusher run on ≥50-yr-old road, depth not specified	5 ton ac^{-1} * 5.9 ton ac^{-1} **	Kochenderfer and Helvey (1987)

* Mean over first 2 yr of study.
** Mean over 4 yr of measurement.

(Rodgers et al. 2014). The percentage of fines increases and the percentage of coarser particles decreases as a result of breakdown, thereby creating an aggregate mixture that becomes more prone to road raveling, rutting, and overall failure. Foltz and Truebe (1995) compared a good-quality (resistant to breakdown) and marginal-quality (subject to breakdown) aggregate applied to existing roads and subjected to log truck traffic over 2 yr in Oregon. The marginal-quality aggregate broke down, and 3.7 and 17.3 times as much sediment came off that road section during years 1 and 2, respectively, compared to the road section with the good-quality aggregate. Most of the increase in sediment was due to breakdown of the smallest fines in the clay fraction.

Swift (1984a) further confirmed the reductions in soil loss resulting from using clean gravel compared to crusher run on unpaved roads, but his experimental design also provided insight into how other variables influence the effectiveness of road surfacing. His study was overlain on a commercial logging job, so the experimental design was somewhat complicated. It involved four surfacing thicknesses (5, 15, 20, and 0 cm/bare soil), two types of gravel with different maximum diameters (38-mm-diameter crusher run and 7.5-cm-diameter clean stone, both hard gneiss), two roadbed soil textures (sandy loam and sandy clay loam), and periods with a variety of road disturbance intensities (Table 6). Treatments involving sandy loam roadbeds were monitored during road construction; those on the sandy clay loam were added just after road construction was completed. The bare soil segment was planted to grass immediately after logging ended.

The presence of surfacing was critical during road construction (Swift 1984a). Total soil losses during road construction from both the 15- and 20-cm-thick treatments on sandy loam soil were about 78 percent less than from bare soil (Table 6). Soil loss during construction from the 5-cm-thick crusher run gravel on the sandy loam soil was about half that of the other two surfacing treatments, but the former included only the last month of the construction period compared to the entire construction period for the latter two treatments. However, erosion rates during the subsequent light traffic period suggest that 5 cm of the crusher run material would have been insufficient had it been in place during

Table 6.—Sediment yield from forest roads with different surfacing treatments and soil types (derived from Swift 1984a)

Treatment	TIME PERIOD				
	Road construction May–June 1976	Light truck traffic July–December 1976	Active logging January–June 1977	Light truck traffic July 1977–June 1978	Light truck traffic July 1978–April 1979
	tonne ha^{-1}				
Bare soil control on sandy loam[a]	141[b]	61	198	---	---
Bare soil control planted to grass[a]	---	---	---	134	40
15-cm-deep crusher run on sandy loam	30[b]	3	18	20	Not measured
20-cm-deep clean stone on sandy loam	31[b]	0.5	12	3	Not measured
5-cm-deep crusher run on sandy loam	16[c]	58	146	258	52
15-cm-deep crusher run on sandy clay loam	---	20	146	187	75
5-cm-deep crusher run on sandy clay loam	---	33	212	161	97

[a] The road segment used as the control between May 1976 and June 1977 was planted to grass in July 1977.
[b] Loss during entire road construction period.
[c] Loss during last ~1 month of road construction.

the entire construction period. Five centimeters of crusher run gravel yielded more soil loss with only light traffic than thicker applications of crusher run or clean gravel, and rutting in both of the 5-cm-thick sections began under light traffic.

In contrast, there were substantial reductions in the amounts of sediment that originated from the control, the 15-cm-thick, and the 20-cm-thick surface treatments on the sandy loam during the first period of light truck traffic. Soil losses from the control dropped by almost 60 percent, whereas soil losses from the gravel treatments were only about 0.1 and 0.02 percent of what they had been during construction—even though the period with light truck traffic was three times as long (Table 6). The influence of higher percentages of clay on erosion is obvious from comparisons of the 15-cm crusher run treatments on the sandy clay loam and sandy loam roadbeds throughout the study. Similarly, the influence of a lower clay percentage on erosion control under heavy traffic with large loads is apparent by comparing the erosion losses of the two 15-cm crusher run treatments during logging; soil yields were an order of magnitude lower for the sandy loam than the sandy clay loam. Recovery to much lower erosion levels also was notably quicker for the sandy loam roadbed. Once grass became well established, it controlled erosion even better than the thinly graveled sandy loam section and both sandy clay loam sections.

Soil texture has been found to influence soil loss in other road studies. Sugden and Woods (2007) found relatively low road-erosion rates (5.4 Mg ha^{-1} yr^{-1}) from unpaved roadbeds composed of gravelly sandy loams and gravelly silt loams. Roadbeds with sandier soils in Colorado had 2 to 3.5 times less soil loss than roadbeds with higher silt content in Idaho, even though the sandier roadbeds had more loose soil (Foltz and Burroughs 1990). More energy is required to entrain and move coarser particles in overland flow compared to finer particles (Sugden and Woods 2007), so even if particle detachment occurs, soil loss from the road may be less than the detachment rate.

All of the studies cited in Tables 5 and 6 reported that rut formation increased erosion from the road tread, and this finding is common to many other studies as well. Sediment losses in the range of 100 to 500 percent greater have been reported from rutted road segments compared to unrutted segments (Burroughs et al. 1984a, 1984b; Foltz 1995; Foltz and Burroughs 1990). The amount of increase depends upon specific road and site factors, but it is proportional to the severity of rutting (Burroughs and King 1989, Foltz 1995). The percentage increase also tends to be higher on rutted unsurfaced (i.e., dirt) roads than on rutted roads surfaced with aggregate (Kennedy 1997). Because rut formation increases erosion, it follows that actions taken to control ruts can result in concomitant reductions of sediment production. Reductions in the range of 50 to 75 percent were achieved when rut production was controlled, with the degree of reduction depending on the depth of the rut and site conditions (Foltz 1995, Kennedy 1997). If ruts form, simply filling the ruts with clean or crushed rock probably will not have a substantial effect on controlling soil losses, as Swift (1984a) found. Instead, regrading the road so that the road is reworked down to at least the bottom of the deepest ruts is needed to reduce aggregate losses and road breakdown (Bolander 1997, Bolander and Yamada 1999, Roads and Transportation Association of Canada [RTAC] 1987, Skorseth and Selim 2000).

Four factors lead to rut formation (Dawson and Kolisoja 2006): 1) compaction of unsaturated materials in the surfacing aggregate; 2) local shear on the aggregate road surface, which pushes aggregate located below the wheel downward and then upward to the outside of the wheel tracks; 3) shear deformation of the subgrade soil, in which the mechanisms described for factor 2 occur in the subgrade, and the surface materials sink; and 4) local shear on the aggregate road surface from contact with wheels, which damages and displaces particles. Toman and Skaugset (2011) found that ruts developed only within the surfacing aggregate on three newly built roads in California and Oregon. However, it is usually a combination of all four factors that causes ruts to form (Dawson and Kolisoja 2006), so rutting control requires that all of these causes be addressed in roadbed preparation and surfacing/resurfacing considerations.

In most cases, the greatest gains in erosion control from surfacing come from newly built or newly disturbed (e.g., widened) roads, such as those involved in the studies just described. However, applying aggregate to existing roads can provide some protection against road erosion (Kochenderfer and Helvey 1987). Kahklen and Hartsog (1999) found that resurfacing existing roads generally reduced sediment yields by an order of magnitude at three sites in southeast Alaska. Improvements on existing roads tend to be less because, as most studies show, erosion rates decrease substantially as roads age (e.g., Beschta 1978, Megahan and Kidd 1972, Sullivan 1985). The exceptions to this generalization may be roads that are severely damaged and unmaintained, or roads that have been recently maintained. MacDonald et al. (2001) found no difference in erosion between older and newer roads on the island of St. John in the U.S. Virgin Islands, but the older roads were deeply rutted and had not been regraded for years. Appelboom et al. (2002) compared three freshly graded road sections with different surface treatments—preexisting gravel, 10 cm of new gravel over geotextile, and bare soil—in the lower Coastal Plain of North Carolina. Sediment production from the new and older graveled sections was not significantly different at 49.2 and 69.1 kg km^{-1}, respectively, whereas both graveled sections yielded less than half the sediment of the bare soil section (151.1 kg km^{-1}). For 109 native surface road segments monitored over 3 yr in the Sierra Nevada Mountains of California, Coe (2006) found that recently graded segments produced about twice as much sediment as ungraded segments.

Dust Palliatives

Dust is fine particulate matter (SynTech Products 2011), and it is an important issue for road managers. Fine particulates emanating from roads are more than a nuisance to drivers and those who live near roads; these particles also can affect human health, vegetative growth, water quality, and aquatic habitats (Forman and Alexander 1998, Sanders and Addo 1993, Succarieh 1992, WTIC 1997). Of particular concern are particulates that are less than 10-µm diameter. These are referred to as PM_{10} and are regulated by the U.S. Environmental Protection Agency (EPA) (Jones 2000a), primarily because of their influence on human health. Although both paved and unpaved roads contribute to particulates of this size (Claiborn et al. 1995, Jones 2000a, Norman and Johansson 2006, U.S. EPA 1996), unpaved roads are the largest sources of PM_{10}-sized particles in the United States (U.S. EPA 1996). Dust levels from unpaved roads are directly related to the percentage of fines (particles <75 µm) present in the road surfaces (Sanders et al. 1997, U.S. EPA 1995). Consequently, application of dust control agents on unpaved roads is important in airsheds that do not meet the PM_{10} standards established by the U.S. EPA (Watson et al. 1989). Normally, paved roads receive dust abatement treatments only when an area is out of attainment of the PM_{10} standards (Claiborn et al. 1995, Norman and Johansson 2006).

Dusts generated in open areas are termed “fugitive dusts” (U.S. EPA 1995). Fugitive dusts from roads come from a variety of sources, including wear of vehicle parts (e.g., brakes, clutches, tires), vehicle exhaust, abrasion of the road surface and aggregate pullout due to traffic, and deposition from the atmosphere (U.S. EPA 1995, Watson 1996). But in general, most originate from fine particles in the bed of dirt roads or from the matrix of the surfacing materials (Jones 2000a, WTIC 1997). These become loosened primarily as the result of slippage of tires on the road surface (Succarieh 1992), and they then can be transported from the road surface by water or wind erosion (WTIC 1997). With the increasing loss of fines, the road surface becomes more dominated by larger particles that are susceptible to being kicked off the road by subsequent traffic or eroded by water. The loss of small and large particles leads to the formation of ruts, potholes, and corrugations (washboarding), which further contribute to erosion potential (Jones 2000a).

Aggregate applications, such as crushed rock or gravel, can be used to suppress dust on roads (Alaska Department of Environmental Conservation [DEC] 2010); however, proper composition of the aggregate is needed because some aggregate mixtures can contribute to dust generation. The proper amount of fines (including dust-sized fines) and the composition across all particle size classes in the aggregate matrix are important. A proper mixture ensures that voids are filled and held together by sufficient fines so the surfacing is strong enough to resist raveling, washboarding, and rutting by traffic (Bennett 1994, Bolander and Yamada 1999). Conversely, too many fines will create a surface mixture that cannot resist displacement by tires (American Society of Civil Engineers 1992), and too few fines will not effectively bind together particles in the road matrix (Bolander and Yamada 1999, Jones 2000a, WTIC 1997). However, it is often difficult to obtain desired mixtures or to ensure that aggregate mixtures meet specifications before their application (Foltz and Truebe 1995, Lunsford and Mahoney 2001). In these situations another method may be needed to address dust control.

Other techniques include limiting road use and traffic speed, paving, applying dust control agents, or implementing some combination of these techniques (U.S. EPA 1995). Road use and traffic speed controls are discussed in Chapter 4. Paving with asphalt or concrete is the most common and most efficient dust control technique. Paving can suppress dust generation by as much as 90 percent (Succarieh 1992, Watson 1996). But paving is usually restricted to higher-volume roads because its high cost usually can be justified only where there is greater potential for excessive dust creation (Bolander and Yamada 1999). Dust palliatives typically are used on roads with moderately low to moderate traffic volume, such as 150 to 500 vehicle passes per day, and they are most beneficial and most cost effective on these types of roads. On low-volume roads, the cost of dust palliatives typically cannot be justified and dust generation may not be much of a concern due to infrequent traffic (Kirchner and Gall 1991, WTIC 1997).

Dust palliatives are chemicals that work in one of three ways: agglomerating fine particles (i.e., causing them to ball up), adhering or binding surface particles together, or increasing the density of the road surface (Bergeson and Brocka 1996, Bolander and Yamada 1999). Once a road is treated, particles resist being suspended in the air by traffic or the wind (Bolander and Yamada 1999).

Dust palliatives should not be confused with soil conditioners (see Chapter 6); they are designed to do two different things. Soil conditioners focus on controlling water-driven erosion by binding clay particles at the surface of soils that do not receive repeated disturbances. Soil conditioners are ineffective on surfaces that are mechanically disturbed after treatment (e.g., Orts et al. 2007); therefore, they do not abate dust on roads. Additionally, with respect to forest road prisms, soil conditioners are applied primarily only as a short-term stopgap measure until vegetation becomes established to control erosion. In contrast, dust generation is expected in the long term on unpaved roads unless road use is suspended or drastically reduced. As such, palliative retreatment is required through the long term.

Road surface characteristics influence the success of dust palliatives. In general, they are much less effective on roads that have more than 30-percent fines in the surface aggregate material or in the soil surface (in the case of dirt roads). Excessive amounts of fines (>30 percent) tend to overwhelm the ability of the dust palliative to control dust. Conversely, there is limited benefit in applying dust control to roads with less than 5-percent fines because these roads generate little dust (WTIC 1997). Dust suppression also is generally less effective for sandy soils that have little plasticity. These soils tend to lack sufficient fines (i.e., clays) to allow binding to occur, so the palliative may simply leach through the road materials (Kirchner and Gall 1991). However, these limitations are only generalizations, and some agents are more effective than others for either extreme of fines or low soil plasticity. Consequently, to maximize effectiveness and reduce the need for maintenance and retreatment, selection of a specific type of dust palliative should be based on the local road characteristics, weather, available equipment, and expected needs (Bolander and Yamada 1999, Jones 2000a, Langdon and Williamson 1983, Succarieh 1992). In addition, for dust palliatives to be effective, the road surface must be prepared properly before the dust control agent is applied. For unpaved roads this typically involves scarification and regrading the road surface, smoothing the road and ensuring proper crowning and surface drainage, and creating optimal compaction (Bolander 1997, Jacobson 1992, Sanders et al. 1997).

Dust palliatives have been in use since at least the 1940s (Gebhart et al. 1996). They exist in many forms and formulations and can be assigned to a variety of different classifications depending upon the user (e.g., Jones 2000a, Succarieh 1992, WTIC 1997). Bolander and Yamada (1999) suggested seven broad categories: water, water-absorbing (hygroscopic and deliquescent chemicals), organic petroleum, organic nonpetroleum, electrochemical, synthetic polymers, and clay additives (Table 7); they also provide tabular summaries of the attributes, limitations, application, origin, and environmental impacts of each type of road dust suppressant. An abbreviated summary of that information is given in Table 8.

Water is the simplest, but the most short-lived, dust control agent (Addo and Sanders 1995). It controls

Table 7.—Categories of dust suppressants with examples (after Bolander and Yamada 1999)

Dust suppressant category	Examples
Water	Water
Water-absorbing	Calcium chloride Magnesium chloride Sodium chloride
Organic petroleum	Asphalt emulsions Cutback asphalt Dust oils Bitumens
Organic nonpetroleum	Animal fats Lignosulfonates Molasses/sugar beet Tall oil emulsions Vegetable oils
Electrochemical	Enzymes Ionic products Sulfonated oils
Synthetic polymers	Polyvinyl acetate Vinyl acrylic
Clay additives	Bentonite

Table 8.—Common dust suppressants, treatment rates[a], limitations, application methods, and longevity of effectiveness (abbreviated summary, developed from information in Bolander and Yamada 1999)

Dust suppressant	Treatment rates	Limitations	Application methods	Longevity
Water	Frequent, low rates	Very short duration	Spray	≤1 day
Calcium chloride	Flakes: 0.9 kg m^{-2} Liquid: 1.6 L m^{-2}	Corrosive to vehicles, potential pollutant	Mix solids into surface, or spray brine on surface	6 months
Magnesium chloride	2.3 L m^{-2}	Corrosive to vehicles, potential pollutant	Mix solids into surface, or spray brine on surface	6 months
Organic petroleum products	0.5–4.5 L m^{-2}	Rutting in weak bases, could be toxic	Mix into or spray on surface	6 months
Lignosulfonates	2.3–4.5 L m^{-2}	Potential pollution from leaching	Mix into or spray on surface	6 months
Vegetable oils	1.1–2.3 L m^{-2}	Limited availability, becomes brittle	Mix into or spray on surface	1 yr
Tall oils	2.3 L m^{-2}	Highly soluble	Mix into or spray on surface	≥1 yr
Electrochemical derivatives	Diluted 1/100 or 1/600	Depends on clay mineralogy	Mix into surface with light compaction	Unknown
Synthetic polymers	2.3 L m^{-2}	Not studied well, difficult to maintain as hard surface	Mix into or spray on surface	≥1 yr
Clay additives	1–3% by weight	Rutting in wet conditions	Mix uniformly into surface	1–5 yr

[a] Treatment rates often depend upon characteristics of residual concentrate.

dust by flocculating particles through capillary cohesion (Hoover 1987). Repeated applications, sometimes as frequent as every few hours, are necessary for continued effectiveness (Bolander and Yamada 1999). Therefore, its use typically is limited to applications where dust suppression needs are very short term (Jones 2000a). Seawater is preferred to fresh water, if available, because the salts are water absorbing, and thus provide better and longer-duration dust control (Addo and Sanders 1995; also see the description of chloride salt palliatives that follows). Water should be applied in small amounts frequently to avoid overwetting the road surface, as overwetting can promote rutting (Bolander and Yamada 1999, Foley et al. 1996, Koch and Ksaibati 2010) and pullout and carryout of sediment on tires of vehicles traveling on the road (Watson 1996), especially on dirt roads with no surfacing.

Water-absorbing chemicals used for dust control are primarily calcium chloride and magnesium chloride (Bennett 1994). Calcium chloride is derived from the manufacture of soda ash, and magnesium chloride is obtained from seawater (Heffner 1997). Both of these salts are hygroscopic (sometimes erroneously referred to as hydroscopic, e.g., as described in RTAC 1987, Singer et al. 1982) and deliquescent, though calcium chloride is more so. Hygroscopic chemicals can absorb water from the surrounding environment, including the atmosphere, to help maintain a moist film around the soil particles and bind them together. Deliquescent chemicals can absorb so much moisture from the surrounding atmosphere that they dissolve and convert to a solution, so the generated moisture contributes to the soil binding properties and improves soil compaction (Bennett 1994, Kirchner and Gall 1991, Succarieh 1992, WTIC 1997). The ability to absorb and retain moisture also retards evaporation from the road surface, thereby prolonging the presence of moisture in the surface matrix (Kirchner and Gall 1991, Morgan et al. 2005, Sanders and Addo 1993, WTIC 1997). A crust is created on the road surface by the salts' absorption of water, which contributes to retaining fines on the road surface (WTIC 1997). Although sodium chloride can be used as a dust palliative, it is much less deliquescent and less hygroscopic than either calcium chloride or magnesium chloride (Addo and Sanders 1995).

The greater hygroscopic and deliquescent characteristics of calcium chloride allow it to be applied within a broader range of conditions than magnesium chloride. Calcium chloride can enter into solution at very low humidities (as low as 18 percent) in high temperatures (Addo and Sanders 1995, Kirchner 1988), so it works in dry conditions when dust formation is most likely (Sanders and Addo 1993). In contrast, magnesium chloride works as a dust suppressant only above 70 ºF and greater than 32-percent relative humidity (Kirchner and Gall 1991, WTIC 1997). Outside those values it does not remain in solution, so its ability to control dust varies through time depending upon the ambient conditions (Kirchner and Gall 1991). When conditions are suitable for magnesium chloride, it is sometimes preferred because it creates a harder crust that is less prone to dust formation; however, it requires about 18 percent more salt to be used compared to calcium chloride (Kirchner and Gall 1991, WTIC 1997).

Both of these salts are hygroscopic under reasonably low humidities, but they are generally less effective at controlling dust than many organic petroleum and nonpetroleum products in those conditions. For example, in the Las Vegas Valley under desert conditions, disturbed soils treated with magnesium chloride as a dust suppressant resulted in more soil loss than no dust treatment (Loreto et al. 2002). Consequently, salts tend to be recommended for use in more humid climates (Langdon 1980). In addition, hygroscopic dust control agents should not be used on roads with more than about 25-percent clay in the surface because the water retained by the palliative will be passed to the clay and the road will remain wet, potentially compromising stability (Kirchner and Gall 1991). Chloride salts work best on roads of which 10 to 20 percent of the surface materials are composed of particles less than 75-µm diameter (Bolander 1997).

Salts can be sprayed onto the road surface as brine, or they can be mixed into the road surface during construction or maintenance as dry granules (Addo and Sanders 1995, Bennett 1994). Salts have little dust control effectiveness if applied during or immediately before rain events because they are water soluble and will leach or wash from the road (Federation of Canadian Municipalities and National Research Council 2005). They also are ineffective in wet climates because of losses due to leaching (Addo and Sanders 1995, Bennett 1994, Federation of Canadian Municipalities and National Research Council 2005). On the other hand, some road moisture is needed for them to maximize their hygroscopic character, so wetting a dry road before application can improve effectiveness (Kirchner and Gall 1991). In dry conditions, salts that have penetrated into the road over time can be rejuvenated to some degree with light applications of water. The water allows salts to migrate back to the surface by capillary forces; however, at the surface, they again become susceptible to washoff (Slesser 1943, Succarieh 1992). Because effectiveness is lost due to washoff and leaching, reapplication of chloride salts two to three times per year typically is necessary to control dust continuously (Bennett 1994, Bolander and Yamada 1999, Monlux 1993). With no retreatment, salts provide no dust control within a year or less from the time of initial application, depending upon local conditions (Morgan et al. 2005).

Organic petroleum products are used commonly for dust suppression (Succarieh 1992). These chemicals include tars from coal distillation, bitumens from crude oil distillation, cutback asphalt (solvents such as gasoline, napthal, or kerosene combined with asphalt cement), and asphalt emulsions (asphalt dispersed in water + an emulsifier) (Addo and Sanders 1995). They act by binding or agglomerating road surface particles (Bolander and Yamada 1999). Petroleum products exist in a variety of viscosities, so they are suitable for a wide range of road surface characteristics. Higher viscosity petroleum agents are more effective on surfaces dominated by larger particles, and lower viscosity agents are more effective with higher amounts of fines (Giummarra et al. 1997). However, petroleum products are relatively ineffective when large percentages of clay exist in the road surface because the clay absorbs the petroleum rather than allowing it to act as a binding agent, so more binder must be added (Bennett 1994, Hoover et al. 1981). Asphalt recycled from roadways and roof shingles also is becoming a more common road surface treatment. It is typically used in highway resurfacing, but can be used as aggregate or as stabilizing material in the road base, sub-base, or embankment fills, or applied in thin coats, such as with chip seal (Koch and Ksaibati 2010).

Waste oils are organic petroleum products that once were used commonly for dust suppressants in the United States. However, because they were environmentally damaging and were contaminated with toxic materials (Addo and Sanders 1995, U.S. EPA 1990, Yanders et al. 1989), their use for dust control has been banned in this country (Foley et al. 1996, U.S. EPA 1991). This ban is not problematic because waste oils provide inadequate dust control due to poor aggregate binding capabilities (Addo and Sanders 1995), and other available organic petroleum dust suppressants provide better adhesive properties, are insoluble and resistant to washoff by water, and have fewer environmental concerns attached to their use (Addo and Sanders 1995).

Aside from pavement, asphalt emulsions are probably the most commonly applied petroleum-based dust control agents. They tend to remain effective longer than many products if applied under appropriate conditions, though they still require occasional retreatment to retain low dust emissions (Table 8). Repeated use of asphalt emulsion products can harden the road surface and create a surface similar to thicker recycled asphalt, making it particularly effective at dust control (Bennett 1994). However, recycled asphalt is prone to rutting in areas where turning, accelerating, or decelerating is concentrated (WTIC 1997), so asphalt emulsion treatments may have the same problem. Once ruts develop, erosion of subgrade materials can occur, but the emulsion-hardened surface makes regrading and the removal of ruts difficult (Monlux 1993, WTIC 1997). Tars and bitumens have been used for dust palliation (Addo and Sanders 1995, Jacobson 1992, Jones 2000a), but there is relatively little research on their effectiveness on aggregate surfaced roads (Jones 1999, 2000a). In general, bitumens require reapplication at least once or twice a year (Bennett 1994, Bolander and Yamada 1999), though the thickness and type of application affect their life span. Spray applications tend to be effective for only short periods (e.g., weeks to months), whereas thick applications mixed with sand and applied to the surface have been reported to be effective for up to 3 yr (Jones 1999, 2000a; WTIC 1997). Often tars and bitumens are used to temporarily control dust and stabilize roads until road upgrades can be made (Jones 1999).

Many types of organic nonpetroleum binders exist, but almost all of those that are used for dust suppression are lignosulfonates (Succarieh 1992), which are alternatively termed “lignin sulfonates” (WTIC 1997). Lignosulfonates are residues resulting from the sulfite wood pulping digestion process (Heffner 1997, WTIC 1997). Consequently, their chemical composition depends upon the tree species from which they were derived and the chemicals added during the pulping process (Bolander 1999, Jones 2000a, Morgan et al. 2005, Succarieh 1992).

The lignin and sugars that are present naturally in lignosulfonates (Addo and Sanders 1995, Ledingham and Adams 1942, Pandila 1973) adhesively bind soil particles together, which prevents fines from forming and being released into the air by traffic (Jones 2000a). The sugar also makes them hygroscopic (Bolander and Yamada 1999), thereby giving them some of the dust control advantages of hygroscopic salts. Other advantages of treating unpaved roads with lignosulfonates are that they remain slightly plastic, which permits maintenance (e.g., regrading and reshaping) of the treated road, and the surface aggregate can become more compacted (Morgan et al. 2005).

Like many other palliatives that work by binding soil particles, lignosulfonates can be used effectively on roads with low levels of surface fines (e.g., 4 to 8 percent) (WTIC 1997), though they are most effective when between 8 and 20 percent of the road surface materials can pass through a 75-µm sieve (Bolander 1997). Lignosulfonates can be applied in liquid or solid form, but because of their physical composition, they do not penetrate well and are ineffective if not mixed into the road surface at the time of application (Kirchner and Gall 1991, WTIC 1997). When mixed into the top 2.5 to 5 cm of the road surface, they provide better penetration and dust control than at other depths (Bolander 1997, WTIC 1997). Lignosulfonates are biodegradable and water soluble over time (Adams 1988, Bolander and Yamada 1999, Lunsford and Mahoney 2001, Succarieh 1992), so at least annual applications are needed to retain effective dust suppression (Bennett 1994, Bolander and Yamada 1999) (Table 8). Lignosulfonates are particularly susceptible to being washed off during heavy rain events, so they are better suited to application in drier conditions (Giummarra et al. 1997; Jones 1999, 2000a). Although they remain a relatively common dust palliative, lignosulfonates are used less today than in the past due to changes in waste stream management by pulp producers (WTIC 1997).

The other types of nonpetroleum chemicals (Table 8) bind or agglomerate particles (Bolander and Yamada 1999), but they tend to be less effective at dust control and less available than lignosulfonates (Batista et al. 2002, Federation of Canadian Municipalities and National Research Council 2005). Animal fats and vegetable oils, of which soybean oil soapstock (or simply soapstock) is most common, agglomerate particles; molasses residues, sugar beet extract, and tall oils bind particles (Batista et al. 2002, Lohnes and Coree 2002). Soapstock is a fatty acid by-product from the refining of edible oils, so its availability is subject to soybean market conditions (Morgan et al. 2005). Some vegetable oils do not work well as palliatives, both during and following application, due to their physical characteristics at ambient temperatures (Morgan et al. 2005). After application they become brittle, which diminishes their effectiveness (Federation of Canadian Municipalities and National Research Council 2005). PM_{10}-sized dust losses are strongly associated with roads treated with dust suppressants that create brittle surfaces (Gillies et al. 1999). Because organic nonpetroleum compounds are derived from animals or plants, they also have associated unappealing odors for up to several weeks (Morgan et al. 2005). Molasses residues are water soluble, and substantial amounts will dissolve and wash off roads during heavy rains, meaning multiple applications may be needed annually (Addo and Sanders 1995). No specific recommendations on application intervals are available.

Electrochemical dust palliatives are prepared from sulfonated petroleum or highly ionic chemicals, such as enzymes or ammonium chloride. Climatic conditions have little influence on their effectiveness, and they are not easily lost by leaching (Federation of Canadian Municipalities and National Research Council 2005, Giummarra et al. 1997, Lunsford and Mahoney 2001). However, enzymes must be protected from freezing conditions during transport and application, and sufficient moisture must be present at the time of application (Lunsford and Mahoney 2001). Electrochemical materials interact with the clay fraction of soils (Batista et al. 2002, Lunsford and Mahoney 2001), so they are most effective when at least moderate percentages of clay are present (Giummarra et al. 1997, Lunsford and Mahoney 2001) although there apparently is substantial variability in their effectiveness (Federation of Canadian Municipalities and National Research Council 2005). Many of the most common electrochemical enzymes are cultured bacteria; when exposed to air, they begin to reproduce and create large molecules that are absorbed into the soil clay lattice and form a tight compacted surface within days or hours. This process prohibits water absorption (Lunsford and Mahoney 2001) and increases compaction, thereby reducing dust generation (Jones 1999, Lunsford and Mahoney 2001). Electrochemical palliatives generally are mixed into the road surface during application (Bolander and Yamada 1999). There has been little widespread testing of electrochemicals, so almost no information is available on their duration as dust suppressants (Lohnes and Coree 2002) (Table 8).

Synthetic polymers are monomers that have been polymerized in an aqueous medium (Jones 2000a). They are usually by-products of paint and adhesive manufacturing (Bolander 1999, Lunsford and Mahoney 2001), so they suppress dust by binding particles together and forming a thin semi-rigid tackifier coating on the road surface (Alaska DEC 2010). They are currently some of the most expensive dust palliatives available (Lunsford and Mahoney 2001). When used for dust control, these polymers are applied in liquid form and require a drying or curing time at well above freezing temperatures for 12 to 24 h. During summer temperature and moisture conditions, they can increase the tensile strength of roads, which presumably would mean greater dust control. Tensile strength increases are as much as 10 times that of clay additives (described next), depending upon the product. Freeze-thaw cycles reduce the tensile strength of synthetic polymers, but it still remains about two to three times greater than that of clay additives (Bolander 1999).

Clay additives are made of bentonite, or impure montmorillonite clays (Bolander 1999). The most common bentonite clays are sodium and calcium montmorillonite. Sodium montmorillonite is a swelling clay in the presence of moisture, which allows it to seal and become waterproof (Murray 2000). Consequently, sodium montmorillonite is the clay used for dust suppression. Clays control dust through agglomeration (Bergeson and Brocka 1996). Because they have a net negative charge, bentonite clays are best suited for use as dust suppressants on unpaved roads that are surfaced with limestone or other material that has a net positive charge. This allows the formation of agglomerates through electrostatic bonding (Wahbeh 1990). Bentonite also is most effective in conditions where low levels of fines are present in the surface materials because the added clay particles provide fines needed to bind surface aggregates (Bennett 1994, Lohnes and Coree 2002). Relatively small levels of clay are applied to avoid retaining too much moisture in the road surface (Bergeson and Brocka 1996, Wahbeh 1990). Recommendations include 1- to 3-percent clay by weight of aggregate (Bennett 1994, Bolander 1997), with the total materials passing through a 75-µm sieve at ≤15 percent (Bolander 1997). Because of their affinity for moisture, clays are best suited to drier climates (Lohnes and Coree 2002). If the treated road surface remains stable and unrutted, clay may have the longest effectiveness of all palliatives; Bolander and Yamada (1999) estimated that the longevity of clay additives is 1 to 5 yr.

Although dust suppressants have been used for decades, there are remarkably few quantitative studies of their effectiveness (Sanders and Addo 1993, Sanders et al. 1997). Far more studies involving dust suppressant chemicals focus on road strength and stability (e.g., Bolander 1999, Butzke 1974, Monlux 2003). Although the relationship between road strength/stability and dust control is acknowledged (e.g., see Bader 1997 and Foley et al. 1996), papers that did not specifically describe dust reduction results are not included in this synthesis because they did not focus on the control of nonpoint source pollution. Also, many unpublished reports of dust control effectiveness are based on interviews with state highway agency personnel or contracted grader operators rather than on measurements. These observations may have merit, but these reports also are excluded from this review of effectiveness due to the lack of supporting data.

A major limitation of dust suppressant research is the lack of standard ways to measure dust emissions (Addo and Sanders 1995). Many techniques have been used to measure dust, but the equipment and protocols usually have been developed for the local research purposes (e.g., Addo and Sanders 1995, Hoover et al. 1973, Irwin et al. 1986, Jones 1999, Koch and Ksaibati 2010, Morgan et al. 2005, Schultz 1993, Taylor et al. 1987, Wellman and Barraclough 1972). Some methods measure dust at stationary points along the road with passive or active collection devices; others use a variety of devices that measure dust behind a moving vehicle traveling at one or several fixed speeds (Sanders and Addo 1993). Consequently, individual studies can show dust reduction from palliative use and relative effectiveness among agents, but interstudy quantitative comparisons typically cannot be made (Addo and Sanders 1995).

Another limitation of dust abatement studies is that they do not quantify the contribution of dust control to the reduction of soil losses from roads. Dust control agents commonly have been reported to reduce dust by 30 to 80 percent compared to no dust control (WTIC 1997), but a given reduction in dust does not correspond to an equivalent reduction in soil material losses from the road during the same period. This is because about two-thirds of fugitive dusts are estimated to return to the unpaved road surface after settling out of the air (Jones 2000a). Much of the remaining dust is believed to settle out within 20 m of the road unless wind carries it farther (Forman and Alexander 1998, WTIC 1997). But dust has been shown to affect corridors of more than 140 ft, which were up to six times the right-of-way width (Hoover et al. 1981). Therefore, the lack of specific information on the transport and fate of dust particulates (Forman and Alexander 1998) from various types of dust suppressants makes it virtually impossible to estimate the soil losses that accompany these reductions without field measurements.

Of all the studies reviewed for this synthesis, only one directly measured the effectiveness of dust palliatives on reducing erosion losses from roads (Burroughs et al. 1984a). They applied dust oil and a bitumen treatment to road segments in Idaho and applied artificial rain events to the untrafficked road segments. They compared soil losses from them to an unsurfaced road segment made of native granitic soil material. All road segments were newly reshaped and graded before treatment, and ranged in slope from 5.3- to 10.3-percent grade. Both the dust oil and bitumen treatments had substantially lower soil losses than the bare soil, but the bitumen was much more effective (Table 9). Compared to the unsurfaced road, sediment losses from the bitumen were 28.7 times lower on an area basis, whereas the losses from the dust oil were only 3.2 times less. The bituminous treatment also was superior to a 10-cm-thick application of 3.8-cm crusher run gravel in terms of controlling soil losses from the road (Table 9). However, this study involved only four simulated rain events closely spaced in time and the longevity of bitumen effectiveness is only about 6 months (Table 8), so in the longer term, erosion protection would diminish without repeated treatments.

In terms of dust reduction alone, petroleum-based products (excluding asphalt paving) sometimes have been found to be inferior to many other products, especially salts. Kirchner (1988) measured dust with air samplers for five road segments treated with different dust suppressants and one control. The suppressants were an asphalt emulsion, a natural brine, a semi-processed brine, liquid calcium chloride, and liquid magnesium chloride. A truck was driven on the road 50 times at 45 mi h^{-1} (mph). At the end of the first summer season, the section treated with asphalt emulsion performed almost as poorly as the control section even though it received two application treatments. A thin

Table 9.—Sediment yield from road surfacing treatments in the Silver Creek experimental watershed, Idaho (data from Burroughs et al. 1984a)

Treatment	Average sediment yield	Sediment loss compared to untreated soil	Sediment reduction from treatment
	kg 100 m^{-2}	*percent*	
Native untreated soil	54.5		
Native soil with dust oil	17.2	31.6	3.2 times less
Native soil with bituminous surface treatment	1.9	3.5	28.7 times less

crust had formed but became broken up, allowing the road surface to show through. The natural brine also did not work well because it did not retain moisture effectively. Consequently, it rutted and substantial dust was generated. The semi-processed brine exhibited slightly better dust control, but it also rutted and developed potholes. In contrast, the calcium chloride- and magnesium chloride-treated sections performed much better, though slightly more dust was generated in the sections treated with magnesium chloride than in those treated with the calcium salt. They generated 48 to 65 percent less dust than the control during the summer season. The road segments treated with these two salts also resisted rutting and pothole development. At the end of the summer, a second treatment of each road section except the asphalt emulsion did little to improve the effectiveness of the brines, but the magnesium and calcium chloride treatments retained high levels of dust control through the next 9 months (i.e., a full year from the time of first treatment). The effectiveness of both salts was within a few percentage points at the end of the year compared to the effectiveness observed soon after retreatment.

A year-long study in Colorado that compared asphalt emulsion to calcium chloride found similar results (Kirchner and Gall 1991). At the end of just 4 wk, calcium chloride resulted in average dust control of 72.6 percent compared to only 31.1 percent for the asphalt emulsion. The road segment treated with emulsion also developed many potholes, and a substantial amount of road surface aggregate was lost and accumulated along the side of the road. A common petroleum emulsion, Coherex® PM (Tricor Refining, LLC, Bakersfield, CA), was not considered effective at controlling PM_{10} dust emissions from unpaved roads in California for a 3-month period (Gillies et al. 1999), though its average efficiency over nine measurement periods was 67 percent. However, about 1 yr after the application, the average efficiency dropped to 44 percent.

In other studies, petroleum-based products performed well as dust suppressants. Hoover et al. (1975) found that two different asphalt emulsions performed well and had significantly lower dust emissions after a year compared to untreated controls. Conversely, at the end of that year, calcium chloride had become ineffective. In light of the inconsistent performance of petroleum-based dust suppressants, Langdon (1980) suggested that petroleum products would be more effectively applied toward meeting the nation's energy needs than used for dust control.

In a visual rating (qualitative) study reported by Monlux (1993) on 3 mi of road in Montana, a concentrated emulsified oil, an emulsified asphalt, a polymer emulsion, an emulsion copolymer, a light mineral oil, a lignosulfonate, and magnesium chloride were examined for dust control and compared to a control section over 7 wk. Emulsified asphalt was outperformed by magnesium chloride, but both controlled dust better than all other treatments. Potholes and corrugations developed on all road segments due to insufficient crowning and inadequate drainage. Regrading was performed to improve the road surface, but the emulsified asphalt surface was the most difficult to reshape because of the crust that had formed. The magnesium chloride and emulsified asphalt continued to provide dust control benefits following maintenance even without further treatment. Magnesium chloride was an effective dust suppressant for 12 wk, the emulsified asphalt for 8 wk, and all other treatments for 5 wk or less.

Although lignosulfonates were not particularly effective in the study by Monlux (1993), they have been at least as effective as other types of treatments in many other situations. For example, Fox (1972) found that treatments of ammonium lignosulfonate, and ammonium lignosulfonate in combination with calcitic lime or aluminum sulfate as additives, each were effective and reduced dust by 80 percent on granular surfaced secondary roads. A 1 percent ammonium lignosulfonate treatment with no additives was as effective as a 1 percent ammonium lignosulfonate treatment + 0.5 percent of either of the additives. Measurements 1 yr after treatment in Iowa showed that lignosulfonate and lignosulfonate + herbicide each significantly lowered dust emissions compared to untreated roads (Hoover et al. 1975).

Lignosulfonate appears to be particularly effective for conditions with moderately high traffic levels and more effective than many treatments with poor-quality aggregate surfacing. In Colorado, 2-km-long graveled road segments that had been treated with lignosulfonate, magnesium chloride, calcium chloride, and a formulation of calcium chloride containing no magnesium all had lower dust emissions than an untreated segment (Fig. 4), but lignosulfonate was most effective (Addo and Sanders 1995, Sanders and Addo 1993, Sanders et al. 1997). This occurred even though vehicle use on the treated sections was twice that of the untreated control section (400 versus 200 vehicle passes day^{-1}). This study was continued a second year following regraveling and retreatment of the road sections (Fig. 4). The calcium chloride treatment with no magnesium was not used as a treatment the second year, but instead that road segment was used as the untreated section (Addo and Sanders 1995, Sanders et al. 1997). Dust levels were more similar among the three palliatives than they had been in the first test period, and again all three were more effective than no treatment (Fig. 4). The lignosulfonate treatment usually had the lowest dust emissions, and aggregate losses from public road use calculated from cross-sectional measurements were least from the section treated with lignosulfonate (Fig. 4). Losses equated to depths of 5.8, 5.2, and 7.0 mm for the lignosulfonate, magnesium chloride, and calcium chloride treatments, respectively, and were about one-half to one-third the 15.6-mm depth of aggregate lost from the untreated graveled road segment. Unexpectedly, even though the temperatures were high and relative humidity was generally low throughout the study, magnesium chloride resulted in lower dust generation and aggregate pullout than the more deliquescent calcium chloride.

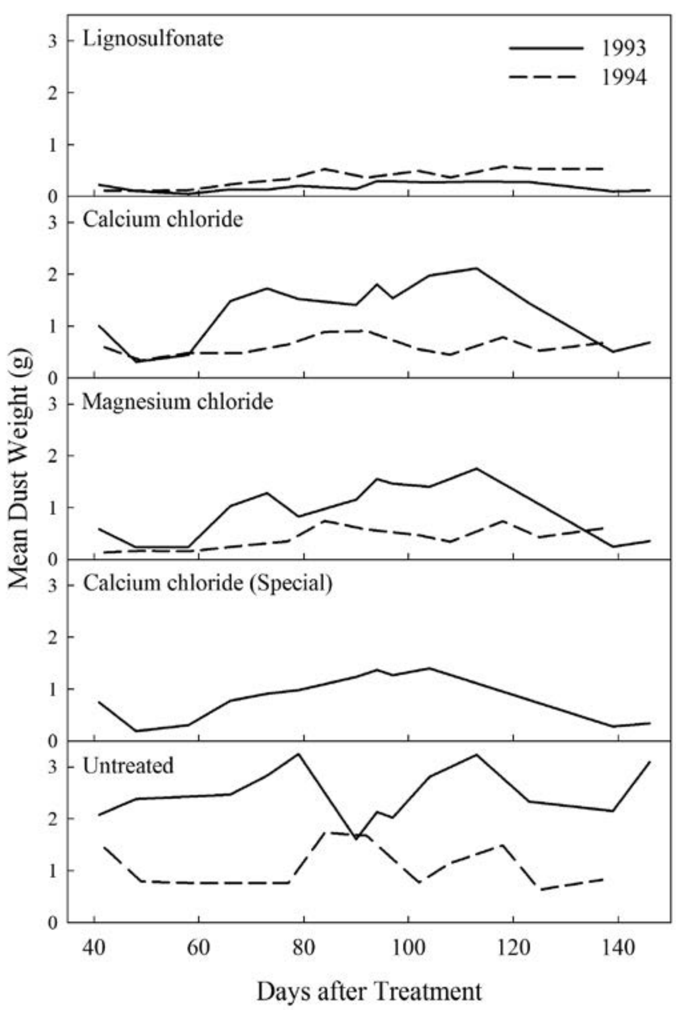

Figure 4.—Mean dust weight by dust suppressant treatment and year (graphs developed from data in Addo and Sanders 1995, and Sanders and Addo 1993). Calcium chloride (Special) includes magnesium chloride.

Morgan et al. (2005) measured dust generation in Iowa from road segments treated at the start of the study and again at 8 wk with lignosulfonate, calcium chloride, and a soybean oil soapstock. Their study also allowed investigation of the influence of road aggregate quality (crushed limestone rock and alluvial sand/gravel mixture), traffic volume (low = 45 to 60 vehicles day^{-1} and high = 240 vehicles day^{-1}) and time (16 wk and 52 wk). During the first 16 wk, average dust reduction was similar, about 50 percent, for both road surfaces compared to similarly surfaced roads with no palliation (Table 9). Traffic volume made the influence of road surface quality much more apparent over that short timeframe, with much poorer effectiveness on the gravel/sand surface compared to the crushed limestone. Only the lignosulfonate was effective on the alluvial surfacing, and it reduced dust by only one-fourth to one-third during the first 16 wk. Suppressant effectiveness declined through time, and by the end of 1 yr, average dust reductions of the palliatives that remained effective were between 10 and 30 percent (Table 10). In the long term, traffic volume was deemed to be a more important factor for controlling dust than surfacing, and lignosulfonate was the only palliative that consistently contained dust at levels below that of the untreated road segments.

Gebhart et al. (1996) also studied soybean oil soapstock effectiveness on unsurfaced roads and tank trails at the U.S. Army's Construction Engineering Research Laboratories in Fort Hood, TX, and Fort Sill, OK. It was compared to a 38 percent calcium chloride mixture, calcium lignosulfonate, and a polyvinyl acrylic polymer

Table 10.—Average dust reduction by surface, traffic volume, and treatment over two time periods for Story County, Iowa (data from Morgan et al. 2005)

Surface	Traffic volume	Dust suppressant	Dust reduction compared to no treatment[a] Average over 16 wk	Average over 52 wk
			----------- *percent* -----------	
Alluvial sand/gravel	Low	Lignosulfonate	56	28.2
		Calcium chloride	40	17.9
		Soapstock	50	27.8
	High	Lignosulfonate	27	Not tested
		Calcium chloride	-1[b]	-26.0
		Soapstock	-2	-27.4
Crushed limestone rock	Low	Lignosulfonate	76	22.0
		Calcium chloride	51	11.6
		Soapstock	51	6.9
	High	Lignosulfonate	61	10.5
		Calcium chloride	46	-24.0
		Soapstock	24	14.6

[a] Percentage by weight of aggregate.
[b] Negative values indicate an increase in dust.

emulsion (PVA; the latter results are described in greater detail on page 37). At Fort Hood after 30 days, all four treatments resulted in at least 50-percent dust reduction compared to a control. The calcium chloride salt was most effective at abating dust, but it was not significantly better than the soapstock or other treatments. By day 60, the soapstock continued to have lower dust emissions than the control, and was not significantly different from the lignosulfonate or PVA. But the soapstock was significantly less effective than the chloride salt. At that time the calcium chloride reduced dust levels by at least 50 percent; the soapstock and lignosulfonate did not retain that level of protection.

Only a few studies have measured dust abatement from bentonite. Bergeson and Brocka (1996) used sodium montmorillonite as a dust palliative on limestone-surfaced secondary roads in Texas. Applications tested ranged from 0.5 to 9 percent by weight of aggregate. Dust generation was reduced by about 45 percent on sites where 3-percent bentonite was employed and 70 percent on sites with 9-percent bentonite. Wahbeh (1990) also examined the effectiveness of different application rates of sodium montmorillonite using two application techniques on limestone-surfaced roads in two counties in Iowa. The treatments were a spray slurry application of bentonite + water + soda ash to loose-surface material, and a dry application of bentonite mechanically mixed with the crushed limestone surface aggregate and then saturated with water and soda ash. Bentonite application rates were 0.5, 1.0, 1.5, 2.0, 2.5, and 3.0 percent in Adair County; only the first three application rates were used on Dallas County roads. The bentonite treatments also were compared to spray applications of calcium chloride or magnesium chloride. Dallas County roads received 75 to 80 vehicle passes day^{-1} over the 2-yr study, though the dust measurements were determined only from 10 test passes by a ½-ton truck traveling at 40 to 45 mph in or out of the wheel tracks. Wheel track performance was tested on 21 days and out-of-the-wheel track performance was tested on 16 days throughout the 2 yr. In Dallas County, the maximum bentonite concentration (1.5 percent) and both chloride treatments resulted in similar dust control originating from the wheel track: about a 20-percent reduction compared to no palliative treatment of the limestone aggregate (Table 11). Dust was best controlled out of the wheel tracks by the 1.5-percent bentonite treatment, where a 27-percent reduction in dust was obtained.

Bentonite effectiveness endured in Dallas County even with a reapplication of limestone aggregate to the road surfaces about halfway through the study. The short-term effectiveness of calcium chloride is apparent in the 3-month-long study on Adair County roads (Table 11). The best dust reduction both in and out of the wheel tracks occurred with the calcium chloride treatments and the highest percentage of bentonite clay (3.0 percent).

Little information could be found on the effectiveness of other dust control agents. Only a small discussion about the effectiveness of a sugar beet molasses, Molex, was available (U.S. Roads 1998). Even though its effectiveness was praised, only general, qualitative comments were given. There are few field applications with measures of dust control effectiveness from electrochemical palliatives or comparisons of their effectiveness to more traditional suppressants in the literature. This may be because many electrochemical palliatives are relatively new to the market (Jones 1999, 2000a; Lunsford and Mahoney 2001). A product called EMC SQUARED® (Soil Stabilization Products Co. Inc., Merced, CA) is the most commonly tested electrochemical palliative. Gillies et al. (1999) found that it was only marginally effective (33-percent dust reduction) during a period of 1 wk, and then its effectiveness declined quickly. McHattie (1994) reported that a single application of EMC SQUARED was as effective as montmorillonite clay and calcium chloride about 1 yr after application on a 2.5-mi length of a highway reconstruction project in Alaska, but an additional treatment was required after 20 months. The degrees of effectiveness of EMC SQUARED may vary because of the need for very specific moisture content during application; many papers also have noted that electrochemical dust suppressants may work only with certain types of soils (Bolander 1999, Piechota et al. 2002, Scholen 1992).

Table 11.—Average percent dust reductions with bentonite clay and chloride salts for Dallas and Adair Counties, Iowa (data from Wahbeh 1990)

Location	Treatment	Dust reduction: In wheel track	Dust reduction: Out of wheel track
		---------- *percent* ----------	
Dallas County[a] (21 test passes)	0.5% bentonite	2	6
	1.0% bentonite	6	15
	1.5% bentonite	20	27
	Calcium chloride	20	6
	Magnesium chloride	22	16
Adair County[b] (11 test passes)	0.5% bentonite	8	9
	1.0% bentonite	11	11
	1.5% bentonite	17	12
	2.0% bentonite	5	14
	2.5% bentonite	10	19
	3.0% bentonite	42	33
	Calcium chloride	30	48

[a]Data are from October 1987–October 1989.

[b]Data are from August 1989–November 1989.

Synthetic polymers traditionally have been used primarily as soil conditioners, and there is little research on their ability to suppress dust on traveled roadways (Jones 2000a, Lunsford and Mahoney 2001). The information that does exist is mixed and typically provided as general statements with no data; for example, Jones (1999, 2000a) indicated that studies from South Africa showed little effectiveness for synthetic polymers due to poor penetration into the road surface. Scholen (1992) also reported poor dust control by an acrylic polymer in Florida. Bolander (1997) stated that polymers provide some promise from tests in the Pacific Northwest. Two studies at the U.S. Army's Construction Engineering Research Laboratories showed limited success of PVA (Gebhart et al. 1996). At Fort Sill, OK, PVA was tested over a 60-day period on unsurfaced roadways and tank trails. Thirty days after application, the PVA had three to four times lower dust emissions than untreated controls. At day 60, dust levels still were less than from the control road segment, but dust emissions had doubled from those present at day 30. The heavy track and wheel equipment traveling the roads broke up the PVA surface seal. Similar results were obtained from an identical study at Fort Hood, TX, though the levels of dust abatement at 60 days were even less than those measured at Fort Sill. Gillies et al. (1999)

found a polymer emulsion to be highly effective over 12 months, even with reasonably heavy use (6,400 vehicle passes) and wet winter conditions. They found the product, Soil-Sement® (Midwest Industrial Supply, Inc., Canton, OH), reduced PM_{10}-sized dust by an average of more than 80 percent throughout the entire study.

Environmental Effects of Dust Palliatives

The studies just summarized illustrate that most of these various chemicals have the ability to abate dust on unsurfaced roads, and thus, provide some control of soil and aggregate losses, at least temporarily. However, because of their chemical composition, some dust suppressant chemicals can themselves be nonpoint source pollutants.

There are two general types of studies that have been performed to examine the impacts of dust palliatives: 1) those that have worked toward establishing toxic thresholds for animals, humans, or plants, and 2) those that have quantified the chemistry of runoff and leachate or have examined the environmental effects of the chemicals or runoff/leachate on flora and fauna following field application or accidental spills. The first type of study dominates the literature, and despite the relatively common and widespread use of many types of dust suppressants, few studies have quantitatively examined their environmental impacts (Sanders et al. 1997). Toxicology studies and studies with extremely high loads or concentrations of palliatives indicate that most mammals and humans show little effect from most palliatives. Instead, the primary environmental concerns about their use at concentrations normally encountered tend to be their potential effects on groundwater quality, freshwater aquatic organisms, and to some degree plants (for some specific chemicals) (Bolander and Yamada 1999).

The effects of lignosulfonates have been studied far more than any other dust suppressant. In part, this may be because there are many studies showing the toxic effects of spent sulfite liquors on aquatic organisms, including trout (*Salvelinus* and *Oncorhynchus* spp.) (Fisher 1939, Griffin and West 1976, Walden 1976). Because spent sulfite liquors are precursors to lignosulfonates and a common waste product of the pulping industry, there also has been concern about potential effects of lignosulfonates. However, lignosulfonates are created by evaporating water from spent sulfite liquors, and during distillation, sulfur dioxide, and acetic and formic acids also are removed to yield about 50- to 60-percent solids. Thus, the resulting calcium, sodium, or ammonium lignosulfonates are less toxic than the original wastes (Adams 1988). Lignosulfonate toxicity also has undergone extensive study because the Food and Drug Administration allows the same formulations used for dust palliation (ammonium, calcium, magnesium, and sodium lignosulfonates) to be used as binders in pelletized animal feeds and in paper and paperboard products used to package liquid and fatty foods (Singer et al. 1982; Watt and Marcus 1974, 1976).

Overall, lignosulfonates seem to pose little toxicity to mammals and humans at concentrations or loads that normally would be encountered with dust suppressant use (Adams 1988, Bolander and Yamada 1999, Succarieh 1992). Studies of direct fluid ingestion of calcium, magnesium, or sodium lignosulfonates at concentrations of 40 g L^{-1} by guinea pigs (*Cavia porcellus*) over 8 wk found average lower weight gain by animals receiving sodium and calcium lignosulfonate (117 g and 144 g, respectively) compared to animals receiving no lignosulfonate (253 g). The guinea pigs treated with each of these chemicals also developed ulcerated colons. In contrast, the guinea pigs receiving magnesium lignosulfonate had much less difference in average weight (210 g) than the control animals and no development of ulcerated colons (Watt and Marcus 1976). These results confirmed earlier findings of lower weight gain and development of ulcerated colons when only sodium lignosulfonate was administered at several concentrations to guinea pigs (Watt and Marcus 1974). The data to support nontoxicity of lignosulfonates to humans primarily comes from workers handling lignosulfonates. No chronic health problems have been reported over the 40 yr they have been commonly used—including from people who have worked over the long term with these chemicals (Adams 1988).

Lignosulfonates are most harmful to aquatic species (Bolander and Yamada 1999, Schwendeman 1981). The sensitivity to lignosulfonates is attributed to the effects that these chemicals have on the biological oxygen demand (BOD) and through direct toxicity (Adams 1988, Poole et al. 1978). Lignosulfonates may contain as much as 35 percent wood sugars, and these are fermented relatively easily by microorganisms, which increases BOD and reduces available oxygen in the short term (Adams 1988). The remaining fraction of the chemicals that are low molecular weight are degraded much more slowly, primarily by fungi (Engen et al. 1976, Pandila 1973, Singer et al. 1982, Stapanian and Shea 1986, Watkins 1970), allowing long-term elevations in BOD (Raabe 1968). Because microbial breakdown of lignin in lignosulfonates is incomplete (Ledingham and Adams 1942, Watkins 1970), the remaining compounds of higher molecular weight undergo decomposition via desulfonation, demethoxylation, and depolymerization, similar to the processes that occur for natural breakdown of lignin in wood (Engen et al. 1976).

Even in trout, which are considered to be the fish species most sensitive to lignosulfonates (Succarieh 1992), concentrations at which toxicity occurs are fairly high. In a study of sodium lignosulfonate effects on rainbow trout (*Oncorhynchus mykiss*), the 48-h LC50 was 7,300 ppm, and 50-percent survival occurred for 25 h even at a concentration of 2,500 ppm (Roald 1977a). The LC50 is the concentration of a chemical in water or air required to kill 50 percent of the test population during the observation period. Survival was 100 percent over 28 days for concentrations up to 1,875 ppm. In a follow-up study, rainbow trout were subjected to sublethal concentrations of 0 (control), 40, 80, 160, and 320 ppm for 60 days and concentrations of 0, 640, 1,280, and 1,920 ppm for 35 days (Roald 1977b). Fish exposed to concentrations of 160 ppm or greater had slower growth rates than those with no exposure. All rainbow trout subjected to the 1,920-ppm concentrations died during the 35-day exposure. The lignosulfonates did not affect the bacterial flora in the intestines, but the activities of some digestive enzymes were lower than the controls at concentrations as low as 320 ppm. It was not clear whether the lowered growth from reduced food consumption was due to decreases in enzyme activities, or if enzyme activities declined because of lower food consumption.

Lignosulfonates also are considered to be of little risk to plants, though this assumption is based on few studies. Stapanian and Shea (1986) applied calcium lignosulfonate with 50-percent solids at rates of 12, 42, and 63 ton of solids ac^{-1} (Adams 1988) directly to the ground of two clearcuts planted to Douglas-fir (*Pseudotsuga menziesii*) 1 or 3 yr before treatment. These rates were much higher than rates typically applied to roads for stabilization or for dust control (5 and 1.3 ton of solids ac^{-1}, respectively) (Adams 1988). Neither the growth of the Douglas-fir nor the aboveground woody biomass was affected through 12 wk of monitoring. Applications of 42 and 63 ton ac^{-1} significantly reduced herbaceous biomass, though the mechanism driving the decrease was unknown. Lignosulfonate migration through the soil occurred at approximately the same rates, regardless of the amount initially applied, but over the 12 wk it disappeared from the top 0.2 m of soil because of its high water solubility (Stapanian and Shea 1986). Yardley et al. (1980) found that lignosulfonate application to soil did not prevent seed germination.

The greatest risks related to use of lignosulfonates are associated with contamination of water. Lignosulfonates are water soluble, especially when associated with soil or road surface matrix pH values of 6 or greater. Consequently, lignosulfonates can leach through the soil, especially as rainfall increases (Singer et al. 1982). However, they also have low penetrability in soil because clays can adsorb lignosulfonates and retard leaching. Mobility can be retarded even in regions with high rainfall if the percent clay is sufficiently high, but even clay cannot stop transport of lignosulfonates by flowing water (e.g., concentrated overland flow) (Singer et al. 1982). Schwendeman (1981) suggested that 70 to 100 percent of road surface materials should pass through a ¾-inch sieve and 20 to 50 percent should be composed of silt or clay for optimal lignosulfonate retention and dust abatement.

Whether there are harmful or even measurable levels of lignosulfonates in soil leachate, streamwater, or groundwater will depend upon the concentrations applied (Stapanian and Shea 1986), road surface and soil texture, precipitation characteristics, and general road condition. At the levels applied as dust suppressants and when applied according to manufacturer's instructions, threats to these parts of the environment are believed to be minimal (Bennett 1994, Morgan et al. 2005, Singer et al. 1982, WTIC 1997). Any effects that do occur would be expected to be in the immediate area around the application site (Singer et al. 1982).

Only one study was found in which runoff from a lignosulfonate applied to abate dust on disturbed land (not a road) was measured and analyzed for concentrations of toxic materials, including volatile and semivolatile organic compounds, organic pesticides, metals, BOD, and a variety of other chemicals (Loreto et al. 2002). The lignosulfonate used, Dustac® (Quattro Solutions, Welshpool, Western Australia), had the fewest contaminants present in runoff of all the types of palliatives compared (Table 12). Volatile organic compound concentrations were less than 25 µg L^{-1}, and semivolatile organics and pesticides were not detected. Some of the highest concentrations of copper, chromium, and manganese were found in runoff from this plot, but none of these metal concentrations was more than 34 ppb. BOD was only about 1 mg L^{-1}, but chemical oxygen demand (COD) was approximately 350 mg L^{-1}. In this study, some of the chemicals measured may have been from the palliative itself, from chemical reactions (e.g., exchange) resulting from palliative chemicals, or from chemicals in the soil that would have been present in runoff regardless of whether the palliative was used.

Consequently, many of the statements describing the limited concern about lignosulfonates in the environment are based upon current understanding of their biogeochemical behavior rather than on data.

The environmental effects of petroleum-based dust suppressants are largely unknown. These may be the most difficult products about which to make generalizations because they exist in many different formulations, and new products are being developed regularly. To further complicate matters, very little field monitoring of contamination by these products using typical application rates has been accomplished. Therefore, much of the research applied to this subject extends from environmental emergencies (e.g., spills), or from toxicology studies that use concentrations exceeding the small amounts of petroleum in dust palliatives (Succarieh 1992).

Ettinger (1987) examined the effects of an accidental spill of commonly applied petroleum emulsion dust suppressant (Coherex) on aquatic organisms. An

Table 12.—Constituents with the highest concentrations, and other negative characteristics[a], observed in runoff from dust suppressant-treated plots (data from Loreto et al. 2002)

DUST SUPPRESSANT TYPE AND PRODUCT NAME						
Acrylic polymer EK35®	**Lignosulfonate Dustac®**	**Petroleum-based ------------ Coherex® ------------**		**Nonpetroleum-based Road Oyl®**	**Mulch Plas-Bond[b]**	**Salt ($MgCl_2$) Dust Gard®**
2-butanone	Nitrate	Acetone	Thallium	Acetone	Hardness	Chloride
Acetone	COD	Benzoic acid	Lead	Ammonia-nitrogen	Conductivity	pH >8
Nitrate	TOC	Ammonia-nitrogen	Arsenic	Sulfide	pH >8	Cyanide
Cyanide	Chromium	Sulfate	Selenium	pH >8	TDS	Turbidity
Turbidity	Barium	Sulfide	Copper	BOD	TSS	TS, TSS
TS, TDS, TSS	Silver	Conductivity	Manganese	COD	Sulfate	Thallium
Arsenic	Conductivity	Coliform	Nickel	TOC	Nickel	Lead
Benzoic acid	Copper	Alkalinity	Zinc	Aluminum		Arsenic
Pentachlorophenol		Cyanide	Cadmium	Iron		Boron
Hardness		BOD	Barium	Zinc		Zinc
BOD		Hardness	TVS	TSS		Silver
COD		TDS	Iron	Nickel		Sulfate
TOC		TSS		Cadmium		
TVS		COD				
Alkalinity						

[a] BOD = biological oxygen demand, COD = chemical oxygen demand, TOC = total organic carbon, TS = total solids, TDS = total dissolved solids, TSS = total suspended solids, TVS = total volatile solids.

[b] Manufactured by Soil Solutions Co.

unknown amount of the material was spilled into a ditch that led directly into a tributary of the Schuylkill River in southeastern Pennsylvania. The resin in the mixture adheres to soil and quickly attached to the stream bed. Concentrations of Coherex in tributary streambed sediments were as high as 3,550 mg kg $^{-1}$ during the first 3 days. Within the first 8 hours of the spill, benthic macroinvertebrate density in the affected reach was significantly less than in unaffected reaches, and there were 6.5 times more dead benthic macroinvertebrates as live ones. By day 3, most macroinvertebrates and many of the fish, both adult and larval forms, in the area had died. The dead fish were primarily blacknose dace (*Rhinichthys atratulus*) and white suckers (*Catostomus commersonii*). Crayfish (*Cambarus* spp.) were not as severely affected, but they were very sluggish. Thirty two-lined salamander larvae (*Eurycea bislineata*) and one adult American toad (*Bufo americanus*) also were found dead. Coherex is biodegradable, and by day 10 the concentrations declined substantially and the affected reach had become repopulated with macroinvertebrates at numbers similar to unaffected reaches. There was little change in the streambed concentrations of the product from day 10 through day 59; the concentration at day 59 was 385 mg kg $^{-1}$.

Coherex was included in the runoff study of dust palliatives applied to disturbed soils in the Las Vegas Valley by Loreto et al. (2002). By a substantial amount, Coherex had the largest number of contaminants with the highest concentrations in the collected runoff (Table 12). Most notably, the concentration of acetone (a toxic volatile organic compound) was 66.2 µg L $^{-1}$, and the concentration of benzoic acid (a toxic semivolatile compound) was 225 µg L $^{-1}$. Coherex also had high concentrations of copper, chromium, and manganese.

Hoffman and Eastin (1981) examined the toxicity of a dust oil called RDCO on mallard duck (*Anas platyrhynchos*) embryos. Eggs exposed to 0.5 µl of oil on day 3 after being laid had 60-percent mortality by day 18. Exposure to the same concentration 8 days after being laid reduced mortality by about half (32 percent) by day 18. All of the ducks that survived this dose to be hatched had bill, brain, or eye defects, or incomplete ossification of the skeleton. A more dilute (2 percent) aqueous emulsion spray of RDCO applied at day 3 or day 8 resulted in lower mortalities by day 18: 13 percent and 17 percent, respectively. Ducks exposed at day 3 showed no abnormalities; 12 percent of those exposed at day 8 had abnormalities.

Kimball (1997) evaluated the potential for pollution of groundwater by PennzSuppress® D (PennzSuppress Corp., Lago Vista, TX), a petroleum-based dust suppressant and road stabilizer. In place of a road study, he examined the chemistry of laboratory leachate obtained from limestone road-base material treated with the product. The leaching results were employed in a mathematical modeling exercise of fate and transport of those materials. He concluded that there was a low risk for this specific product to negatively affect groundwater quality, but warned against extrapolating these results to other geologic (soil and bedrock) situations or other products.

A cursory review of the literature suggests that chloride-based salt palliatives have undergone the most extensive studies related to their environmental effects. However, deeper examination of these studies indicates that these evaluations typically rely on information obtained from deicing studies using chloride salts (e.g., Ettinger 1987, Piechota et al. 2002, Singer et al. 1982). There have been very few studies in which off-road salt levels or environmental effects from dust control by chloride salts have been measured. Although the physics and chemistry related to environmental impacts for both types of applications are the same for like salts, the specific environmental effects that can be expected may be very different due to the different objectives, timing and frequency, and application techniques used for deicing versus dust abatement. For example, deicers may be applied to road surfaces repeatedly during and immediately after snow and ice events, resulting in high chloride concentrations in snowmelt runoff. In contrast, effective use of chloride salt dust suppressants requires application during dry periods with the compound either mixed into or sprayed on the road surface. Applications are made as needed, but generally occur one to two times per season (Bolander and Yamada 1999).

Chloride and magnesium salts are highly soluble, so they can move through the environment relatively easily with moisture (Addo and Sanders 1995, Federation of Canadian Municipalities and National Research Council 2005). Their deliquescent behavior and capillarity provide short-term retention (e.g., over months), but eventually they will fully succumb to leaching or washoff (Singer et al. 1982, Slesser 1943, Succarieh 1992).

The potential environmental effects of calcium chloride and magnesium chloride come from the chloride anion and the base cation, calcium or magnesium. The chloride anions dissociate from the calcium or magnesium ions. Because soil has a net negative charge due to the presence of clay particles, the chloride ions are repelled by soil and largely will remain in solution. In contrast, the positive charges of calcium and magnesium ions allow them to be retained by soil clays via cation exchange (Bohn et al. 1985). Initially, they will be exchanged for other base cations held less tightly by the soil, such as sodium and potassium. Because anions pair with cations to retain electroneutrality during leaching, the cations that have been released from the soil during cation exchange will pair with the chloride ions (Christ et al. 1999, Reuss and Johnson 1986). Thus, sodium chloride and potassium chloride will be leached first. As inputs of calcium and magnesium increase, reserves of sodium and potassium become depleted so incoming calcium and magnesium ions must exchange for other cations held on exchange sites; these can include toxic metals such as mercury, copper, and lead (Feick et al. 1972). These metals have been found in runoff and soil leachate in association with the use of deicing salts (Bäckström et al. 2004, Feick et al. 1972, Granato et al. 1995). Increases of calcium and magnesium from dust palliatives applied properly and at rates normally associated with dust control generally would be expected to be small compared to background levels of those base cations, so damaging environmental effects from release of toxic metals would not be expected (Singer et al. 1982). However, no studies could be found in which toxic metal concentrations were monitored to confirm this expectation.

One of the largest concerns of salt mobilization is contamination of domestic wells, because humans tend to be much more sensitive to salt intake than most organisms (Hanes et al. 1970). Elevated salt consumption by humans is linked to a number of health effects, including hypertension, coronary heart disease, and cardiovascular disease (Asaria et al. 2007). Increased salt concentrations have been found in wells, groundwater, and surface water from deicing (Bäckström et al. 2004, Granato et al. 1995, Kaushal et al. 2005), but similar studies could not be found for the use of dust control chloride salts. This does not mean that dust palliatives cannot pollute wells—only that studies of this type are rare or nonexistent, based on the available literature. It should be noted that because salinity levels in fresh waters are very low, small additions of salt can result in measurable increases; with sustained increases over time, elevated salinity can persist and result in ecosystem changes (Kaushal et al. 2005).

Fish, animals, and plants—unlike humans—can tolerate high concentrations of chlorides and salts before detrimental effects occur. Most freshwater fish species can withstand salt concentrations at least up to 400 ppm (Hanes et al. 1970), and many species can tolerate concentrations in thousands or tens of thousands of parts per million (Doudoroff and Katz 1953, Hanes et al. 1970). Salt tolerance or toxicity for fish depends not only on the concentrations and duration of exposure, but also the type of salt. Wiebe et al. (1934) found that golden shiners (*Notemigonus crysoleucas*) were much more sensitive to magnesium chloride and calcium chloride than sodium chloride, and fish that appear to be killed in high concentrations of sodium chloride could be revived if placed in fresh water; that was not the case for fish killed in magnesium or calcium chloride solutions.

Doudoroff and Katz (1953) reported that newly hatched rainbow trout could not tolerate concentrations of chloride (in sodium chloride) of about 3,256 ppm. However, older trout withstood chloride concentrations that were five to six times that level. One-day-old pickerel (*Stizostedion v. vitreum*) and whitefish (*Coregonus clupeaformis*) fry could withstand calcium chloride concentrations of 12,060 ppm and 22,080 ppm (i.e., 5,632 ppm and 10,326 ppm of chloride), respectively (Edmister and Gray 1948). Pike (*Esox* spp.), bass (*Micropterus* spp.), and perch (*Perca* spp.) were harmed when chloride concentrations reached 4,000 ppm (Hanes et al. 1970). Golden shiners exposed to 20,000 ppm sodium chloride survived an average of 1.33 h compared to 6.4 h and 0.5 h in the same concentrations of calcium chloride and magnesium chloride, respectively. At only 5,000 ppm chloride, survival times increased substantially, and were 148, 143, and 96.5 h for the three respective salts. There was little difference in survival times for calcium chloride salt at

20,000 ppm and 15,000 ppm for bream (*Abramis brama*). Survival times were 19.5 h and 17.7 h, respectively, but increased by more than double to about 49 h at 10,000 ppm (Wiebe et al. 1934).

Mortality of some small mammals and game birds, as well as pets, has been reported due to the consumption of deicing salts, but there is some debate as to whether the salt was responsible for the death. Deicing salts also contain chemicals, such as sodium ferrocyanide, to help prevent caking or reduce corrosion of vehicles, and these are highly toxic; thus, they may be responsible for or contribute to salt toxicity in birds, mammals, and fish (Hanes et al. 1970). However, controlled studies of salt intake by domestic animals have shown that toxicity and even death can occur at relatively high concentrations. Concentrations of 15,000 and 20,000 ppm resulted in reductions in food consumption and toxicity, respectively, to sheep (*Ovis aries*) (Peirce 1966). Dairy cows (*Bos* spp.) showed no health effects at sodium chloride levels of 9,000 to 10,000 ppm over 1 to 3 months, but they did show illness and reduced growth when drinking water contained 15,000 to 20,000 ppm sodium chloride (Weeth and Haverland 1961, Weeth et al. 1960). Conversely, nonlactating cattle could tolerate salt concentrations of 20,000 ppm (Heller 1933). Horses (*Equus caballus*) and sheep did not show any symptoms from drinking salt water with concentrations of 9,123 and 11,400 ppm, respectively (Ramsey 1924). In the laboratory, Heller (1932) and Heller and Larwood (1930) studied rats (*Rattus* spp.) supplied with drinking water containing 10,000, 15,000, 20,000, and 25,000 ppm of calcium chloride. These doses resulted in reproduction interferences, growth rate reductions, lactation problems, and mortality, respectively.

Chloride can be toxic to plants in low concentrations in the soil, but plants overall have a wide range of tolerance for salt that depends upon plant species, age, tissue type, overall nutrient conditions, and season (Hanes et al. 1976). Grasses are not easily injured by salts, but trees can be susceptible to damage (Hanes et al. 1970). Some of the tree and shrub species that are most intolerant to chloride concentrations are green ash (*Fraxinus pennsylvanica*), yellow-poplar (*Liriodendron tulipifera*), eastern white pine (*Pinus strobus*), Norway spruce (*Picea abies*), Canadian hemlock (*Tsuga canadensis*), spirea (*Spiraea* spp.), and rose bushes (*Rosa* spp.) (Hanes et al. 1976). Vegetation damage reported from deicing salts includes reduction in leaf color, leaf browning, premature leaf fall, twig and branch dieback, and mortality of roadside trees (Sucoff 1975). If dust-control salts are properly applied, plant damage or toxicity should result only through elevated salt concentrations in the soil and not through spray, as often occurs with deicing. Even with spray applications, if the liquid is applied following appropriate techniques, salt in soil should be the dominant concern, as splash from cars and plowing would not occur with dust palliation. Additionally, chloride movement (without splash) is primarily vertical; the lack of horizontal movement tends to limit terrestrial effects to near the area of application (Singer et al. 1982).

Slesser (1943) confirmed that there was little movement of chloride dust control suppressants laterally from roads. He found that 23 percent of calcium chloride added to a road to improve its stability was present in the upper 15 inches of the road 55 months after treatment, and horizontal transport was limited. Only 4.1 percent of calcium chloride was found to a depth of 15 inches 2 ft from the road edge after 55 months. On a second road, 9.6 percent of the calcium chloride salt was found in the upper 15 inches of the road 46 months after treatment, and only 6.2 percent was measured to a depth of 14 inches 1 ft from the road edge 25 months after application.

The few studies specifically focused on dust palliative salts suggest they do have lower potential for environmental effects than deicing salts. This is probably largely due to the much lower application rates used for dust control (Singer et al. 1982). Capillarity also contributes to slowing losses vertically by pulling salt back up toward the road surface during dry periods. Consequently, if chloride-salt dust palliatives are applied during appropriate conditions (i.e., avoiding rain and wet conditions), large slugs of salts normally would not be leached in short time periods (Singer et al. 1982). Monlux and Mitchell (2006) compared the performance of magnesium chloride liquid, calcium chloride liquid, and two solid calcium chloride applications (77 percent and 94 percent) on unpaved roads in four western states. They measured chloride concentrations pre- and post-application in river water, in soil samples (48 samples each, before and after application, including controls), and in 101 conifer trees adjacent to the road. After 2 yr there were no increases in chloride concentrations in river water samples. Chloride concentration increases were found in soil samples and tree tissue, but in both cases the concentrations were below levels that would be of concern for soils or that would threaten tree survival. Loreto et al. (2002) included magnesium chloride (DustGard®, Compass Minerals, Overland Park, KS) in the study of runoff quality in the Las Vegas Valley. The only notable chemicals in the runoff were chloride, boron, thallium, lead, and arsenic (though these soils have high naturally occurring arsenic concentrations). Thallium is very poisonous, but the concentrations found were only about 0.23 mg L^{-1}, and fish and aquatic invertebrate toxicity occurs at much higher concentrations: 10 to 60 mg L^{-1} and 2 to 4 mg L^{-1}, respectively (Zitko et al. 1975).

For all of the other types of dust suppressants, there is little specific information available about their environmental effects. Bennett (1994) stated that because clays are naturally occurring geologic materials, they are harmless to the environment. This probably is a reasonable assumption. Lunsford and Mahoney (2001) stated that enzymes are nontoxic and harmless to the environment, but they offered no studies or literature to substantiate that statement. Little is known about the environmental effects of synthetic polymers, but their cost and the high amount of quality control required during application may impede their widespread use (Lunsford and Mahoney 2001). Loreto et al. (2002) found that an acrylic polymer (EK35®, Midwest Industrial Supply, Inc., Canton, OH) applied to disturbed soil was second only to a petroleum-based dust suppressant for the number of contaminants and the concentrations of those contaminants in runoff. This polymer had the highest concentrations of volatile organic compounds in runoff (slightly less than 200 µg L^{-1}) and the second highest concentrations of semivolatile organic compounds (~160 µg L^{-1}). BOD and COD also were high, about 42 mg L^{-1} and 900 mg L^{-1}, respectively. Runoff from a nonpetroleum-based road oil (Road Oyl®, BoVill Industries LLC, Redmond, WA) contained moderate numbers of contaminants, but most were at very low concentrations. Aluminum was the exception, with the highest concentrations in runoff of any suppressant tested, and exceeding 2,800 ppb.

Clearly, an important key to reducing the potential environmental impacts of every type of dust palliative is to follow the manufacturer's instructions for application after reviewing the safety data sheets (SDS). Further, local, state, or federal regulations should be understood and followed at all times. Particular attention should be paid to ensure that chemicals are not unintentionally applied to water bodies or ditches leading directly into water bodies (Bolander and Yamada 1999). Spray and mechanical applications should cease when crossing bridges (Piechota et al. 2002).

Gravel application on a new cut-and-fill forest road, and woody material from the right-of-way positioned as a windrow on the fillslope. (Photo by U.S. Forest Service, Northern Research Station.)

CHAPTER 4

Road Use

Road use results in soil losses through both water-driven erosion, and wind- and traffic-generated dust. The road use variables that are most influential in generating sediment losses and that can be manipulated with respect to BMP implementation are axle load and wheel configuration, vehicle weight, tire pressure, traffic intensity (number and timing of vehicle passes), and vehicle speed. This chapter is divided into sections by these major variables.

Axle Load, Vehicle Weight, and Tire Pressure

Large axle loads, vehicle weights, and tire pressures all contribute to increased soil compaction, which in turn can result in several negative soil-related effects. Among these effects are restricted root growth, poor root zone aeration and gas diffusion, poor drainage, and decreased vegetative growth (Abu-Hamdeh et al. 2000, Håkansson and Petelkau 1994, Helms and Hipkin 1986, Johnson et al. 1986, Stone and Elioff 1998). Much of the early research on these variables was in agricultural studies in an attempt to identify ways to avoid reductions in crop yields as a result of using farm equipment in fields. Other significant preliminary information came from studies in the early 20th century by the military in an attempt to alleviate problems with transporting and operating extremely heavy pieces of equipment. Not until relatively recently have research studies involving use of low-volume roads for forestry-related practices been initiated. Because soil loss on low-volume roads is tied strongly to rut formation (Foltz and Burroughs 1990, 1991), maintaining optimal compaction without rutting and avoiding pavement breakup have been the objectives for unpaved and paved roads, respectively.

Road surface damage is a function of three variables: surface materials and thickness, environmental conditions, and the traffic applied to the road (Kestler et al. 2007). However, damage to the road surface by traffic can occur as the result of changes to both surface layers and subsurface soil. Large axle loads and vehicle weights primarily contribute to elevated soil compaction in subsoil layers (Danfors 1994, Håkansson and Petelkau 1994, Janzen 1990), whereas high tire pressures primarily affect compaction near the soil surface (Danfors 1994, Janzen 1990).

The depth to which subsoil compaction occurs increases with the axle load (Danfors 1994). Axle loads of 4 Mg resulted in significant compaction at a depth of 30 cm, while 6-Mg and 10-Mg axle loads showed significant compaction extending down to 40- and 50-cm depths, respectively (Håkansson and Petelkau 1994). Subsoil compaction resulting from large axle loads or vehicle weights can be lessened by reducing the weight of the vehicle and its load, or alternatively by adding more axles and wheels (McLeod et al. 1966, Sebaaly 1992). However, the position of added wheels has a substantial influence on whether changes to subsurface compaction will ultimately result. For example, dual wheels mounted close together may reduce surface compaction but not subsurface compaction; therefore widely spaced extra wheels are preferable (Håkansson and Petelkau 1994). The use of tandem axles does not allow for doubling the load that can be carried because there is an interaction of the two axles that will increase compaction at some depth in the subsoil by more than the sum of each axle individually (Danfors 1994, Håkansson and Petelkau 1994).

Studies involving alterations of wheel or axle configurations sometimes entail reduced tire pressures because more wheels are used (e.g., Danfors 1994, McLeod et al. 1966, Shalaby and Reggin 2002). This makes it difficult to tease out the reductions in compaction that are attributable to changing wheel or axle configurations from those attributable to changing tire pressures. However, because tire pressure is primarily responsible for contact pressure at the soil surface (Janzen 1990), changing tire pressures exerts little influence on subsoil compaction. Danfors (1990) reported only minimal differences in soil compaction at 30- and 50-cm depths among tire pressures of 21.75, 14.5, and 7.25 lb inch^{-2} (psi) with the same loads. Similarly, changes in subsoil porosity were very similar for 30- to 40-cm or 40- to 50-cm depths from passes made by single-axle vehicles with 7.25- and 21.75-psi tire pressures (Danfors 1994).

Of the three variables described in this section, tire pressure has been the most studied on low-volume forest roads, though forest applications still remain less studied compared to agricultural applications (Kestler et al. 2007). Ground contact pressure is primarily responsible for compaction at the soil surface (Håkansson and Petelkau 1994), and reduced tire pressure acts as a BMP by reducing the pressure of the tires transferred to the road surface as the tire flattens out and elongates to support the vehicle weight (Fig. 5). This increase in the tire footprint reduces the pressure exerted at any point where the tire is in contact with the road surface compared to a normally inflated tire. However, every tire has inflation design limits to ensure the tire can support its load and keep tire deflection at levels that will allow

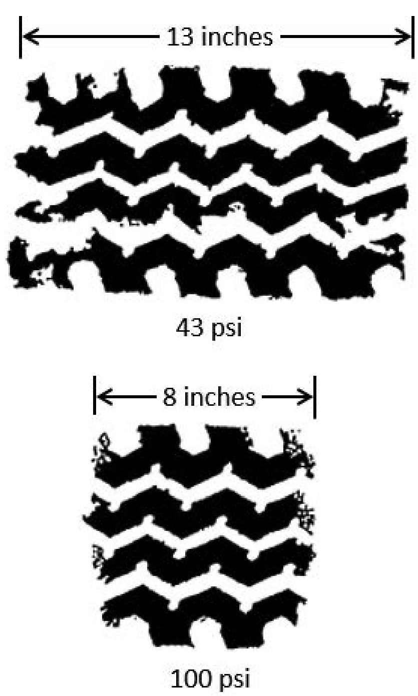

Figure 5.—Example of tire surface area contacting the driving surface at different tire pressures (psi = lb inch^{-2}). From U.S. Forest Service (1993).

tires to remain sealed on the wheel and resist failure (Janzen 1990). These limits vary with the materials and techniques used to manufacture the tire (Yap 1989), so different types of tires can have markedly different low-end inflation pressures. Under reduced tire pressures, 20-percent sidewall deflection is optimal (Kestler et al. 2007). Reduced vehicle weight also can reduce pressure on the road surface, but if the tires are inflated within their design limits, the reduced vehicle weight translates to a much smaller difference in tire deflection and tire footprint than changes that result from reducing tire inflation (Douglas et al. 2000, Steward 1994).

Wide tires specifically designed to be driven under high loads with very low inflation pressures, known as "terra tires," have been developed for agricultural use to achieve the advantages of tire underinflation. Terra tires that ran under 4, 6, and 9 psi resulted in less soil compaction than single or dual tires at higher or even similar tire pressures when run off-road (McLeod et al. 1966). Under 100 ft of forward travel at 6 psi, the terra tires displaced 31.1 ft^3 of soil compared to 44.0 ft^3 for dual tires. At 6 psi and 9 psi, the terra tires displaced 24.6 ft^3 and 29.2 ft^3 of soil compared to 32.4 ft^3 and 38.0 ft^3 of soil for dual tires at those inflation rates.

There are two ways to reduce tire pressures. One is to manually reduce the pressure so that re-inflation can occur only at a garage or similar facility; this is called the constant reduced pressure method, or CRP (Foltz 1994). In this approach, the target pressure is determined by the vehicle gross weight, the type of tire employed, and the maximum speed of travel that will be used. Because the tire pressure stays low, it has a low ground contact pressure on all of the roads the vehicle travels. Therefore, tire pressures are near optimal for low-volume roads, but suboptimal for highway use where higher travel speeds are used (Kestler et al. 2007). Conversely, if normal highway tire pressures are used continually, these higher pressures result in higher contact pressures that are less than optimal for controlling soil losses on low-volume roads.

The second method to reduce tire pressures allows the driver to change tire pressures while the vehicle is moving (Foltz and Elliot 1997). This automatic pressure changing system is called the central tire inflation system (CTIS) and allows optimal tire pressures to be used for the conditions present. Because tire pressures can be adjusted to lower values on low-volume, unpaved roads, rut creation and erosion can be reduced and greater comfort and longer tire wear can be achieved by re-inflating the tires for paved surfaces.

CTIS has existed since before World War II to improve the mobility of large military vehicles (Foltz 1994, Foltz and Elliot 1997, Kestler et al. 2007). It is expensive to install ($15,000 to $20,000; reported in 2007), so in the United States its use and testing have been confined largely to within the military. The U.S. Forest Service has been the other primary user, with objectives of using CTIS to reduce road maintenance and aggregate thickness of low-volume roads, but the agency has far fewer vehicles equipped with CTIS than does the military (Kestler et al. 2007).

Studies of reduced tire pressures have shown fairly consistent and substantial reductions in rutting and sediment losses compared to higher tire pressures. Foltz (1995) measured sediment losses and rut formation attributable to highway tire pressures (90 psi in all tires), CRP (70 psi in all tires), and CTIS (70 psi on wheels on steering axles, 30.5 psi on other axles for unloaded trucks, or 50 psi on other axles on loaded trucks) on segments of a low-volume road with marginal-quality aggregate. Total sediment losses after three winter/early spring seasons of use in Oregon from the three respective tire pressure systems were 2,678 kg, 1,465 kg, and 530 kg. This equates to average reductions of 80 percent for the CTIS and 45 percent for CRP compared to highway pressures. The best improvements for both systems occurred during the wettest of the 3 yr, and the least during the driest year.

Tests using an individual simulated rain event showed a similar pattern of sediment loss and nearly identical percentage reductions compared to the highway pressure. Highway tire pressures resulted in 116 kg, CRP 54 kg (47-percent reduction), and CTIS 38 kg (83-percent reduction) (Foltz 1995). Foltz and Burroughs (1991) also found reduced tire pressure to be more effective at decreasing sediment losses under wet conditions than dry. Three replicate runs with loaded log trucks using normal tire pressures (95 psi) had only 7 percent greater sediment losses than reduced tire pressures (70 psi) during use with no simulated rain. Under wet conditions with simulated rainfall, normal tire pressures yielded 120 percent more sediment than reduced tire pressures. Under very wet conditions, the normal tire pressures yielded 73 percent more sediment than reduced tire pressures.

Differences in sediment losses reported in most tire pressure studies have been attributed to the degree of rut development. Foltz (1995) and Foltz and Elliot (1997) reported 70- to 87-percent reductions in sediment losses from lower tire pressures because of less rutting. Ruts worsen under wet conditions because surface soil and subsoil are more susceptible to compaction, and as water concentrates in ruts, soil detachment accelerates. For the three tire pressures examined by Foltz (1995), rut depths were 133 mm for highway inflation rates, 32 mm for the CRP, and 8 mm for the CTIS after 3 yr (January–March only) of use. There was little change in the latter two depths from year 2 to year 3, but the highway rut depths continued to increase throughout the study and deepened from 38 mm to 127 mm from year 2 to year 3. Rutting also occurred more quickly with tire pressures of 100 psi than with 60 psi on a bituminous surface treatment applied over a 20-cm-thick aggregate layer (Kestler and Nam 1999). Tests with reduced tire pressures in Canada decreased rut development and even allowed ruts to be healed and smoothed out if truck drivers varied their wheel paths slightly (Kestler et al. 2007, Shalaby and Reggin 2002). Foltz and Burroughs (1991) found that ruts formed under wet conditions with normal tire pressures were 23 cm deep after 16 passes with a loaded log truck compared to 4 cm under reduced tire pressure; after 16 passes the subgrade also began to fail under normal tire pressures.

Research findings have supported the U.S. Forest Service's objectives for testing reduced tire pressures. The frequency of required road maintenance is reduced due to less rut development (Foltz 1995, Foltz and Elliot 1997, Steward 1994), and the amount of aggregate required for surfacing roads can be decreased by as much as 50 percent (Steward 1994). Both of these advantages can result in substantial savings in construction and maintenance (Smith 1993, Steward 1994). Because disproportionate amounts of sediment loss and road damage, including rutting, occur during wet periods, one of the additional major benefits of reduced tire pressures is earlier access to sites (e.g., 1 to 2 wk earlier) during spring thaw or wet periods (Kestler et al. 2007). This is particularly applicable for poor-quality roads or roads surfaced with poor-quality aggregate because these are simultaneously most susceptible to damage during wet seasons and most helped by reductions in tire pressures (Foltz 1995). However, if the road's subgrade is extremely weak, tire pressure reductions will not be sufficient to overcome these problems and early reentry will not be possible (Taylor 1988).

Traffic Intensity

Traffic intensity (number of passes) on unpaved roads, particularly by heavy equipment, substantially influences sediment losses and the persistence of sediment availability (Kahklen and Hartsog 1999, Luce and Black 1999, Reid and Dunne 1984). Therefore, controlling traffic, including requiring periods of no use, is considered an important management tool for limiting sediment supply and sediment transport from unpaved roads (Croke and Hairsine 2006). Unused and abandoned roads have more-permeable road surfaces than actively used roads (Reid and Dunne 1984), in part because freeze/thaw and wet/dry cycles help break up road surfaces (Gatto 1998, Knapp 1992) and vegetation can become established (Jordán and Martínez-Zavala 2008, Knapp 1992), both of which improve infiltration. On lightly used roads, sediment control comes from the reduced production of available (loosened) sediment from tire friction, slippage, and other factors. Croke et al. (2006) measured available sediment levels that were half as great per unit area from infrequently used roads as those from well-used main access roads.

Reid and Dunne (1984) estimated sediment production on unpaved roads across a range of use levels using mathematical relationships between precipitation and culvert runoff characteristics for watersheds in the Pacific Northwest. Sediment losses were strongly related to the amount and type of traffic to which the road was subjected. Heavily used roads (more than four loaded log trucks per day) accounted for almost 71 percent of the average sediment yield, while moderate use (fewer than four log trucks per day) and temporary non-use (heavy use for 2 days, then no use) roads each accounted for slightly less than 10 percent of the annual sediment yield. Light use (light vehicle use but no log trucks) and abandoned roads accounted for 3.8 percent and 1.2 percent of sediment production, respectively. MacDonald et al. (2001) found sediment production doubled in a road segment subjected to heavy traffic compared to one with only light vehicle use on the island of St. John in the U.S. Virgin Islands.

Annual yields of sediment from a heavily used existing graveled road were 13 times greater (44.2 ton mi^{-1} of road) than from an unused road (3.4 ton mi^{-1}) in western Washington state (Wald 1975). The average suspended sediment concentration from the heavily used road was 1,306 mg L^{-1}. Much higher suspended peak concentrations were measured from sediment generated by 30-min simulated rain events performed before, during, or after 20 passes of loaded trucks in New Zealand (Coker et al. 1993). Rainfall simulation during the truck passes resulted in the highest peak sediment concentration of 130,000 mg L^{-1}, which fell to 12,000 mg L^{-1} when use stopped. Road use generated peaks of 21,000 mg L^{-1} and 10,000 mg L^{-1}, respectively, when the rain was applied immediately after and before the 20 passes, and fell to 6,000 mg L^{-1} and 3,000 mg L^{-1}, respectively. Similar results were reported by van Meerveld et al. (2014) for rainfall simulation events with loaded log truck traffic in British Columbia. Peak sediment concentrations ranged from 5,200 to 15,000 mg L^{-1} during medium-intensity (~15 mm h^{-1}) rainfall simulations with three to six truck passes compared to peak concentrations of 900 to 3,800 mg L^{-1} with no truck traffic. Thirteen to 40 percent of the total sediment captured during simulated events was attributed to truck traffic. Wooldridge (1979) also observed increased suspended sediment as a result of heavy-truck traffic during periods of light and moderate rainfall in Washington.

Not surprisingly, traffic intensity also affects sediment generation at stream fords. Thompson and Kyker-Snowman (1989) evaluated the effects of individual and simulated multiple vehicle crossings (4 to 10 trips in rapid succession) by four-wheel all-terrain vehicles (ATVs), four-wheel-drive pickup trucks, and motorcycles on unimproved ford crossings on two small streams. Turbidity and total suspended solids increased from the off-road vehicles, but the average increases were significantly lower than values obtained from heavier logging equipment measured in an earlier phase of the study. Some individual passes by the pickup truck in spring and summer generated turbidities as high as that of some of the individual logging equipment passes 15 ft downstream of the crossing. The impacts of ATVs and motorcycles traveling in groups were similar to that of logging equipment 15 ft downstream of the crossing, but 100 ft downstream the impact was much less than that of logging equipment. Individually, the four-wheel-drive pickup trucks generated more sediment than the ATVs and motorcycles. Less sediment was generated from streambanks with steeper approaches because vehicles had to slow down to enter and exit the fords.

Foltz and Truebe (1995) studied the simultaneous effects of traffic intensity and aggregate quality on sediment production. Log trucks with an inflation rate of 90 psi made 268 and 616 passes during the winters of 1992 and 1993, respectively, on road segments with good- and marginal-quality surface aggregate (Table 13). Sediment production (kg) was 30 times lower for the lower traffic intensity on marginal-quality aggregate in 1992 than it was for approximately double the traffic in 1993. By comparison, reduced traffic in 1992 resulted in six times lower sediment losses compared to higher traffic in 1993 on the good-quality aggregate. Some of the differences in sediment loss between years were due to greater precipitation during 1993; but when those differences are accounted for by expressing losses as kilograms of sediment per millimeter of precipitation, sediment production was 13 times higher in 1993 with the increased traffic loads for marginal aggregate and 3 times higher with good aggregate. Foltz (1999) extended this study by adding a year with another approximate doubling of the number of passes (1,205) in 1994 and then adding a fourth year when no traffic was allowed on the road. Even after 3 yr of heavy traffic use, as defined by Reid and Dunne (1984), eliminating traffic in 1995 substantially reduced sediment losses (Table 14) where significant rut formation had occurred in the marginal-quality surface materials (Foltz 1999, Foltz and Truebe 1995).

Traffic Speed

Little research has been published in which the influence of changing travel speed has been studied in relation to changing water-driven sediment losses on unpaved low-volume roads. Travel speed probably has some effect for a given road surface in that speeds that result in tire slippage hasten rut development and washboarding (Shoop et al. 2006), and presumably result in elevated

Table 14.—Sediment production from forest road sections with different aggregate quality (data from Foltz 1999)

Year	Number of passes	Aggregate quality	Sediment loss		
			kg	*kg ha^{-1}*	*g mm^{-1a}*
1992	268	Marginal	47.3	1,850	334.7
		Good	12.7	500	82.0
1993	616	Marginal	1,400.0	54,800	2,643.0
		Good	81.0	3,170	157.8
1994	1,205	Marginal	1,231.3	48,180	3,875.1
		Good	262.1	10,260	823.7
1995	0	Marginal	149.0	5,830	188.7
		Good	32.2	1,260	41.6

[a] Mass of sediment in g per mm of precipitation.

Table 13.—Sediment production from forest road sections with different aggregate quality following natural precipitation events (data from Foltz and Truebe 1995)

Year	Aggregate quality	Sediment mass	Mass ratio	Sediment production		Average concentration
		kg	*marginal:good*	*kg ha^{-1}*	*kg mm^{-1a}*	*g L^{-1}*
1992	Marginal	47.3	3.7	1,850	0.57	2.1
	Good	12.7		500	0.15	1.2
1993	Marginal	1,400.0	17.3	54,800	7.58	27.6
	Good	81.0		3,170	0.44	7.7

[a] Mass of sediment in kg per mm of precipitation.

sediment production. However, because the surfaces of low-volume roads normally are not smooth or conducive to high speeds in terms of either comfort or safety, there are already de facto physical limitations in place that dictate lower speeds on these roads, which may explain the general lack of research in this area.

The published literature involving studies of traffic speed focuses primarily on dust suspension. Road dust can be generated from both paved and unpaved roads. It includes many different types of particles, including metals, organometals, rubber, and exhaust chemicals, as well as soil materials (Rogge et al. 1993). More than 100 types of chemicals are associated with nonsoil particulates, and these tend to be of most concern in urban environments, where humans are subjected to high concentrations of potentially toxic fine particles and aerosols through respiration (Nicholson et al. 1989). On unpaved roads where traffic intensity is much less than on paved roads, most fine particles originate from the bed of dirt roads or from the matrix of the surfacing materials (Jones 2000a, Wisconsin Transportation Information Center [WTIC] 1997). Consequently, these particulates are the focus of this section.

The premise that decreased traffic speeds reduce dust emissions from roads is well accepted—probably because this phenomenon is commonly and easily observed. Decreasing speed reduces turbulence, which lowers particle suspension (Nicholson et al. 1989); however, the specific reduction is dependent upon the road and traffic conditions. For example, increased moisture at the road surface reduces particle suspension (Nicholson et al. 1989). Taller and heavier vehicles also suspend greater amounts of dust at a given speed than shorter, lighter vehicles (Dyck and Stukel 1976, Gillies et al. 2005). The recency of past traffic can influence the amount of particles available for resuspension. Fewer particles, especially large particles, remain available at the road surface for resuspension if several vehicles have recently used the road at moderate speeds (Nicholson et al. 1989).

Even though these other variables have some influence on dust emissions by traffic, substantial reductions have been reported as the result of reducing travel speeds. A reduction from 45 mph to 35 mph resulted in a 22-percent reduction in dust (SynTech Products 2011). Reducing speeds from 40 mph to 35 mph and from 40 mph to 20 mph reduced dust emissions by 40 percent and 65 percent, respectively (Succarieh 1992, WTIC 1997). On paved roads, the minimum speed needed to suspend dust into the air is about 15 to 20 mph depending on particle size (Nicholson et al. 1989), but the threshold velocity on unpaved roads is probably much less (Watson 1996). Sanders and Addo (1993) determined that dust generation was linearly related to vehicle speed during tests on an unpaved road in Colorado. For speeds from 20 to 50 mph, the approximate equation was:

$$Y = 0.85 + 0.16X,$$

where Y was grams of dust generated and X was vehicle speed in mph ($r^2 = 0.98$; equation derived from Figure 10 in Sanders and Addo 1993).

On unpaved roads, dust particles suspended in the atmosphere by traffic are composed of aggregates of fine clays to sand-sized particles; thus, emitted particles can vary in size by several orders of magnitude (Pinnick et al. 1985). Small clay particles are particularly susceptible to loss (Succarieh 1992). In certain cases the linearity between vehicle speed and dust observed by Sanders and Addo (1993) extended across the entire suite of particle sizes within the PM_{10} class size (Etyemezian et al. 2003, Gillies et al. 2005); in another situation, linearity existed only for particle sizes less than or equal to $PM_{2.5}$ (Williams et al. 2008).

Conventional theory regarding PM_{10} dust emissions from roads has been that they redeposit near their sources (Countess 2001, Watson et al. 2000), particularly when wind speeds are low. Consequently, implications for water quality effects would be expected to be limited to water bodies that are next to or cross under roadways, or to water bodies that are connected to ditches adjacent to roadways. However, where moderate wind speeds (3 to 5 m s^{-1}) have been present in combination with little roadside vegetation, equivalent amounts of road dust have been measured at towers located 9 m and 50 m downwind of unpaved test-road sections (Etyemezian et al. 2003). More recent literature reflects a great deal of uncertainty about deposition velocity and dispersion of dusts, and the actual fate of road-dust emissions is not well understood (Gillies et al. 2005). Thus, it is not currently possible to make generalized statements about how well controlling vehicle speed protects water quality.

CHAPTER 5

Stream Crossings, Stream Crossing Approaches, and Wet Soil Crossings

This chapter examines crossings in two fundamentally different types of environments: streams and wet soils (i.e., wet/weak soils, wet meadows, and wetlands). Studies of the effectiveness of stream crossing BMPs are focused directly on water quality or channel condition. In contrast, wetland studies focus primarily on BMPs to control rutting depth and damage to the surface because those effects can alter hydrology (especially subsurface flow). As noted in the Introduction (Chapter 1), BMP effectiveness of aquatic passage designs is not considered in this synthesis, but information about effects on fluvial geomorphology and substrate characteristics upstream and downstream of stream crossings is included.

Stream Crossings

An abundance of forest hydrology and watershed management literature states that stream crossings (or water body crossings in the broader sense) are the road areas with the greatest potential for contributing nonpoint source pollution to water (e.g., see Kruetzweiser and Capell 2001, Lane and Sheridan 2002, Rothwell 1983, Swift 1988, Weaver and Hagans 2004). This conclusion has been based largely on measurements of in-stream sedimentation or turbidity after stream-crossing construction and from comparisons of substrate, habitat, or channel condition above and below existing water body crossings. For example, Eaglin and Hubert (1993), Schnackenberg and MacDonald (1998), and Cornish (2001) reported increases in turbidity and sediment input to streams from stream crossings. Eaglin and Hubert (1993) found lower amounts of cobble substrate associated with stream crossings. Similarly, Schnackenberg and MacDonald (1998) found that the percentage of fine particles (<8-mm diameter) in the streambed was significantly and positively related to the number of road crossings, and explained 61 percent of the variability in the amount of those particles present. Because of the relatively large numbers of these types of studies and the inherent connectivity between crossings and water bodies (Weaver and Hagans 2004), crossings are well accepted as the primary conduit of sediment inputs to water in forested watersheds.

Given the broad acceptance of this tenet, one would expect that a substantial amount of research would have focused on identifying types of crossings, crossing features, or construction techniques that could reduce sediment inputs. Instead, there is surprisingly little research in these arenas, and this lack of information on crossings is noted commonly (Blinn et al. 1998; Bouska et al. 2010; Thompson et al. 1994, 1996; Tornatore 1995; Welch et al. 1998). Papers by Thompson et al. (1994, 1995, 1996) were the first that documented the effects of different types of stream crossings on water quality. Before their research, most studies focused on forest roads as a whole, rather than the stream crossings in particular (Welch et al. 1998).

Stream crossings include a wide variety of techniques and structures. Fords, culverts, and bridges are the most common types of stream crossings, and virtually every type of crossing used for streams and rivers (Clarkin et al. 2006) can be placed into one of these three categories. For lower order streams (e.g., ephemeral and intermittent channels), less complicated and less expensive crossings typically are employed (Clarkin et al. 2006). As stream width and discharge increase, the cost and complexity of crossing structures increase.

For the purpose of motorized vehicle use, a ford is a location in a stream or river that is shallow enough, and without large rocks or boulders, to allow traffic to pass through. Fords can be classified as either unimproved (or natural) or improved (or mitigated or renovated) (Milauskas 1988, Welch et al. 1998). Unimproved fords have had no changes made to the water body or approaches[1] to help with vehicle passage or resource protection. Sediment levels in unimproved fords can originate from many causes. Waves produced by vehicles moving through the ford can erode soil from streambanks. Ruts can be created in wheel tracks in the approaches that allow overland runoff during storm events to become concentrated and directed to the channel. Sediment can be washed off vehicles as they contact streamwater in the ford. Sediment present in the channel that would not otherwise move under some flow conditions (e.g., baseflow) can become mobilized by the presence of the vehicle or the influence of the vehicle on the water in the stream (Clarkin et al. 2006). Additionally, streamwater can be polluted from vehicles traveling through fords. No studies could be located where pollutants from cars or trucks were measured, but U.S. Forest Service monitoring found measurable levels of naphthalene and hydrocarbons present in streamflow during an off-highway vehicle event that had 200 to 500 vehicle crossings day^{-1} for 6 days (Clarkin et al. 2006).

[1]Approaches are defined as the length of road or ditch line from which water would drain directly to the crossing. The outer boundaries of an approach are usually definable by road-surface drainage features or grade changes on the road surface and ditch line.

However, all of the compound concentrations were within limits established in water quality standards.

In contrast, at a minimum, improved crossings typically involve laying back streambanks to reduce the slope of the approaches to the channel, and installation of gravel, rock, or some type of synthetic pavement in the approaches and on the streambed to reduce the amount of rutting and detachment or suspension of sediment that occurs with travel through the ford. Improved fords also can employ other techniques to reduce contact and disturbance between vehicles and the streambed and banks (Looney 1981). A fairly common type of improved ford uses multiple small culverts encased in concrete (Fig. 6) which allow passage of low flows through the culverts, and the concrete pavement is overtopped during higher flows (Milauskas 1988); these structures are termed “vented fords.”

Culvert crossings are composed of single or multiple pipes or box-shaped or arched structures placed into the channel, and positioned parallel to the flow of streamwater with the road surface located above the normal bankfull level. Culverts can be constructed of metal, plastic, or masonry products. Culvert installation normally involves excavating the streambed or banks, or both, within the area where the culvert will be placed and then refilling over and around the culvert after installation. These disturbances can result in short-term inputs of sediment to the stream, even if water is

Figure 6.—A vented ford. From Keller and Sherar (2003).

controlled (e.g., by diversion) during the installation. Redisturbance of the stream can result in another subsequent short-term pulse of sediment into the channel if the culvert is temporary and removed after its required use has ended. Such redisturbance often is considered acceptable because long-term, chronic sediment inputs associated with permanent culverts are avoided. Permanent culverts that remain in place for the long term may have chronic inputs of sediment because of road drainage (see Chapter 2), or even more-extreme inputs during high flows that cause washouts, especially if there is diversion potential. Temporary and permanent culverts are installed on small and moderate-sized streams, but as streams become very wide, bridge installation becomes economically competitive.

Bridges, like culverts, can be temporary or permanent installations. Temporary bridges are portable structures that typically are used on lower volume roads crossing smaller streams (Blinn et al. 1998, Taylor et al. 1996b), whereas permanent bridges are installed on wider streams and rivers where decadeslong use is planned (Taylor et al. 1996b). Temporary bridges almost never include pilings (i.e., supports extending into the channel bottom) and simply span from streambank to streambank. They may be hinged and foldable, or modular (disassembled) to allow easier transport and installation (Blinn et al. 1998, Keliher et al. 1995). Bridges with no pilings may result in little or no streambed disturbance, but bridges that require pilings necessarily involve channel disturbance during construction.

Crossing Effectiveness

Studies investigating the effectiveness of stream crossings fall into two categories: those that are relatively short-term, such as during crossing installation/improvement or during discrete periods (e.g., logging use), and those that examine longer term influences. The former tend to involve monitoring of the receiving stream's water column for suspended solids/sediment or turbidity. The latter tend to focus on geomorphic and substrate conditions.

Improved fords consistently produce less erosion and sedimentation than unimproved fords. Sample et al. (1998) found that hardening a ford by replacing the original streambed materials with compacted rock and gravel resulted in substantially less measurable sediment downstream than the amount measured downstream of an unimproved ford with traffic. A pole ford (i.e., a ford filled with logs and two 16-inch-diameter iron pipes laid parallel to flow) and a ford filled with randomly oriented sawmill slabs in conjunction with hay bales anchored downstream from the ford to capture and filter sediment had significantly lower turbidity and suspended solids concentrations 15 ft and 100 ft downstream (resulting from passes by logging equipment) than unimproved fords (Thompson and Kyker-Snowman 1989). During some periods, elevated turbidity was measured as far as 1,000 ft and 2,640 ft downstream of the unimproved fords.

Tornatore (1995) and Tornatore et al. (1994) employed a similar crossing on a skid road in which a metal pipe culvert was placed in a stream with pole-sized logs filled in around it. Installation of the crossing occurred during extremely low flow, but median suspended solids concentrations were 412 mg L^{-1} and 28.5 mg L^{-1} 1 m and 61 m, respectively, downstream from the outlet during installation, compared to only 1 mg L^{-1} upstream. The peak suspended solids concentration was more than 1,000 mg L^{-1} 1 m downstream. Installation effects disappeared after about 96 hours, and high flows during snowmelt several months later did not result in increased suspended solids from the poled ford. About 6 months after installation, six simulated skidder passes during a period of baseflow yielded median suspended solids concentrations of 6.2 mg L^{-1} and turbidity of 7.2 NTU 1 m downstream, compared to 2,560 mg L^{-1} and 863 NTU 1 m downstream of an unimproved rocky-bottom ford.

Looney (1981) observed that unimproved fords had between 70 and 150 percent greater sediment losses during short periods of whole-tree skidding compared to improved fords in which rubber mats with side walls (constructed from conveyor belting) were installed. Although this design was referred to as a "dam bridge," it acted as an improved ford during use. The mat floated on the water surface when not in use, but was pressed to the stream bottom when a vehicle drove on it, thereby providing protection to the channel bed and banks. During 1.33 h of skidding, an unimproved ford yielded 52.7 kg of sediment, compared to 31.2 kg from the dam bridge. Two hours of skidding at another site yielded 208.4 kg of sediment from the ford, and 82.3 kg from the dam bridge.

Improvements to fords may help decrease total sediment inputs to streams compared with unimproved fords, but the process of installation or removal of ford mitigation measures can contribute to short-term sediment inputs. Thompson et al. (1996) examined the immediate effects of improving two existing fords in a third-order stream. The fords were cleared mechanically of logs that had been placed in the channel, ruts were removed, and 132 tonne of surge stone was placed in the bed and on the approaches of each ford. The mechanical in-channel work resulted in the highest peak sediment concentrations for ford 1 (2,815 mg L^{-1}), whereas the addition of gravel to ford 2 resulted in its peak sediment concentration (1,355 mg L^{-1}). By comparison, Blinn et al. (1998) reported that a ford containing a pipe mat constructed of bundled polyvinyl chloride (PVC) pipe laid parallel to flow and underlain with geotextile material was very effective at controlling sediment losses during use by logging equipment. They provided no data, but reported no visual increase in turbidity immediately below or farther downstream from the ford after 20 passes with a log forwarder. However, some sediment did enter the stream when the geotextile was removed as the ford was being dismantled. Such sediment losses are common when fabrics used to protect streambeds are extracted (Mason and Greenfield 1995).

Fords (including those that have been improved) tend to be less effective at controlling sediment inputs than other structural types of crossings—at least in the short term. A portable bridge with cribbing (logs laid parallel to streamflow against each bank and at the middle of the bridge to support the portable bridge) had much lower downstream turbidity measurements and suspended sediment concentrations than a pole ford and a ford filled with sawmill slabs and downstream hay bales (Thompson and Kyker-Snowman 1989). The pole ford was more effective than the ford filled with slabs, and the effectiveness of the slabs + hay bales was reduced further when the hay bales were less than 50 ft downstream from the ford because they backed water up into the ford. Looney (1981) found a culvert also to be more effective at controlling sediment than an improved ford (dam bridge) and unimproved ford. This is because the primary periods of sediment generation for the culvert occurred during installation and removal, whereas sediment continued to be generated by the dam bridge and unimproved crossings with each pass. Consequently, within only a few hours of use, total sediment yields for both types of fords could easily exceed that associated with culvert installation and removal.

Thompson et al. (1994, 1995) compared sediment concentrations from the installation of one temporary corrugated metal culvert and two temporary glue-laminated (glulam) timber bridges. Mean net sediment concentration increases (downstream minus upstream) were 341 mg L^{-1} for the culvert, compared to 66 mg L^{-1} and 38 mg L^{-1} for the bridges. The mean sediment increase was higher for the culvert because its installation involved channel excavation, placement of crusher run gravel for a bed, and refilling around the culvert after placement of the pipe. Neither bridge installation disturbed the streambanks or channel other than preparing the top of the streambanks for bridge placement, making sediment inputs much lower. Sediment contributions remained negligible when one of the bridges was removed during the study, presumably due to the lack of in-channel disturbance and limited bank disturbances. In contrast, when even limited (one equipment pass) in-channel disturbance was required for installation of a portable steel bridge, Tornatore (1995) found that turbidity 1 m downstream was significantly greater than upstream (483 NTU 1 m downstream versus 1.8 NTU upstream, respectively), and exceeded that associated with a pole ford (238 NTU 1 m downstream). However, the effects from bridge installation were more localized, with turbidity of 14.6 NTU compared to 56 NTU 61 m downstream from the pole ford during installation. After installation, there was little difference in median turbidity among the portable bridge, the pole ford, or a culvert backfilled with shale on skid roads, but during use these all performed significantly better than an unimproved ford.

Witt et al. (2011) reported similar results when suspended solids and turbidity from unimproved fords were compared to PVC-pipe bundle crossings, culvert crossings, and portable bridges on ephemeral channels after logging equipment use. The improved crossings resulted in significant turbidity reductions ranging from 67 percent for the pipe bundles to 77 percent for culverts and 84 percent for the bridges versus the unimproved ford. A study by Aust et al. (2011) comparing temporary bridge, culvert, and pipe + pole crossings to improved fords found no significant differences between crossing types though average sediment concentrations were 217 mg L^{-1} higher below the crossings compared to above the crossings. Mean sediment increases were highest at 253 mg L^{-1} for culvert crossings and 249 mg L^{-1} for improved fords, and lowest for the pipe + pole crossings at 145 mg L^{-1}. Sediment concentration increases of 221 mg L^{-1} for the temporary bridge crossings were similar to the overall mean increase of 217 mg L^{-1} for all crossings.

Thompson and colleagues' (1994, 1995, 1996) culvert installation and ford renovation studies illustrate how in-channel work can alter short-term streamwater sediment concentrations, and the bridge study by Thompson et al. (1994) illustrates the advantages of avoiding in-channel work. However, when in-channel work cannot be avoided, there are data, though extremely limited, that support the importance of controlling streamflow during the disturbance period. In a comparison of sediment generated from the installation of a culvert at two sites, where flow was diverted around construction, 0.2 lb of sediment was contributed to the stream. In contrast, where flow was not diverted or controlled in the construction area, 46 lb of sediment was contributed to the stream (U.S. Forest Service 1981).

In the longer term, culverts often result in the greatest changes to sediment inputs when compared with other types of crossing structures. For example, Tchir et al. (2004) found that 73 percent and 65 percent of crossings in two watersheds in Canada that were categorized as having at least moderate sediment inputs were culvert crossings. Sediment originated primarily from unstable soils where mass slumping occurred (e.g., in crossing fills) and from areas adjacent to crossings with exposed soil. Witmer et al. (2009) also identified poor crossing fill condition as a factor contributing to increased sedimentation risk at round culvert crossing structures compared to box culverts (i.e., rectangular reinforced concrete culverts) and bridges on unpaved roads in southeastern Alabama.

Norman (2006) reviewed seven 3-barrel pipe culverts, one 4-barrel pipe culvert, one box culvert, and two 3-barrel box culverts in Georgia for changes to channel geomorphology. Although there was substantial variation in the geomorphic conditions and changes present among sites, the only consistent changes observed across all types of culverts were shallower upstream thalweg depths and increased fining of streambed substrate downstream of culverts. Like Tchir et al. (2004), Norman (2006) attributed these changes to unstable banks around the crossings, as well as overwidening of the channel for culvert installation and undersized culvert diameters. Miller et al. (1997) evaluated 40 culverts, 21 bridges (temporary and permanent), and 9 fords that were between 2 and 28 yr old on first- and second-order streams. They examined geometric, sediment, habitat, and channel stability differences above and below the crossings, and found that most of the metrics were not affected by the crossings, but streambed fine sediment levels were elevated near the crossings due to the road and stream crossing fills. Wellman et al. (2000) measured the effects of construction or replacement for 18 culverts and 23 bridges on second- and third-order streams in Tennessee. Areas at and downstream of the culverts again had elevated percentages of fines (silt and clays) and greater depths of fines, whereas there was no evidence of elevated sediment accumulations at or downstream of bridges. Box culverts tended to be most susceptible to elevated sediment because their placement was not level with the streambeds, which resulted in scour pool formation and deposition of sediment near the culvert outflow. Perched pipe culverts that restricted high flows also resulted in scour pools downstream of their outlets (Merrill 2005).

Bouska (2008) and Bouska et al. (2010) found increasing entrenchment ratios, and hence, greater channel incision downstream of four of five vented fords, three of five box culverts, and one of two single, large corrugated pipe culverts in Kansas. The changes in entrenchment ratios were large enough to cause a shift in Rosgen stream classifications (Rosgen 1996). Spacing between riffles was increased by low-water crossings, but the influence was an upstream effect due to water backing up and causing inundation and submergence of upstream riffles. Mean riffle spacing was 8.6 bankfull widths upstream, compared to about 4.4 bankfull widths downstream (Bouska 2008, Bouska et al. 2010). Merrill (2005) also reported that channels downstream from all eight culvert and six bridge crossings in North Carolina had increased cross-sectional areas and they tended to have decreased hyporheic zone depths downstream of the crossings. The effects were greater for culverts than bridges. He attributed the changes observed for all types of crossings to channel constriction and disconnection of the channel from the floodplain caused by the presence of the culverts. Channel morphology changes were least for box culverts, apparently because they tended to be oversized compared to other crossing structures, including bridges on small streams.

Similarly, Norman (2006) reported that large box culverts had less effect on cross-sectional areas downstream of crossings than pipe culverts. In contrast, Bill (2005) found no significant changes in channel width or depth characteristics in cross sections measured downstream from three large pipe culvert crossings 1 yr after construction. Although changes downstream from box culverts may be limited, Bouska (2008) and Bouska et al. (2010) found that the average cross-sectional area within box culverts at bankfull discharge was five times greater than regional curves and regional control streams (41.2 m^2 versus 8.29 m^2, respectively).

Bridges often are considered to result in the least effect to fluvial geomorphology because they are believed to modify and restrict channel geometry the least (Blinn et al. 1998, Norman 2006). However, even small bridges without in-channel pilings can constrict the channel and disconnect it from the floodplain, as Merrill (2005) found. Additionally, the driving surface of the bridge can influence water quality, even if channel geometry is unaffected. Gaps in bridge floors can allow excess organic material, sediment, and other pollutants to fall into the water from vehicles and transported materials (Blinn et al. 1998, Tchir et al. 2004). This is particularly a concern with temporary bridges, which tend not to be solid to simplify their transport and installation (Tornatore 1995).

Bridges with in-channel supports typically show evidence of bed scour around pilings. Wellman et al. (2000) found scour around pilings of all of the 18 bridges they evaluated. Downstream changes in channel morphology also can result from the presence of bridges if they constrict the channel, and the effects can be long lasting. Gregory and Brookes (1983) evaluated local channel adjustments for four types of 18th- and 19th-century bridges in England; three of the four types caused channel constriction (Table 15). Increases in width-to-depth ratios were evident from bridges that constricted the channel, but where channel constriction did not occur and the channel bed was not hardened, the downstream channel was not changed by the bridge (Table 15). Channel adjustments associated with the three former types of bridges persisted downstream for as much as 20 bridge widths. Further analyses using maps and aerial photographic techniques of 15 other 18th- and 19th-century multiple- or single-arched bridges showed channel widening 1.14 to 2.78 times more than observed upstream of the crossings. Resulting changes in capacity were evident 8 to 390 m downstream. In many cases, the effects of lateral and bed scour were evident to the first major meander in the river downstream of the bridges.

Table 15.—Changes to channel geometry downstream of 18th- and 19th-century bridges in England (data from Gregory and Brookes 1983)

	Increase in downstream width–depth ratio		Distance change is observable downstream
Bridge type	**Average**	**Maximum**	**(Bridge or channel widths)[a]**
Bridges with constricted channel width, paved bottom	1.6	2.35	4–20
Bridges with constricted channel width, unpaved bottom	2.0	4.5	Up to 12
Bridges with 1.25-m-diameter culverts conveying water beneath the road	Data not provided, but significant increases were noted		Less than 20 but the actual values were not specified
Bridges with no channel constriction, unpaved bottom	~0	~0	~0

[a] Channel and bridge widths were used interchangeably in the paper, so bridge width is assumed to approximate channel width.

Changes to channel morphology from crossing installation or reconstruction appear to occur relatively rapidly; however, the few studies that include older crossing structures suggest that changes reach equilibrium soon after their installation, and although they do not continue to worsen over time they do persist as long as the constricting features remain in place. Wellman et al. (2000) found that the depth of sediment downstream was not related to the age of the associated culvert. Similarly, the analysis of the geomorphology of 18^{th}- and 19^{th}-century bridges in England (Gregory and Brookes 1983) indicated relatively small increases in downstream width-to-depth ratios relative to the age of the respective bridges (Table 15).

Constricting structures can result in scour and increased channel capacity, width-to-depth ratios, and incision, but channel widening and the installation of oversized crossing structures also can result in changes to sediment routing and channel morphology. Merrill (2005) reported that in-channel bars were formed during low flows downstream of oversized culverts. Channel widening caused the cross-sectional area also to be oversized, so normal sediment transport could not occur and deposition resulted.

Geomorphic and substrate changes tend to be expressed most near crossing structures. Wellman et al. (2000) observed decreasing sediment deposition with distance from culvert outlets. For smaller and intermediate streams, measurable changes were typically confined to within 50 m of the structure (Bouska 2008, Miller et al. 1997, Norman 2006, Wellman et al. 2000). Although crossings can alter sediment erosion and transport at the reach scale (Wargo and Weisman 2006), such effects were reported only for some of the larger rivers (30 to 60 m wide) examined by Gregory and Brookes (1983). Even for those rivers, most of the channel adjustments remained more localized and within 20 channel widths downstream of the bridges.

Turbidity and sediment increases within the water column can extend much farther downstream, but they typically are short-lived, such as during installation activities, during fording with heavy equipment, or during isolated storm events. Only one study was found where elevated sediment inputs from crossing installation were reported as resulting in visually observable accumulations of silt on the streambed surface downstream beyond the reach scale (Bill 2005). However, the sediment inputs were not sufficient to alter width-to-depth ratios locally or downstream of the installed culverts.

Stream Crossing Approaches

Virtually all of the sediment that is eroded at or delivered to a crossing from the road surface or ditch line enters the associated water body (Weaver and Hagans 2004). This type of sediment delivery occurs by movement through the crossing approaches, so their length and design can substantially influence sediment transport and delivery. For example, Thompson et al. (1994, 1995) found that better road drainage in the approaches to a bridge (broad-based dips at 20 m on one side and 50 m on the other side) resulted in about one-half the average sediment concentrations delivered to the receiving waters (38 mg L^{-1}) during construction compared to another bridge (66 mg L^{-1}) that had no drainage control in the approaches. Brown et al. (2015) used simulated rainfall events to compare mean sediment concentrations attributable to three surface treatments on six additional stream ford approaches in the southwestern Virginia Piedmont. The "low gravel" treatment (9.8 m of approach closest to the stream was graveled; 34 to 60 percent of the approach had gravel cover) resulted in a median total suspended sediment concentration of 1.1 g L^{-1}, which was not significantly different from the 0.82 g L^{-1} median concentration reported for the "high gravel" treatment (19.6 m of the approach length was graveled; 50 to 99 percent of the approach had gravel cover). Both gravel treatments resulted in significantly lower total suspended sediment concentrations than the ungraveled approaches, where the median concentration was 2.84 g L^{-1}.

Other sources of sediment delivery to the stream channel in approaches originate from crossing fills, and adjacent cutbanks and fillslopes (Lane and Sheridan 2002, Swift 1985). However, almost no data exist that quantify the importance or effectiveness of BMPs associated with these sources of inputs, either during crossing construction or in the longer term. The only applicable data found during this review are provided in Hamons (2007) and Stedman (2008), which both involve a single study in West Virginia. Sediment inputs from three crossings and their approaches were quantified during and for several years following the installation

Table 16.—Characteristics of three culvert crossings and their approaches on a newly constructed forest haul road in West Virginia (data from Stedman 2008)

Culvert crossing	Culvert diameter	Fill depth downstream	Road approach angle (left; right)	Sediment reaching the channel
	---------------- *m* ----------------		*degrees*	*kg*
1	1.52	7.4	15; 7.5	1,143.7
2	1.22	7.9	46; 42	25.8
3	0.91	4.5	39; 24	7.6

of culverts on a logging haul road. All of the crossings had deep fills (Table 16) and the approaches all had high, steep fillslopes. Most of the sediment that was delivered to the channel originated from the construction of the fillslopes in the approaches by mechanical additions (bulldozer pushing sediment into the streams) and from the approach fillslopes before they became vegetated.

However, the amount of sediment that reached the stream channels at each crossing during that time differed substantially (Table 16). This difference was attributed to the differences in the angle at which the road approached the stream channels. Sediment inputs attributable to 153 m of approaches to the stream crossing at angles of 8 to 15º (i.e., the road was nearly parallel and very close to the stream) exceeded the annual hillside contributions of sediment to that entire 33-ha watershed before road construction (Stedman 2008). The close proximity of the stream to the road made it virtually impossible to keep fillslope soil out of the channel during construction and before revegetation. Based on the results from the three crossings, the author recommended that approach angles of ≥25º should be used if fillslopes are constructed, or alternatively, full bench construction (i.e., in which no fillslopes are created) should be used where small approach angles cannot be avoided. However, these suggestions were not tested at the West Virginia location or elsewhere to determine their degree of effectiveness in this type of situation (see Chapter 2 for additional discussion of full bench construction).

Although the Stedman (2008) study showed that sediment inputs from approaches can be relatively short-term with fillslope revegetation, flow diversion at culvert crossings also can contribute to elevated, long-term sediment inputs. Diversion potential (Fig. 7) exists where the road and ditch system (if present) of the approaches slope away from the crossing on one or both approaches (Hagans et al. 1986). During high flows when culverts are overtopped, streamflow can become diverted down

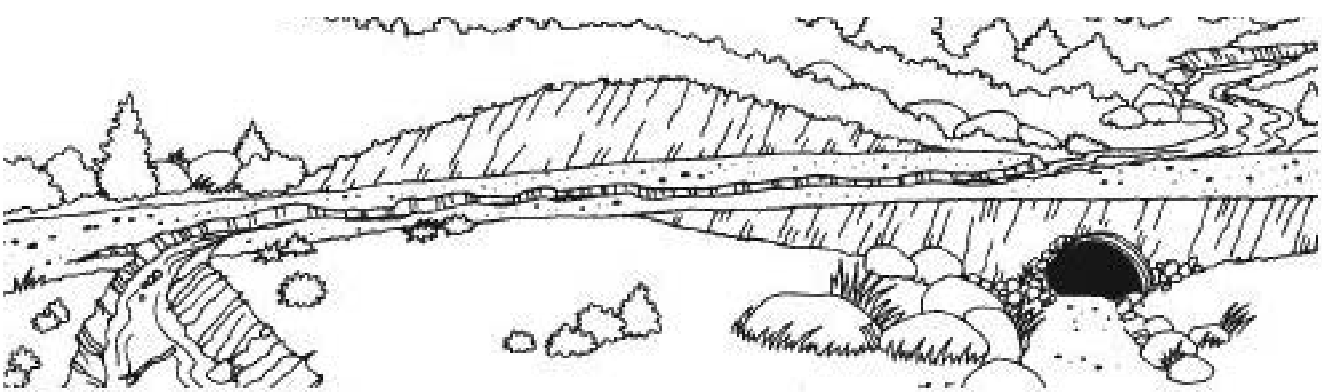

Figure 7.—Illustration of a stream crossing with diversion potential. If at least one of the approaches to the stream crossing slopes away from the crossing, overflowing water will flow down the road or ditch when the culvert is overtopped, creating erosion and damage to the road prism and surrounding hillside. From Keller and Sherar (2003).

the road or ditch line, forming new gullies on the road or hillside where constructed or natural drainage features turn water off the road. Hagans and Weaver (1987) estimated that gullies created by stream diversions were responsible for at least 40 percent of the total sediment production in the 419-km^2 Redwood Creek basin in California, and 89 percent of the gully erosion that followed harvesting in the lower portion of the watershed also resulted from streamflow diversions (Hagans et al. 1986). Hagans and colleagues recommended ensuring roads and trails are constructed without diversion potential, incorporating adequately sized culverts with debris filters, performing routine maintenance on roads and drainage features, and removing culverts and other crossings and their fills when the road or trail is no longer needed to reestablish an unobstructed channel. Again, however, there are no data to indicate how effective these suggestions are for long-term control of sediment inputs.

Wet Soil Crossings

Many road and crossing techniques are used to travel over wet soils (e.g., see Blinn et al. 1998, Mason 1990), yet there have been few tests of effectiveness for most techniques or few comparisons of alternative techniques. Rummer (1999) found that conventional crowned roads in bottomland hardwoods underwent sediment deposition during the flooding season, whereas those with a "zero profile" (i.e., trees were cut at the ground surface in the right of way, but root wads were not grubbed and no ditches were constructed) had net losses of sediment. However, the differences were explained in part by differences in flow direction but more by the speed of flood flows resulting from the variable water depths across the road prism due to its shape. The crowned road was oriented perpendicular to flow, so the ditches served as areas of deposition because there were slower velocities in the deeper waters present in the ditches. The zero-profile road was parallel to the direction of flood flows, and the velocities and water depths were about the same over the entire prism. As a result, erosion of the exposed soil occurred with no concomitant deposition within the prism.

One of the few studies comparing road surfacing effectiveness on sediment losses in wet soils in Georgia bottomland hardwoods showed no significant differences among native soil, 15 cm of gravel over geotextile, and seeded native soil during the flooding season (Rummer et al. 1997). However, a 6-cm thickness of gravel on native soil had significantly greater sediment yields than all the other treatments except native soil. The relatively high losses from the gravel were attributed to the lack of embeddedness into the roadbed because the gravel was applied to an exising road. As a result, the floodwaters transported some of the gravel, which contributed to the sediment load. For all of the surfacing treatments, sediment movement and deposition were confined to the area within the road right-of-way due to the low water velocities during flooding.

As these aforementioned studies suggest, sediment is not typically much of a concern for roads traversing wetlands and wet soils because slower flows and the greater roughness can keep sediment effectively contained near the area of displacement. Instead, the focus of roads in wetlands and on wet soils is on maintaining hydrologic function, as the integrity of wetlands and wet soils depends upon surface and subsurface hydrologic function (Blinn et al. 1998).

Hydrologic alteration associated with roads often occurs because of the development of ruts in wet soils, so most studies of wet soil crossing BMPs look at techniques to allow transportation while controlling the negative effects. Results of studies involving several types of surface BMPs to reduce rutting in wetlands are shown in Table 17. For those studies in which a control (no crossing mitigation) was employed, all of the techniques tested controlled rutting depth better than the control. Wooden mats and wooden pallets underlain by geotextiles resulted in similar reductions in rutting (Table 17), but wooden mats are generally considered a better alternative because they tend to cost less, are easier to assemble and install, and have less breakage and fewer repair needs than wooden pallets (Hislop 1996b).

Although none of the individual studies in Table 17 compared specific techniques (e.g., wooden mats or pallets, metal grating) with and without geotextile underlayment, comparisons of studies employing geotextiles (Hislop 1996a, Mason and Greenfield 1995) to the study by Blinn et al. (1998) where some of the same techniques were used without geotextiles suggest underlying geotextiles contribute substantially to

Table 17.—Rutting depths in wet soils using a variety of surfacing BMPs

Location/ Soil	Type and number of passes	Treatment	Rutting depth	Reference
			inches	
Florida silty sand	Loaded log truck, 300 passes	Geotextile beneath wooden pallets	0.5	Mason and Greenfield (1995)
		Control	6–10	
	Loaded log truck, 260 passes	Geotextile beneath deck span safety grating and expanded metal grating	0.5–1	Mason and Greenfield (1995)
		Control	12	
Florida silty sand	Loaded log truck, 240 passes	Geotextile beneath wooden pallets	1.5	Hislop (1996a)[a]
		Geotextile beneath wooden mats	1.5	
		Control	8	
	Loaded log truck, 75 passes	Geotextile beneath expanded metal grating	5	Hislop (1996a)[a]
		Control	15	
	Log truck, 30 loaded passes and 30 unloaded passes per day for 14 days	Geotextile beneath wooden pallets and wooden mats[b]	1–1.5	Hislop (1996b)
		Control	3.8–7.6	
Michigan deep black muck	Unloaded flatbed truck with a loader, 20 passes for mitigations, 1 pass for control	Wooden mat	12.5	Blinn et al. (1998)[a]
		Expanded metal grating	8	
		Tire mat	6.5	
		Wooden planks	4.5	
		Control	11.5	
Minnesota ponded histosol	Loaded forwarder, 20 passes	Geotextile beneath wooden mat	5.1	Blinn et al. (1998)[a]
		Geotextile beneath expanded metal grating	4.8	
		Geotextile beneath tire mat	21	
		Geotextile beneath wooden planks	6.8	
		Geotextile beneath PVC pipe mat	1.3	

[a] Reported as maximum rutting depth.

[b] Both techniques were used in series, but measurements were not separated between the two.

controlling rutting depth. The exception is for tire mats. The improvement observed with geotextiles may be because they help distribute loads more evenly and allow water to move through the fabric when under load while maintaining the soil below the fabric (Hislop 1996b, Mason and Greenfield 1995).

The effectiveness of tire mats has been inconsistent. In Michigan, Blinn et al. (1998) showed tire mats to be comparable to other methods when no geotextiles were used, but they were far inferior to other methods tested in Minnesota when used in combination with geotextile fabric (Table 17). Mason and Greenfield (1995) also reported problems with using tire mats made from recycled sidewalls (Terra Mat) on wet soils, but the soil disturbance came from equipment during placement of these heavy mats, rather than use of the mats.

After road use, removal of commonly used surfacing materials on wet or weak soils is generally relatively easy, but the removal of underlying geotextiles can be difficult. Because most are nonwoven materials (see the section on Effectiveness of Rolled Erosion Control Products in Chapter 6 for a description of nonwoven geotextiles), they tend to retain soil and water, which can make them too heavy to remove. To alleviate this problem, they can be installed in shorter lengths, rather than directly off a roll. This technique helps with removal, and still allows the material to be reused (Hislop 1996b).

Some wet soil crossing techniques involve application of wood chunks or wood fibers on the road surface or placed within the road subgrade. For these materials, there are concerns about water quality and aquatic health effects associated with the wood leachate and wood decomposition processes. The effects have not been examined in detail, and much of the information comes from sources not directly related to road investigations. Taylor et al. (1996a) reported that chemicals that are potentially toxic to aquatic organisms can be leached from trembling aspen (*Populus tremuloides*) wood, so care should be taken if these woods are used for road surfacing. They did not investigate the application on roads, however. Leachate from wood waste (mostly bark) disposed of in rock quarries near St. John Harbor on Zarembo Island, Alaska, was found to substantially reduce dissolved oxyen levels in nearby settling ponds and receiving streams to below 7 mg L^{-1}, with some values below 1 mg L^{-1} (Reed and Wolanek 1994). However, after the water was aerated by running through about 1,000 linear ft of small cascades in the stream, oxygen increased to acceptable levels (>7 mg L^{-1}). Sphagnum (*Sphagnum* spp.) moss in contact with the leachate died, but filamentous bacteria and slime mold growth appeared to benefit from the leachate effects.

The influences of mill-generated wood fiber and bark used as the principal subgrade material on a road in the Tongass National Forest in Alaska generally were found to be small, and in some cases beneficial to water quality (Wolanek 1995). Total organic carbon (TOC) and chemical oxygen demand (COD) levels (indicators of the influence of wood materials on dissolved oxygen concentrations) in leachate collected within the road increased to levels that exceeded 800 mg L^{-1} several times during the first 50 days after construction. These concentrations then exponentially declined. The influence on stream chemistry was examined by using water samples collected from two streams crossed by the road. Downstream (below the crossings) chemistries were adjusted using upstream (above the crossings) chemistries to determine the effects of the leachate. Dissolved oxygen was not affected, but pH was increased by 0.5 to 1.5 pH units. The receiving streams were naturally acidic (mean pH 5.5 and 5.8 prior to road construction), so the pH increases slightly buffered the streamwater. All of the water quality data were within limits of the state's water quality standards.

A temporary bridge installed over a perennial stream to reach a timber harvest area. (Photo by U.S. Forest Service, Northern Research Station.)

CHAPTER 6

Protecting Soil off the Driving Surface

During construction and sometimes during road maintenance activities, the width of disturbed soils can extend well beyond the width of the driving surface of the road. This is especially true on steeply sloping lands where road cutbanks are laid back to reduce their slopes to a more stable grade. Fillslope construction on the downhill side of roads also results in oversteepened areas with unconsolidated soil that is particularly susceptible to erosion (Edwards and Evans 2004, Megahan and King 2004, Rothwell 1978). Consequently, the need to provide soil cover on cutbanks, fillslopes, and drainage areas of forest roads has long been acknowledged (Hursh 1939). The focus of this chapter is restricted to protecting large areas of exposed soils, such as fillslopes and cutbanks; it excludes areas in which road drainage is or has become concentrated, as these are covered in Chapter 7. The following review considers two distinct types of practices to protect disturbed soils off the driving surface: soil conditioners applied to exposed soil and more conventional types of soil cover.

Soil conditioners traditionally have had more use in agriculture, especially tilled soils, but they have been used to some extent where soils have been disturbed at construction sites (including roads), open pit and strip mines, and landfills (Faucette et al. 2006; McLaughlin et al. 2009a, 2009b; Sojka et al. 2007; Soupir et al. 2004; Vacher et al. 2003). Soil conditioners also have been considered as a rehabilitation treatment to reduce erosion on burned forested sites (Davidson et al. 2009, MacDonald and Robichaud 2007, Wohlgemuth and Robichaud 2007).

The more conventional soil protection techniques described in this chapter are vegetation, mulches, rolled erosion control products (RECPs), and surficial slope stabilization techniques that primarily are aimed at controlling shallow mass failures from road construction and are particularly applied on fillslopes.

The effectiveness of mulches and RECPs is focused on controlling erosion only. Even though their application often includes the objective of restoring vegetation, the success of these materials at achieving that objective is not considered here. This is because vegetation establishment depends upon many other conditions that are extremely site-specific (e.g., seed selection and viability, soil nutrient levels, precipitation characteristics) and are independent of the soil cover treatment; often they are not described well enough to determine what factors contributed to or detracted from vegetative success.

Soil Conditioners

Soil conditioners are chemicals applied to soil to improve soil stability and aggregation, which in turn results in reductions in erosion and increases in infiltration (or reductions in surface runoff). Their use began in the 1940s and 1950s (Green and Stott 2001, Weeks and Colter 1952), when they were developed primarily for application on temporary roads and runways during wartime (Seybold 1994). It was not until the 1990s, however, that research began to suggest field use of soil conditioners could be effective and economical (Sojka et al. 2007).

The chemical composition of soil conditioners has evolved substantially since their initial development. Some of the first soil conditioners were water-soluble polymers of hydrolyzed polyacrylonitrile, vinyl acetate maleic acid, or calcium carboxylate (Trout et al. 1995, Weeks and Colter 1952). Today, soil conditioners primarily take the form of gypsum, less pure forms of other gypsiferous materials, and polyacrylamides (PAMs). Use of PAMs far exceeds that of other soil conditioners. A few other natural polymers, such as polysaccharides, also have been tested as erosion control agents (Agassi and Ben-Hur 1992, Lentz et al. 1992, Levy et al. 1992). They often do not perform as well as other conventional soil conditioners and they are more expensive (Sojka et al. 2007). Research is continuing into development of organic biopolymers derived from by-products of crop production and shellfish processing because they are considered to have more environmental benefits and to be more sustainable (Sojka et al. 2007).

Contemporary gypsiferous materials are by-products of air pollution control technologies at coal-burning power plants. These include compounds resulting from flue gas desulfurization and fluidized bed combustion (Norton et al. 1993). Until 1989, a gypsiferous compound known as phosphogypsum also was used commonly as a soil conditioner in the United States. Phosphogypsum is a by-product of the phosphate fertilizer industry (Norton et al. 1993) and is composed of high percentages (~97 percent) of calcium sulfate (Shainberg et al. 1990, Zhang and Miller 1996a). However, it is commonly contaminated with radon gas (Norton et al. 1993), so phosphogypsum now can be used in the United States only if its average radium-226 levels are less than 10 pCi g^{-1} (Florida Industrial and Phospate Research Institute 2010).

PAMs are water-soluble, synthetic organic polymers (Seybold 1994). Like naturally occurring polymers, such as humic substances and polysaccharides, PAMs provide soil stability (Shainberg et al. 1990), but they provide better erosion control than their natural counterparts (Shainberg and Levy 1994). PAMs are manufactured in a broad range of molecular weights, charge types, and charge densities (Barvenik 1994, Lentz and Sojka 1994, Shainberg and Levy 1994), so their formulations can be tailored to a wide variety of soil conditions and mineralogies (Lentz and Sojka 1994, Shainberg and Levy 1994). In general, however, PAM formulations that have been the most successful for erosion control, prevention of seal formation, and duration of effect are anionic with moderate to high charge densities and high molecular weights (Lentz et al. 1993, Seybold 1994, Shainberg and Levy 1994, Sojka and Lentz 1997, Trout and Ajwa 2001). Low, medium, and high charge densities are defined as <10 mol %, 10 to 30 mol %, and >30 mol %, respectively. Low, medium, high, and very high molecular weights of PAMs are defined as $<10^5$ g mol^{-1}, 10^5 to 10^6 g mol^{-1}, 1 to 5×10^6 g mol^{-1}, and $>5 \times 10^6$ g mol^{-1}, respectively (Barvenik 1994).

Crosslinked superabsorbent or gel-forming PAMs and cationic PAMs also are available commercially, but neither is suitable for erosion control. Crosslinked superabsorbent PAMs do not prevent erosion because they are not water soluble (Sojka and Lentz 1997) and they have been shown to seal soil pores and actually increase surface runoff and erosion due to their viscosity (Ajwa and Trout 2006). Cationic PAMs are toxic to aquatic organisms such as fish and macroinvertebrates (Barvenik 1994, Sojka and Lentz 1997).

In contrast, anionic PAMs show essentially no toxicity to fish (LC_{50} >100 mg L^{-1}) (Barvenik 1994); Seybold (1994) presents ranges of toxicities for fish and invertebrates. PAMs in any form (anionic, cationic, and nonionic) have low toxicity to humans and typically result in only slight dermal or eye irritations in small mammals, even at high concentrations (Stephens 1991). PAMs are also nontoxic to plants, though at very high levels in the soil (i.e., ≥5 percent of soil dry weight, which is orders of magnitude higher than used for erosion control) PAMs can result in phosphorus deficiencies in plants (Wallace et al. 1986).

Anionic PAMs that are manufactured for and used at concentrations suitable for soil conditioning and erosion control are not considered to pose an environmental or human health threat (Barvenik 1994, Leib et al. 2005, Seybold 1994). The primary environmental or health threats related to PAMs involve the amount of residual acrylamide monomer that remains behind from product synthesis and potential contamination of groundwater (Abdelmagid and Tabatabai 1982, Seybold 1994).

The residual monomer is a human neurotoxin (Leib et al. 2005, Seybold 1994). However, all of the products available in the United States for erosion control are formulated to meet U.S. EPA and Food and Drug Administration standards for other common PAM applications, including wastewater treatment, potable water treatment, and food processing and packaging (Barvenik 1994, Sokja and Lentz 1997). Consequently, PAMs may contain no more than 0.05 percent monomer (Barvenik 1994), and actual residual monomer levels are usually less than 0.0002 percent (Seybold 1994). The free monomer that exists from the manufacturing process is metabolized in biologically active environments fairly quickly following application to the soil and does not accumulate in the soil; it has a half-life of hours (Barvenik 1994, Seybold 1994). During breakdown, there is no release of free acrylamide monomers from the synthesized PAM itself (Barvenik 1994). Breakdown of PAM requires several weeks and occurs as a result of mechanical disturbances (e.g., rainfall impact, wetting/drying cycles, tilling), chemical and biological hydrolysis, and ultraviolet radiation from sunlight (Barvenik 1994, Seybold 1994, Sojka and Lentz 1997). Biological hydrolysis of PAMs by soil microorganisms produces ammonium that may then undergo further oxidation to nitrite and nitrate (Abdelmagid and Tabatabai 1982, Seybold 1994).

Anionic PAMs are a potential source of groundwater pollution because they are highly water soluble, and have low soil adsorption potentials (Seybold 1994) in the absence of cations in the soil or application solution. If PAM reaches water before breakdown and hydrolyzation, its degradation in water requires longer than in soil—typically 100 to 700 h (Seybold 1994). However, PAMs are generally applied at relatively low rates only to the soil surface, and they tend not to penetrate more than a few millimeters or centimeters below the surface (Lentz et al. 1995, Mitchell 1986, Seybold 1994). The exceptions to this are if they are applied to dry, cracked soil, in which case they may penetrate more deeply (Malik et al. 1991); this soil condition would be rare in forests.

Contemporary soil conditioners all work in similar ways, though the physical or chemical mechanisms involved may differ. They interact with the clay fraction of soil, and they depend upon cations to flocculate clays and to form cation bridges (Agassi and Ben-Hur 1991, 1992; Chaudhari and Flanagan 1998; Shainberg et al. 1990). That is, anionic conditioners can attach to multiple cations that have become attached to negatively charged clays, thereby forming bridges across clay particles. Cation bridging is most effective with multivalent, rather than monovalent, cations (Levy et al. 1992, Shainberg et al. 1990). Monovalent cations can participate in cation bridging, but these have weaker bonds than multivalent cations (Laird 1997). The conditioner itself may contain and release the bridging cation (e.g., Ca^{2+} in gypsum and gypsiferous materials). In the case of anionic polymers, an additional cation source must be applied with the polymer (e.g., combining the polymer with a gypsiferous material), or cations must be present in sufficiently high concentrations in the soil or in the water used to prepare the application solution (Green and Stott 2001, Orts et al. 2007, Shainberg et al. 1990, Warrington et al. 1989). Depending upon the chemistry of the local water source, tap water may have sufficient cations present to contribute to cation bridging (Shainberg et al. 1990, Smith et al. 1990). Applications should not depend upon subsequent rain events to provide the necessary amount of cations; Wohlgemuth and Robichaud (2007) found the cation loads in rainfall were insufficient to allow PAMs to be effective.

Cation bridges help strengthen aggregate bonding of clay particles by reducing repulsion of the net negative charges of clay particles (Ben-Hur 1994, Seybold 1994, Sojka and Lentz 1997). Bridging maintains large soil aggregates and pore integrity at the surface. In turn, bridging leads to other positive outcomes, including increases in soil wettability (Jańczuk et al. 1991), increased resistance to slaking during rapid wetting (Mitchell 1986), stabilization of shrinking and swelling in clays (Malik et al. 1991), increased resistance to aggregate breakdown and detachment by surface flows and rainfall impact (Bryan 1992, Helalia and Letey 1989, Shainberg et al. 1992, Terry and Nelson 1986, Trout and Ajwa 2001), increased soil infiltration, prevention of soil crusts, and reduction of soil sealing (Davidson et al. 2009, Kazman et al. 1983, Lentz and Sojka 1994, Seybold 1994, Sojka and Lentz 1996, Trout and Ajwa 2001). Thus, the physical conditions of soil, such as structure and permeability, are not improved by soil conditioners; instead they are stabilized against breakdown (Lentz and Sojka 1994, Trout and Ajwa 2001). In addition, conditioners and associated cations act as flocculants (Shaviv et al. 1988, Sojka and Lentz 1996), so fine dispersed clay particles flowing in runoff can clump together and settle out as the aggregate mass increases (Sojka and Lentz 1996, Warrington et al. 1989). Flocculation further helps combat soil sealing because the particles that settle out are larger and ineffective at filling fine voids among soil aggregates (Shainberg and Levy 1994).

Because soil conditioners depend upon flocculation and stabilization of clay particles, they are most effective in soils that have moderate to higher clay content. Indeed, most studies and applications of soil conditioners have focused on finer textured soils (Trout and Ajwa 2001), probably because these soils are most erodible and need stabilization. PAM effectiveness is most likely when the soil clay content is at least 30 percent (Davidson et al.

2009). PAM effectiveness in fine textured soils is further improved by using a formulation with an appropriate charge density (Green et al. 2000, Levy and Agassi 1995); effectiveness has been most commonly associated with charge densities in the mid-teens to mid-30 mol % range (Green et al. 2000, Malik et al. 1991, Michaels and Morelos 1955, Peterson et al. 2002, Sojka et al. 2007). The charge density is the percent hydrolysis of the polymer, which describes the percentage of hydroxyl groups that have been substituted for acrylamide groups in the polymer (Green et al. 2000). High charge densities (e.g., 40 mol %) can result in repulsion of the polymer and the clay particles, thereby reducing adsorption onto clays and effectiveness of PAMs (Ben-Hur et al. 1992, Green et al. 2000).

In a silty clay soil in Indiana, wet and dry applications of PAM both were found to be effective at controlling erosion and runoff (Peterson et al. 2002). Total runoff was reduced by 62 to 76 percent by using wet applications, and sediment yields were reduced 93 to 98 percent. Dry applications were slightly less effective at controlling both variables. Aase et al. (1998) examined PAM use at low concentrations (1, 2, 4, and 6 kg ha^{-1}) on a silt loam. For concentrations greater than 2 kg ha^{-1}, runoff and soil loss were each reduced 70 percent on average. PAM at high concentrations (10, 20, and 40 kg ha^{-1}) in a clay loam retained infiltration at rates substantially higher than clay loam with no soil conditioner (Shainberg et al. 1990). However, the addition of phosphogypsum with PAM resulted in the retention of even higher infiltration rates. The phosphogypsum dissolved over time, and provided a continued source of cations that allowed the soil clay to remain flocculated, thereby helping to retain soil bridging with the PAMs. Phosphogypsum also was found to be an effective soil conditioner in a clay loam in the Midwest (Norton et al. 1993). After 2 h of rainfall (74 mm), total soil yield on a 5-percent sloped plot was only one-third of the amount from an untreated control. Levy et al. (1991) recorded 50- to 70-percent reductions in runoff in a clay loam using PAM; this was the case for plots with and without a crop (cotton, *Gossypium hirsutum*) present. Reductions in erosion were more pronounced for bare soil (62 to 83 percent less than controls) than when the crop was present (22 to 52 percent less than controls), presumably because the crop contributed to soil stabilization.

Soupir et al. (2004) saw 82-percent reductions to sediment yields from a clay soil soon after the application of a dry form of PAM. A wet formulation applied at the manufacturer's recommended rate resulted in only a 40-percent reduction. Higher and lower application rates were not as effective, and resulted in only 28- and 33-percent sediment reductions compared to no treatment.

In coarser textured soils, the effectiveness of soil conditioners is not guaranteed due to the lower clay percentages present. This is especially the case for soils with high sand contents like sandy loams. Separate studies by Trout and Ajwa (2001) and Ajwa and Trout (2006) reported reductions in infiltration with the use of PAM compared to controls. Gypsiferous soil conditioners may result in more desirable effects with sandy loams. Final infiltration rates following phosphogypsum application were almost triple that of controls in a study by Warrington et al. (1989), and they were double that of controls in a study by Miller (1987) using two sandy loams in the Southeast. In the latter study, cumulative and average infiltration rates significantly increased compared to untreated soils. Erosion was reduced by 30 percent and 50 percent in the two soils, and there was essentially no loss of clay particles in runoff; however, silt losses were greater than in the controls. Zhang and Miller (1996a) found that a gypsum application allowed a sandy loam to have a 26-percent higher final infiltration rate after one rain event than with no soil conditioner. In an even coarser soil (loamy sand), erosion was not reduced with the use of PAM after a wildfire in southern California (Wohlgemuth and Robichaud 2007).

To improve the potential for PAM effectiveness in soils with high percentages of sand, the molecular weight of the PAM formulation is key (Green et al. 2000, Levy and Agassi 1995). It is important because it is directly related to the polymer length: the longer the polymer, the greater the molecular weight (Green et al. 2000). Polymers with lower molecular weights may not have chain lengths that are sufficiently long to bind together the relatively small number of clay particles present (Levy and Agassi 1995); polymers with very high molecular weights may be too large to penetrate soil voids (Barraclough and Nye 1979). The molecular weight is not important in clayey soils because the clay particles are close enough together that even short polymer lengths can bind them (Green et al. 2000, Levy and Agassi 1995). However, using a PAM formulation of high molecular weight with a sandy loam still does not guarantee its effectiveness. Although Green et al. (2000) found that a high molecular weight PAM formulation (12×10^6 g mol^{-1}) was effective, molecular weights below (10^6 and 6×10^6 g mol^{-1}) and above (18×10^6 g mol^{-1}) that formulation were not effective. Likewise, similar molecular weights (15×10^6 g mol^{-1} and 12×10^6 g mol^{-1} to 15×10^6 g mol^{-1}) were not effective in studies by Trout and Ajwa (2001) and Ajwa and Trout (2006), respectively.

Medium-textured soils, such as silt loams and loams, tend to be responsive to soil conditioners and seem to be less sensitive to the molecular weight of the PAM formulation. Green et al. (2000) studied PAM formulations across a range of molecular weights (10^6, 6×10^6, 12×10^6, and 18×10^6 g mol^{-1}) and charge densities (0, 20, 30, and 40 mol %) in a silt loam, and all formulations except the one with the highest molecular weight and the highest charge density showed improvements in soil stabilization. Lentz et al. (1992) applied PAM with an even higher molecular weight (10^7 g mol^{-1}) to a silt loam and found that the mean sediment loss was reduced 97 percent compared to untreated soil, but the charge density of this formulation was in the more typical range of 20 mol %. Lentz and Sojka (1994) and Trout et al. (1995) both used PAMs with 15×10^6 g mol^{-1} molecular weights and 18-percent hydrolysis. Lentz and Sojka found soil losses were reduced by an average of 94 percent and infiltration was increased by an average of 15 percent. Trout et al. (1995) found similar reductions in erosion (ranging from 85 to 99 percent, average 70 percent) and 30-percent increases in average infiltration rates.

Gypsiferous compounds alone and in combination with PAM also are effective in medium-textured soils. Lepore et al. (2009) measured soil losses from silt loam soil using six soil conditioners (lime, gypsum, PAM-coated lime, PAM-coated gypsum, gypsum + PAM applied separately, and lime + PAM applied separately) and bare soil. All soil conditioners significantly reduced soil losses compared to untreated soil. Treatments involving lime resulted in less erosion than their gypsum counterparts, but the differences among the conditioners were not significant. Adding PAM to lime and gypsum also improved soil stability more than lime or gypsum alone, and coating the lime or gypsum with PAM was more effective than adding the PAM separately. Soil losses from PAM-coated lime and lime + PAM were 75 and 53 percent, respectively, less than lime alone, and 83 and 67 percent, respectively, less than soil with no conditioner treatments. PAM-coated gypsum and gypsum + PAM also had significantly lower soil erosion than gypsum alone (63 and 49 percent, respectively). The two treatments reduced erosion by 69 and 58 percent, respectively, compared to bare soil. The PAM coating on the lime and gypsum reduced the dissolution rates of both, and this may explain why PAM-coated treatments were more effective than applying lime or gypsum alone or separately with PAM.

In another study with a silt loam, Norton et al. (1993) measured lower soil losses using a nearly pure gypsum conditioner from fluidized bed combustion and using a gypsiferous material from flue gas desulfurization than with no treatment. A second gypsiferous material from flue gas desulfurization resulted in poorer erosion control than no treatment. Infiltration results paralleled those results. Phosphogypsum also improved erosion control and infiltration compared to the control. Miller (1987) found phosphogypsum applied to a silt loam increased infiltration rates and resulted in significant increases in cumulative and average infiltration rates compared to untreated soil. Soil losses also were half of those for the untreated soil. Shainberg et al. (1990) applied PAM to a loam soil and the final infiltration and cumulative infiltration rates were three and four times higher, respectively, than without treatment. Combining PAM with a phosphogypsum application increased the final infiltration rates to levels that were 10 times greater than the controls.

Many of the studies of soil conditioner effectiveness, including those just described, have been performed for agricultural uses. This raises the question of whether soil conditioners have application to nonagricultural soils, and for the purposes of this review, steep soils. These types of applications of soil conditioners have not been examined as extensively. But there are some studies that illustrate the potential for soil conditioner use on disturbed soils of forest road prisms, including some studies directly from road construction, albeit generally highway construction. However, the results across all studies have been mixed. On a 30-percent gradient, Norton et al. (1993) found all of the four treatments tested (phosphogypsum, two gypsiferous materials from flue gas desulfurization, and almost pure gypsum conditioner from fluidized bed combustion) were ineffective at reducing erosion or increasing infiltration, even though three of them had been effective on a 5-percent slope. On the steeper gradient, sediment losses rose 3.5 to 4.8 times and were similar to the control. Warrington et al. (1989) found that phosphogypsum was less effective on steeper slopes than gentle slopes, but outperformed controls with no treatments. On 5-percent slopes, erosion from phosphogypsum treatments was 60 percent less than controls. At a 25-percent slope, the erosion rates doubled with the soil conditioner, but that same increase in slope resulted in a sevenfold increase in soil loss from untreated soil.

After a wildfire in Utah, Davidson et al. (2009) studied the effect of aerially applied granular PAM pellets (made with recycled paper) on soil losses over 3 yr. The slopes to which PAM was applied were 33-percent grade. They found that PAM resulted in significantly lower erosion rates than control soils. But because the controls were located on only 16-percent slopes, the impact of the improvement by the soil conditioner is probably underrepresented by the statistical comparison. MacDonald and Robichaud (2007) tested two PAM treatments in a burn area emergency rehabilitation treatment in the Colorado Front Range. They reported somewhat mediocre results because PAM became preferentially bound to the ash from the fire, rather than the soil. One treatment reduced erosion during two storm events and the other treatment was ineffective.

Flanagan et al. (2002b) examined PAM and PAM + gypsum applications on a 35-percent sloped highway cutbank in a clay loam and a 45-percent sloped reclaimed landfill in a silt loam in Indiana. The responses were compared to untreated controls following individual storm events over summer seasons. For individual storm events on the highway cutbank, erosion was reduced from 44 to 100 percent by PAM alone and from 26 to 100 percent for PAM + gypsum. Cumulative sediment reductions across the entire study period compared to no treatment were 54 and 45 percent for PAM and PAM + gypsum, respectively; the results from these two treatments were not significantly different from one another.

Additionally, reductions in runoff from the two treatments compared to the controls were similar across storms and ranged from 25 to 91 percent for PAM and 36 to 90 percent for PAM + gypsum. Cumulative runoff for the entire study was reduced by an average of 33 percent for both treatments. At the landfill, soil yields were 39 to 100 percent less than the control for PAM and 44 to 100 percent less than the control for PAM + gypsum. Percent cumulative sediment reductions from the landfill were similar to the less steep cutbanks; they were 40 and 53 percent less than untreated soil for PAM and PAM + gypsum treatments, respectively, and again the results from the two were not significantly different from each other. However, PAM + gypsum was more effective at controlling erosion during very large events, even though runoff was similar. In the largest two storms, PAM + gypsum reduced sediment by 58 percent and 85 percent compared to the control, whereas soil losses from the PAM treatments during those events were not significantly different from using no soil conditioner. At the landfill, runoff for individual storms also was not significantly different between the two treatments, but PAM reduced runoff 30 to 55 percent and PAM + gypsum reduced runoff 27 to 74 percent compared to the controls. PAM + gypsum had significantly lower cumulative runoff (28 percent less) than the controls or PAM.

In another study by Flanagan et al. (2002a) in which a silt loam soil at 32-percent grade was subjected to repeated simulated rainfalls following PAM and PAM + gypsum treatments, total runoff over the entire experiment was reduced by 40 percent and 52 percent, respectively, by the two treatments compared to a control. After the first event, runoff and sediment losses were reduced by more than 90 percent by both treatments. Even when subjected to a 25-yr rain event, sediment yields were 60 percent and 77 percent less than the control for PAM and PAM + gypsum, respectively. Total sediment losses following a cumulative rainfall that equated to more than a 100-yr event were 83 percent and 91 percent less than the control for PAM and PAM + gypsum, respectively. On a sandy loam with a 48-percent slope, phosphogypsum significantly reduced runoff and erosion compared to untreated soils (Agassi and Ben-Hur 1991). Runoff was reduced from 23 to 31 percent by the conditioner versus the control, and erosion was reduced two to three times that from the control. However, erosion increased substantially as the plot length increased on these steep slopes. As the length increased from 1.5 m to 10 m, soil loss increased by 6.4 times, but sediment yields still were less than from untreated soil.

In another study by Agassi and Ben-Hur (1992), PAM + phosphogypsum and phosphogypsum + polysaccharide conditioners were compared to a control on earthen dikes in Israel ranging in slope from 33 to 60 percent. Over a winter season, they observed that the soil losses from both treatments were 10 times less than with no treatments. Soil erosion from the two treatments was not significantly different from each other. On 58-percent slopes, a variety of application rates and approaches using both dry and wet applications of PAM were successful in reducing erosion from 75 to 100 percent in a sandy clay soil and a loam soil when subjected to simulated rainfall (Wallace and Wallace 1986). On a 50-percent fillslope at a highway construction site in the North Carolina Piedmont on sandy clay loam soil material, two PAM formulations each were applied at two different (albeit low compared to most literature) rates (0.8 and 1.5 kg ha^{-1} for formulation 1, and 5.2, and 10.5 kg ha^{-1} for formulation 2) (Hayes et al. 2005). Runoff, turbidity, and total sediment losses after seven storms were not significantly different from the control. However, average turbidity and total sediment yield did decrease with increasing polymer application rates. On a 20-percent cutslope of nearly the same soil material, the same PAM formulations and application rates resulted in no differences in runoff or sediment loss compared to the control, but turbidity was reduced.

Soil conditioners have been applied in several ways: furrow advance, overhead sprinkler/irrigation, spraying under high pressure, dry broadcasting or spreading over the surface, and dry or wet aerial application (e.g., Agassi and Ben-Hur 1991; Flanagan et al. 2002a, 2002b; Fox and Bryan 1992; Green and Stott 2001; MacDonald and Robichaud 2007; Mitchell 1986; Terry and Nelson 1986). Based on review of the literature, aerial applications have been undertaken only for wildfire restoration when a substantial area has been severely burned and the potential for catastrophic impact to watershed values is high, and for agricultural purposes when the soil conditioner was spread over an entire field by crop dusting planes (e.g., MacDonald and Robichaud 2007, Wallace and Wallace 1986). Aerial application is not likely to be used to meet road BMP objectives, so it is not discussed further. The others are described further because the technique used to apply a soil conditioner influences its efficacy and the needed application rate.

Furrow advance is directed at reducing erosion from irrigation waters and improving infiltration in agricultural furrows. The actual treatment of the furrow, referred to as "furrow advance," typically is performed by mixing the soil conditioner with irrigation water or injecting it into irrigation water as the water is released into the dry furrow. If cation levels in the conditioner, irrigation water, or the soil are not believed to be sufficient to allow cation bridging, some form of cation is mixed in with the conditioner. This release continues until runoff through the entire furrow is achieved (Sojka and Lentz 1996). Once runoff occurs, irrigation is stopped. Effective erosion control via furrow irrigation illustrates the applicability of soil conditioners to controlling erosion in concentrated flow. Applications of PAM and other soil conditioners to concentrated flow associated with roads and construction (e.g., within swales and in combination with erosion barriers) are considered in Chapter 7. However, additional discussion of furrow irrigation is included in this chapter to keep the fundamental information about soil conditioners in one place.

All other soil conditioner application techniques are aimed at treating larger land areas, with the intent to protect soil from rainfall impact and surface runoff. As with furrow advance, cations also are applied, if needed, in dry or wet form; this can be done before or at the same time as application of the conditioner. Basic information about wet and dry applications is provided in subsequent paragraphs, but Lentz et al. (1995) provide detailed information about some of the pros and cons related to dry and wet applications of PAMs. Although these are directed at agricultural uses, many of the points also are applicable to any field use of PAM.

In general, wet applications of soil conditioners have been found to be more effective than dry applications (Cook and Nelson 1986, Peterson et al. 2002, Shaviv et al. 1987). Applications of solutions should be made to dry soil. Studies in which soil conditioners have been applied to wet soil have shown inferior infiltration and erosion control benefits compared to applications to dry soil (Roa-Espinosa et al. 1999, Shainberg et al. 1990). Dry soil conditioners can be applied to slightly moist soil, or the soil can be lightly moistened before or after application; however, only enough water to moisten the first millimeter or two of soil should be used to dissolve the conditioner and enhance binding (Wallace and Wallace 1986). On dry soils, dry soil conditioners may not become adequately activated before the first precipitation event (Peterson et al. 2002). Following wet or dry application, a period that allows the soil to dry fully before it is contacted by rainfall (or irrigation water) is critical for maximizing conditioner effectiveness (El-Morsy et al. 1991, Shainberg et al. 1990). The drying or curing period allows the soil conditioner to fully adsorb onto soil clays, and allows PAM to become irreversibly adsorbed to the soil so desorption is negligible (Nadler et al. 1992, Shainberg et al. 1990).

Furrow advance treatments allow more dilute applications than sprinkler or spray treatments (Sojka et al. 2007). This is partly because furrow advance involves treatment of only a relatively small surface area (Lentz and Sojka 1994). Typical furrow advance loads for PAM have been in the range of 0.5 to 1.3 kg ha^{-1} (Lentz and Sojka 1994, Lentz et al. 1992, Sojka and Lentz 1997), though rates above 0.7 kg ha^{-1} had substantially smaller soil losses than rates below 0.7 kg ha^{-1} (Lentz and Sojka 1994). Overhead sprinkling systems have shown success with PAM with rates as low as 2 to 4 kg ha^{-1} (Aase et al. 1998, Bjorneberg et al. 2003), but overhead delivery is more commonly successful in the 20 to 70 kg ha^{-1} range (Agassi and Ben-Hur 1992, Flanagan et al. 1992, Fox and Bryan 1992, Levy et al. 1991, Norton 1992, Wallace and Wallace 1986). Spray application rates of PAM tend to be in this same range. Zhang and Miller (1996b) used rates of 15 kg ha^{-1} and 30 kg ha^{-1}, Green et al. (2000) used 20 kg ha^{-1}, and Flanagan et al. (2002a, 2002b) used a rate of 80 kg ha^{-1} for treating soils on steep slopes. However, Petersen et al. (2007) applied a specially designed PAM emulsion using a hand sprayer and found that only 5 kg ha^{-1} effectively curtailed erosion. Dry application rates of PAMs are less than what has been used for wet applications, 5.6 kg ha^{-1}, but the actual granular forms including the carriers result in much larger total application rates (e.g., 224 kg ha^{-1} and 280 kg ha^{-1}) (Lepore et al. 2009). It should be noted that manufacturer-recommended rates for PAM are sometimes much lower than the typical rates that have been tested in the literature, or states have set maximum loads for application to construction sites; sometimes these low rates have resulted in poor erosion control or runoff control (Hayes et al. 2005, Soupir et al. 2004).

In contrast to PAM treatments, gypsiferous and cationic soil conditioners typically require much higher application rates. Zhang and Miller (1996a) used 5 tonne of phosphogypsum ha^{-1}, and Flanagan et al. (1997) applied fluidized bed combustion bottom ash at a rate of 5 tonne ha^{-1}. Miller (1987) also applied phosphogypsum at 5 tonne ha^{-1}. Lepore et al. (2009) applied gypsum and lime each at 280 kg ha^{-1}.

Except for furrow irrigation, where vegetation is undesirable in the furrows of agricultural fields, the purpose of soil conditioners in field applications typically is to provide soil aggregation and protection against soil sealing until vegetation becomes established (Vacher et al. 2003). Once vegetative cover reaches 50 to 60 percent, it is considered to be capable of taking over the role of soil stabilization (Carroll et al. 2000, Loch 2000). For disturbed soils on forest road prisms, this means that soil conditioners must remain effective long enough for revegetation to occur; otherwise retreatment with the soil conditioner will be necessary. Agricultural studies often recommend reapplication of conditioners repeatedly throughout the growing season (Sojka and Lentz 1997), but that is because repeated mechanical disturbances to the soil are expected during such activities as tilling and pesticide application.

Most soil conditioner studies have been performed for agricultural soils, so the available data on the expected life of soil conditioners are limited. There are, however, a few studies for PAM that show it has been effective over nearly season-long periods. Fox and Bryan (1992) found PAM was effective for 6 wk and Petersen et al. (2007) found it effective for at least 10 wk. Petersen et al. (2007) found runoff was reduced by an average of 100 percent after 2 days following application of PAM, by 59 percent after 3 wk, and by 55 percent after 10 wk. Erosion control was even better, with soil losses reduced by 100, 80, and 74 percent for those three respective time periods. Shainberg et al. (1990) found PAM effective throughout at least 3 wk of study. A study by Soupir et al. (2004) showed relatively poor results for separate wet and dry applications of PAM after only about 4 wk for a construction site on clay soil in Virginia. At 1 month, the wet and dry treatments, respectively, had only 25 percent and 11 percent lower sediment yields than the control. However, they used an application rate of only 3.36 kg ha^{-1}, which was below typical spray application rates, though it was the manufacturer-recommended rates for this formulation.

Soil Cover Protection

Dozens, if not hundreds, of types of soil cover materials have been used for controlling erosion. Some of these are exclusive to specific land uses, such as stubble mulches in agricultural fields (Freebairn et al. 1986, Tibke 1988), that have little applicability to road-related soil disturbances. Consequently, the types of soil covers that are reviewed here emphasize techniques that have been used in forest road settings or have application to forested hillsides where roads may be constructed.

The major types of soil cover applicable to roads fall into four categories (Table 18). Vegetation and nonorganic mulches generally are meant to provide long-term soil cover. Organic mulches and rolled erosion control products are designed to have short-term (e.g., several months) to moderate-term (several seasons to a few years) lifetimes. Organic mulches and rolled erosion control products typically are used in conjunction with vegetative seeding, as these products are designed to allow vegetation to grow through them as they biodegrade or photodegrade. A few types of rolled erosion control products, such as plastic sheeting, are not biodegradable, but these rarely have utility in forest applications off the road surface. Inorganic mulches are nonbiodegradable, though some types may be photodegradable or thermo-degradable (e.g., plastics) over relatively long time periods. Inorganic mulches are designed to stabilize soil by physically keeping it in place, and revegetation typically is not an objective; it may even be an undesirable outcome with their use. However, inorganic mulches are not always incompatible with revegetation. Meyer et al. (1972) observed very good grass establishment with stone mulch, and it exceeded revegetation with wood chips, which seemed to have negatively affected the carbon/nitrogen ratio of the soil.

From the perspective of erosion control there is one primary objective common to all soil cover materials—preventing or reducing raindrop impact. Raindrop energy has been shown to be the most important process by which soil particle detachment occurs (Young and Wiersma 1973); it exceeds detachment even by concentrated flow. This is because of the great amount of energy that raindrops transfer to the soil (Berglund 1976). For example, a 30-minute thunderstorm in the Midwest is capable of resulting in an impact of cumulative dead weight of more than 100 ton ac^{-1} and expending >2,000,000 ft-lb ac^{-1} (Wischmeier and Smith 1958). Hudson (1957) demonstrated the importance of controlling raindrop impact by covering a plot of exposed soil with mosquito gauze. The gauze absorbed the raindrop energy, yet allowed most of the rain water to reach the soil surface. Soil loss from the gauze-protected plot was 127 times less and runoff was 13 times less than from a paired unprotected, bare soil plot. In laboratory studies, reducing raindrop energy by 89 percent resulted in nearly equivalent reductions (90 to 94 percent) in

Table 18.—Types of soil covers generally applicable to disturbed soils off the driving surface

Type of soil cover	Examples
Vegetation	Grasses and herbaceous plants primarily through seeding, tree seedlings less commonly
Organic mulches	Wheat straw, rice straw, barley straw, hay, other agricultural plant residues, wood chips, wood strips, sawdust, needles/leaves, fruit hulls, tree bark, shredded paper/cellulose
Inorganic mulches	Crushed or washed stone, geologic waste materials, shredded/recycled rubber or plastic materials
Rolled erosion control products	Excelsior blankets, jute netting, coir mats, and other types of erosion control mats or blankets; geocells; and various types of synthetic sheets, filaments, and meshes

total soil loss from three soils, whereas eliminating rills (and therefore rill erosion) resulted in only 17- to 37-percent reductions in erosion (Young and Wiersma 1973). Controlling raindrop impact reduces soil particle dislodgement, and associated compaction and sealing of the soil surface, which help reduce surface runoff and associated erosion (Bhatt and Khera 2006, Elwell and Stocking 1976, Lattanzi et al. 1974, Meyer and Mannering 1963, Moss 1991, Quinton et al. 1997, Young and Wiersma 1973).

An additional benefit common to many soil cover materials is that they provide roughness at the soil surface (Berglund 1976, Bhatt and Khera 2006) and sometimes tortuosity (Meyer et al. 1970). Surface roughness and tortuosity slow the movement of interrill and rill runoff and dissipate its energy, further contributing to erosion control (Lattanzi et al. 1974). With organic soil covers, decaying material also returns organic matter to the soil, which improves soil structure, forms stronger soil aggregates, and maintains high soil infiltration capacities—all of which decrease erosion (Baver 1956, Berglund 1976, Bhattacharyya et al. 2007, Loch 2000, Tengbeh 1993, Traoré et al. 2000). In 44 agricultural soils, organic matter content was the most important variable for explaining infiltration and controlling runoff (Wischmeier and Mannering 1965). A variety of soil cover materials improve growing conditions by providing shade that reduces soil surface temperatures, conserves moisture, and encourages fast vegetative growth (Berglund 1976, Bhatt and Khera 2006, McKee et al. 1964, Sheldon and Bradshaw 1977). However, some soil cover materials can increase the temperature at the soil surface (Anderson et al. 1996, Benoit et al. 1986), which can improve or degrade growing conditions, depending upon other variables, such as available moisture (Barkley et al. 1965, Dudeck et al. 1970) and time of year (Benoit et al. 1986, Kohnke and Werkhoven 1963).

The belowground biological processes that plants carry out provide additional benefits that are not available from other types of soil cover. Root growth improves soil structure and maintains high soil permeability, thereby reducing runoff and interrill and rill erosion (Baver 1956, Elwell and Stocking 1976). Plant roots also hold soil in place physically due to the presence of fibrous roots and chemically by binding soil particles with root exudates (De Baets et al. 2006, Gyssels and Poesen 2003, Tengbeh 1993, Traoré et al. 2000). Tengbeh (1993) found soil shear strength (binding between roots and soil) in soil containing fibrous grass roots was at least 500 percent greater than that of bare soil. While aboveground vegetation provides protection against raindrop splash and interrill erosion, roots are at least equally important in controlling rill and gully erosion (Gyssels et al. 2005).

Effectiveness of Vegetation

Of all the types of soil cover used in rehabilitation and stabilization of disturbed soils that are associated with forest roads, the most common is revegetation through seeding of grasses and herbaceous species. Except when expensive native seed mixtures are used to revegetate, seeding is the most cost-effective measure to provide soil cover and erosion control (Miles et al. 1989, Robichaud 2005). Established vegetation (e.g., tree seedlings) can be planted (Megahan 1974a), but tree planting is usually inappropriate for road prism rehabilitation. It is preferable to keep tall vegetation away from the road to maximize visibility for safety and to provide light during the day to reduce moisture and control rutting on the road surface (see Chapter 3).

Unlike soil cover techniques that provide immediate protection, soil erosion control from seeding is ineffective until vegetation becomes sufficiently established. During the initial period of vegetation establishment, soil losses from seeded and untreated, bare soil can be comparable, both in amounts and patterns of loss. For example, Bethlahmy and Kidd (1966) measured nearly similar amounts of soil produced from untreated (no soil cover) and seed + fertilized plots (Table 19). Most of the soil losses occurred during the first 100 days, but they declined exponentially (Fig. 8). Similar exponential declines in sediment yields from seeded and bare soils have been reported by others through the first growing season (e.g., Burroughs and King 1989, Dyrness 1975, Harrison 2011, Megahan et al. 2001, Orr 1970, Wade 2010), and even over periods as short as a week (Burroughs et al. 1984c). This exponential pattern of sediment losses is common when no initial soil cover is available because large amounts of loosened soil from ground disturbance

Table 19.—Results from select erosion studies

Reference	Treatment	Soil loss, as specified	Duration	Field plot/ Simulation
		Average (*ton ac*$^{-1}$)		
Barnett et al. (1967)	Bare soil	96.57	2 simulated storm events: (1) 1.3 inches of rain in 30 min (2) 2.7 inches in 60 min	Field (highway construction site)
	Checkered dams (mulch punched into soil)	44.48		
	Surface mulch and asphalt	31.54		
	Mulch mixed in surface	27.14		
	Mulch mixed in surface + asphalt	27.59		
	Whisker dams (mulch pressed into soil)	9.88		
	Surface mulch	11.20		
		Average (of maximum and minimum sediment yields per event) (*kg ha*$^{-1}$)		
Benik et al. (2003)	Disc-anchored straw mulch	108	5 single storm events	Field (hillslope of a sedimentation basin at a highway construction site)
		99		
		54		
		2,125		
		732		
	Straw blanket	16		
		10		
		9		
		103		
		79		
	Straw/coconut blanket	18		
		17		
		7		
		228		
		179		
	Wood fiber blanket	29		
		15		
		19		
		258		
		157		
	Bare treatment	763		
		788		
		189		
		5,438		
		4,296		

(continued)

Table 19.—Results from select erosion studies

Reference	Treatment	Soil loss, as specified	Duration	Field plot/ Simulation
		Total (*1,000 lb ac^{-1}*)		
Bethlahmy and Kidd (1996)	Control	84.2	322 days, 20.40 inches of precipitation	Field (newly constructed road fillslope)
	Seed, fertilizer (2 plots)	104.7		
		89.4		
	Seed, fertilizer, straw mulch			
	Contour furrows, seed, straw mulch, fertilizer, holes	11.9		
	Seed, fertilizer, straw mulch with asphalt emulsion	36.0		
	Seed, fertilizer, straw mulch, netting			
	Paper netting	1.1		
	Jute netting	0		
	Chicken-wire netting	0.4		
		Average (of total sediment yield for two plots) (*g*)		
Bhattacharyya et al. (2008) (see also Bhattacharyya et al. 2007)	Bare	85.79	March 25, 2002–May 10, 2004	Field (hillslope)
	Grass	13.04		
	Buffer (of borassus mats)	37.16		
	Covered (completely by borassus mats)	31.22		
		Average eroded solids (3 plots) (*kg*)		
Buchanan et al. (2002)	Control (bare)	3.90	8 storm events over 130 days	Field (steep construction slope; 55%)
	Small wood chips (6.4–13 mm)	3.03		
	Mixture of wood chip sizes (13–25 mm)	0.53		
	Large wood chips (>25 mm)	0.86		
		Average (*lbs 1,000 ft^{-2}*)		
Burroughs et al. (1984c)	Bare soil (2 plots)	116.72 (4 reps)	Simulated rainfall for 23–30 min, rate ~2 inches h^{-1}	Field (road fillslope)
		210.65 (1 rep)		
	Vegetated (2 plots)	0.52 (4 reps)		
		1.14 (3 reps)		
	Curlex mulch (1 plot)	10.50 (4 reps)		
	Filter windrow (1 plot)	15.33 (4 reps)		
		Cumulative averages (*cm*) (+ indicates soil accumulation)		
Carr and Ballard (1980)	Hydroseed + fertilizer + binder	+ 1.1	7 months	Field (forest roadside slopes)
	Hydroseed+fertilizer+binder+mulch	+ 1.3		
	Control	2.3		

(continued)

Table 19.—Results from select erosion studies

Reference	Treatment	Soil loss, as specified	Duration	Field plot/ Simulation
		Total (*g* m^{-2})		
Döring et al. (2005)	Control (bare soil)	1,606	1 h simulated rainfall with total amount of 73 mm	Field (agricultural; 8% slope)
	Chopped winter wheat straw (mean length 58 mm)			
	1.25 tonne ha^{-1}	31		
	2.5 tonne ha^{-1}	42		
	5 tonne ha^{-1}	26		
	Unchopped winter wheat straw (avg. length 75 mm; 25% of pieces exceeding 100 mm length)			
	2.5 tonne ha^{-1}	133		
		Cumulative averages (*inches*) (+ indicates soil accumulation)		
Dyrness (1970) (see also Dyrness 1975)	Control	0.83	1 yr following construction; 2 replicates	Field (forest road cutslopes)
		0.84		
	Mulch (wheat straw mulch, 2 ton ac^{-1})	+ 0.05		
		0.07		
	Blue River District mixture (25 lb ac^{-1}, no mulch)	0.77		
		0.31		
	Oregon highway mixture (40 lb ac^{-1} with straw mulch 2 ton ac^{-1})	0.20		
		0.23		
	Experimental mixture 1 (43 lb ac^{-1} with straw mulch 2 ton ac^{-1})	0.65		
		+ 0.17		
	Experimental mixture 2 (43 lb ac^{-1} with straw mulch 2 ton ac^{-1})	0.07		
		+ 0.11		
		Average (*kg*)		
Foltz and Dooley (2003)	Bare		3 simulated repetitions: (1) rainfall only (2) rainfall + 1st inflow (3) rainfall + 2nd inflow	Simulation (30% slope)
	Rainfall only	2.0		
	Rainfall + 1st inflow	4.2		
	Rainfall + 2nd inflow	29.4		
	Straw mulch (“agricultural straw”)			
	Rainfall only	0		
	Rainfall + 1st inflow	0.03		
	Rainfall + 2nd inflow	0.53		
	Wide wood strand (16 mm wide)			
	Rainfall only	0		
	Rainfall + 1st inflow	0		
	Rainfall + 2nd inflow	0.39		
	Narrow wood strand (4 mm wide)			
	Rainfall only	0		
	Rainfall + 1st inflow	0.20		
	Rainfall + 2nd inflow	0.61		

(continued)

Table 19.—Results from select erosion studies

Reference	Treatment	Soil loss, as specified	Duration	Field plot/ Simulation
		Average (*kg*)		
Grace et al. (1996) (see also Grace et al. 1998)	Bare (control)		6-month study period 3 replications for each treatment (i.e., 3 cutslope, 3 fillslope)	Field (forest roads)
	Cutslope	24,769		
	Fillslope	20,204		
	Erosion mat (wood excelsior)			
	Cutslope	345		
	Fillslope	2,358		
	Native grass (11.23 kg ha^{-1})			
	Cutslope	8,352		
	Fillslope	3,866		
	Exotic grass (11.23–28.07 kg ha^{-1})			
	Cutslope	1,742		
	Fillslope	2,669		
		Average (*g* m^{-2})		
Grushecky et al. (2009)	Control		2 yr	Field (downslope terminus of newly constructed skid roads; average slope 18%)
	Year 1	174.3		
	Year 2	615.5		
	Fiber mat			
	Year 1	34.8		
	Year 2	62.3		
		Terminal erosion rate (*kg* ha^{-1} s^{-1})		
Jennings and Jarrett (1985)	Fallow soil	4.32	1 10-min simulated rainfall	Simulation
	Straw (2.24 Mg ha^{-1}, 35% coverage)	1.75		
	Straw (8.96 Mg ha^{-1}, 98% coverage)	0.27		
	Small bark (22.4 Mg ha^{-1}, 85% coverage)	0.78		
	Large bark (22.4 Mg ha^{-1}, 40% coverage)	1.89		
	Large bark (56 Mg ha^{-1}, 98% coverage)	0.54		
	Jute net (1 layer, 50% coverage)	0.42		
	Burlap (1 layer, 60% coverage)	0.83		
	5-mm-wide cellulose with netting (1 layer, 50% coverage)	1.61		
	Small rocks (10 mm thick, 30 mm diameter, 212 Mg ha^{-1}, 80% coverage)	2.10		
	Large rocks (25 mm thick, 100 mm diameter, 506 Mg ha^{-1}, 70% coverage)	1.38		

(continued)

Table 19.—Results from select erosion studies

Reference	Treatment	Soil loss, as specified	Duration	Field plot/ Simulation
		Average (*ton* ac^{-1})		
Kill and Foote (1971)	Straw + an asphalt emulsion tack	19.3	About 1 month (3.7 inches of precipitation) 5 replicates	Field (highway roadside)
	Short-fibered wood pulp mulch (paper pulp)	59.3		
	Short-fibered wood pulp mulch (whole pulp bolt)	62.5		
	Short-fibered wood pulp mulch (using Douglas-fir)	40.4		
	Long-fiber green wood excelsior fiber	15.3		
		Average erosion rate (*kg* min^{-1} m^{-1})		
Kramer and Meyer (1969)	Simulated straw mulch, 4% slope		30 min 2 replicates	Simulation (100 ft length)
	Control	0.35		
	0.125 ton ac^{-1}	0.32		
	0.5 ton ac^{-1}	0.16		
	Simulated straw mulch, 10% slope			
	Control	17.53		
	0.125 ton ac^{-1}	12.26		
	0.5 ton ac^{-1}	11.41		
		Average (*g* m^{-2})		
Lattanzi et al. (1974)	Wheat straw mulch, 2% slope		60 min 2 replicates	Simulation (0.37 m^2)
	Control	951		
	0.5 tonne ha^{-1}	602		
	2 tonne ha^{-1}	244		
	8 tonne ha^{-1}	7		
	Wheat straw mulch, 20% slope			
	Control	2,142		
	0.5 tonne ha^{-1}	1,246		
	2 tonne ha^{-1}	485		
	8 tonne ha^{-1}	2		
		Total (*tonne* ha^{-1})		
Lekha (2004)	Control	151.1	1 yr	Field (hillslope)
	Coir net geotextile	7.75		
		Average at peak discharge (*mg* L^{-1})		
Lemly (1982)	Low-intensity rainfall event (1.5 cm h^{-1})		60 min	Field (highway cutslopes; 30% slope)
	Control	118		
	Tacked straw	48		
	Jute netting	40		
	Mulch blanket	26		
	Wood chips	22		
	Excelsior blanket	21		

(continued)

Table 19.—Results from select erosion studies

Reference	Treatment	Soil loss, as specified	Duration	Field plot/ Simulation
	High-intensity rainfall event (4 cm h^{-1})		20 min	
	Control	720		
	Tacked straw	507		
	Jute netting	489		
	Mulch blanket	316		
	Wood chips	308		
	Excelsior blanket	300		
		Total (*ton ac^{-1}*)		
Mannering and Meyer (1963)	Wheat straw mulch		Simulated storm events over 3 days with a total of 6.25 inches of precipitation	Field (agricultural; 5% slope)
	0 (control) ton ac^{-1}	12.42		
	0.25 ton ac^{-1}	3.23		
	0.5 ton ac^{-1}	1.42		
	1 ton ac^{-1}	0.30		
	2 ton ac^{-1}	0		
	4 ton ac^{-1}	0		
		Average annual erosion (*ton mi^{-2} day^{-1}*)		
Megahan (1974a, estimated from Fig. 8 of that paper)	Control		3 yr	Field (road fillslopes)
	1970	11.6		
	1971	3.4		
	1972	14.5		
	Trees without mulch			
	1970	6.1		
	1971	2.4		
	1972	7.2		
	Straw mulch with netting			
	1970	0.3		
	1971	0.2		
	1972	1.1		
		Average (*tonne ha^{-1} yr^{-1}*)		
Megahan et al. (2001)	Control	16.4	4 yr	Field (road cutslopes)
	Dry seed alluvial soil	1.7		
	Hydroseed + mulch	7.1		
	Dry seed shallow soil + bedrock	7.8		
	Terrace + hydroseed + mulch	9.8		
		Total soil loss (*ton ac^{-1}*)		
Meyer et al. (1970)	Wheat straw mulch applied		3 simulated storm events over 2 days	Field (agricultural; 15% slope)
	0 (control) ton ac^{-1}	27.8		
	0.25 ton ac^{-1}	9.0		
	0.5 ton ac^{-1}	8.7		
	1 ton ac^{-1}	5.1		
	2 ton ac^{-1}	1.1		
	4 ton ac^{-1}	0.7		

(continued)

Table 19.—Results from select erosion studies

Reference	Treatment	Soil loss, as specified	Duration	Field plot/ Simulation
		Total (*ton ac⁻¹*)		
Meyer et al. (1972)	No mulch (control)	39.6	5 inches of simulated rain, at an intensity of 2.5 inches h^{-1}	Field (side slope of a borrow pit that had been dug during highway construction; 20% slope steepness; 35 ft slope length)
	Straw (wheat straw mulch, chopper-blower, 2.3 ton ac^{-1})	12.1		
	Stone (crushed limestone ranging from 0.25 to 1.5 inches diameter)			
	15 ton ac^{-1}	25.6		
	60 ton ac^{-1}	11.4		
	135 ton ac^{-1}	3.5		
	240 ton ac^{-1}	<2		
	375 ton ac^{-1}	<2		
	Gravel (washed road gravel ranging from 0.25 to 1.5 inches diameter)			
	70 ton ac^{-1}	14.7		
	Wood chips			
	2 ton ac^{-1}	27.1		
	4 ton ac^{-1}	8.5		
	7 ton ac^{-1}	5.5		
	12 ton ac^{-1}	<2		
	25 ton ac^{-1}	<2		
	Portland cement	32.7		
		Average erosion rate (*tonne* ha^{-1})		
Mitchell et al. (2003)	Geotextile net	0.08	April 10, 1995– April 15, 1996 (446 mm precipitation)	Field (hillslope)
	Geotextile mat	0.09		
	Grass	3.45		
	Bare soil	7.47		
		Average (*g*)		
Rickson (2006)	RECP 'Geojute'	29	10 min 3 replicates	Simulation
	RECP 'Geojute' fine	30		
	RECP 'Enviromat'	149		
	RECP 'Enkamat' surface laid	133		
	RECP 'Enkamat' buried	100		
	RECP 'Tensarmat'	167		
	RECP 'Bachbettgwebe'	68		
	Control	197		
Robichaud et al. (2010)	Various mulch treatment effectiveness results in Appendix C of cited paper			Field (hillslopes; primarily wildfire-burned areas)

(continued)

Table 19.—Results from select erosion studies

Reference	Treatment	Soil loss, as specified	Duration	Field plot/ Simulation
		Average (*kg ha^{-1}*)		
Robichaud et al. (2013)	Control		2 yr	Field (hillslope)
	Year 0 (postfire)	697		
	Year 1	104		
	Year 2	5.2		
	Straw mulch			
	Year 0 (postfire)	60		
	Year 1	22		
	Year 2	1.3		
	Wood shred mulch			
	Year 0 (postfire)	77		
	Year 1	29		
	Year 2	2.0		
		Average total (3 sites) (*kg day^{-1} ha^{-1}*)		
Rothwell (1983)	Unmulched (control)	6,033	4 months	Field (stream crossings)
	Mulched (brush mulch/logging debris)	1,033		
		Average erosion rate (*g m^{-2} h^{-1} mm precip^{-1}*)		
Sutherland and Ziegler (1997)	RECP 'Futerra'	0.93	Simulated rainfall events over 6 wk	Field (hillslope)
	RECP 'C125'	2.10		
	RECP 'Curlex I'	5.05		
	RECP 'P300'	5.12		
	RECP 'TerraJute'	7.86		
	RECP 'Geojute'	9.98		
	RECP 'SC150BN'	10.77		
	RECP 'BioD-Mat 70'	16.02		
	RECP 'BioD-Mat 40'	33.70		
	RECP 'PEC-MAT'	41.16		
	Control (bare)	134.90		
		Average (*Mg ha^{-1} yr^{-1}*)		
Wade (2010)	Control	137.7	May 2009–July 2010	Field (skid trails)
	Seed (grass seed, 300 kg ha^{-1})	31.5		
	Hardwood slash	8.9		
	Pine slash	5.9		
	Straw mulch	3.0		

(continued)

Table 19.—Results from select erosion studies

Reference	Treatment	Soil loss, as specified	Duration	Field plot/ Simulation
		Average annual (*Mg* ha^{-1} yr^{-1})		
Wagenbrenner et al. (2006)	Control		4 yr	Field (hillslope)
	2000 (8 plots)	>6.2		
	2001 (12 plots)	>9.5		
	2002 (12 plots)	1.2		
	2003 (12 plots)	0.7		
	Seeding (grass seeding, 34 kg ha^{-1})			
	2000 (1 plots)	>3.9		
	2001 (4 plots)	12.0		
	2002 (4 plots)	1.2		
	2003 (4 plots)	0.3		
	"Old"[a] mulch (weed-free wheat straw)			
	2000 (3 plots)	8.8		
	2001 (4 plots)	0.5		
	2002 (4 plots)	0.02		
	2003 (4 plots)	0.001		
	"New"[a] mulch (weed-free wheat straw)			
	2000 (0 plots)	---		
	2001 (3 plots)	0.02		
	2002 (3 plots)	0.006		
	2003 (3 plots)	0.000		
	"Old"[a] contour felling			
	2000 (4 plots)	>5.8		
	2001 (4 plots)	>5.7		
	2002 (4 plots)	0.03		
	2003 (4 plots)	0.02		
	"New"[a] contour felling			
	2000 (0 plots)	---		
	2001 (7 plots)	2.8		
	2002 (7 plots)	0.2		
	2003 (7 plots)	0.07		
		Annual accumulation (*ton* ac^{-1})		
Wollum (1962)	Bare (control)		3 yr	Field (road cutbank)
	Year 1	12.7		
	Grassed			
	Year 2	4.2		
	Year 3	2.3		

[a]The terms "old" and "new" in this case refer to pre- and post-storm, which occurred Aug. 16, 2000. Plots installed before that date are referred to as "old" and after that date as "new."

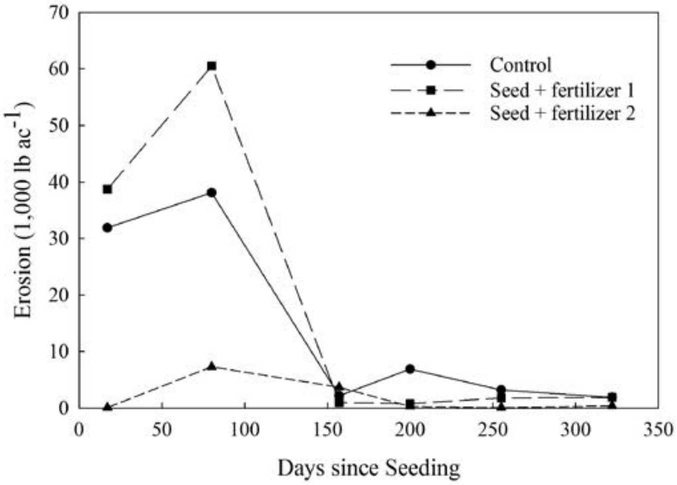

Figure 8.—Soil erosion loads measured from plots on a steep road fillslope in Idaho. The control plot (●) was untreated. One plot (■) was contour furrowed before grass seeding and fertilizing. One plot (▲) was treated with a soil emulsion before and after grass seeding and fertilizing. Data are from Bethlahmy and Kidd (1966).

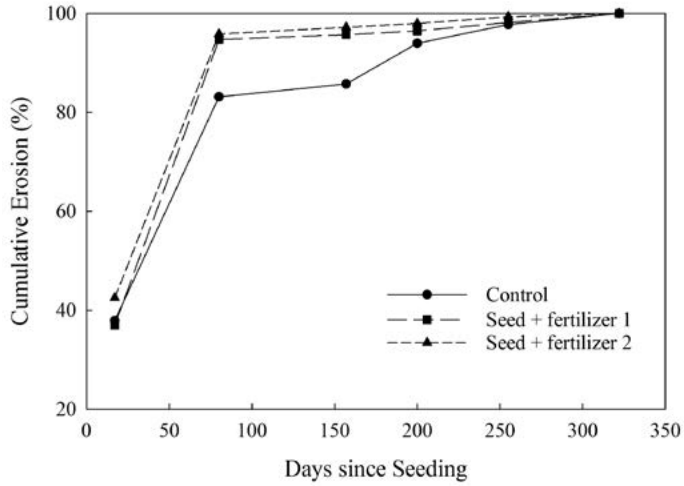

Figure 9.—Erosion loads from the temporal data shown in Figure 8 expressed as cumulative erosion rates.

are easily displaced by gravity and water (and wind in some situations). Once fines are eroded away, coarser materials are left behind at the soil surface (Megahan 1974b, Ollesch and Vacca 2002) because these are more resistant to displacement and transport (Hairsine and Rose 1992). This is often referred to as “armoring” or creation of an erosion pavement (Megahan 1974b).

As vegetation begins to become established, differences in soil losses between unseeded (and otherwise unprotected) and seeded soil become apparent. Erosion from bare soil continues to increase for a longer time than from seeded soil, so sediment yields typically take longer to reach the “minimum” constant values shown in Figure 8. This difference is illustrated by using the same data graphed in Figure 8 but by plotting them as percent cumulative erosion over time (Fig. 9). About 95 percent of total erosion occurred within the first 100 days on the seeded plots, but cumulative erosion did not reach 95 percent for another ~150 days, approximately day 255, on the unseeded plots (Bethlahmy and Kidd 1966). Grace (2002a) reported substantial cumulative increases in erosion until about day 700 from unseeded/unprotected fillslopes and from 900 to 1,000 days on unseeded/unprotected cutbanks.

The greatest benefits from revegetation occur over the long term because erosion can be returned to approximately pre-disturbance levels (Burroughs et al. 1984c). Four years after grass planting on roadside slopes, Grace (2002a) found erosion was at only 0.05 tonne ha^{-1} yr^{-1}, compared to an unseeded control, which continued to have erosion rates that were 80 times greater (4 tonne ha^{-1} yr^{-1}). Cycles of elevated erosion can continue, and may become more chronic for bare soil or soil that has had poor revegetation success. For example, following an exponential decline in erosion (similar to that found in Figure 8) during the first summer of soil disturbance on unseeded plots, Wade (2010) found high erosion rates (10 tonne ha^{-1}) were reinitiated in spring, whereas plots that had been seeded and revegetated that same summer had only small erosion increases during the subsequent winter and spring (2 tonne ha^{-1}). The reinitiation of soil erosion is attributable to freeze/thaw and wet/dry cycles in winter or spring (or other wet seasons), which loosen soil particles and make them available for erosion (Diseker and McGinnis 1967, Hipps et al. 1990, Putthacharoen et al. 1998, Römkens and Wang 1987, Shiel et al. 1988, Tisdall et al. 1978, Vandaele and Poesen 1995). In the presence of vegetation, the amount of soil that is loosened is less, and much of it is trapped by aboveground plant parts, so it is not lost to erosion (Berglund 1976). Carr and Ballard (1980) found net sediment accumulations resulting from vegetation on fillslopes and cutbanks in British Columbia.

A central question about the influence of vegetation is: "How much vegetative cover is necessary to protect soil?" This question has been examined from the perspective of both erosion and runoff, as sediment loss cannot occur without both detachment and transport. However, research on both variables has resulted in similar answers. Almost universally, vegetative cover in the range of about 50 to 75 percent is required to effectively control runoff or erosion, or both (Gifford 1985, Gutierrez and Hernandez 1996, Lang 1979, Loch 2000, Moreno-de las Heras et al. 2009, Orr 1970, Packer 1951, Quinton et al. 1997, Snelder and Bryan 1995), though erosion control often is achieved on the lower end of that range. If litter is present (i.e., on soil not newly disturbed), it also contributes to ground cover, so the 50- to 75-percent cover range may include vegetation plus litter, rather than just vegetation (e.g., Gifford 1985, Orr 1970, Pannkuk and Robichaud 2003, Robichaud et al. 2000). Generally the cover in these studies has been provided by low-growing plants, such as grasses, herbs, shrubs, or even agricultural plants, but from the perspective of erosion control the species present have been far less important than the total amount of ground cover (Gifford 1985, Grace 2002a, Packer 1951). Even on a landscape scale, restoration of ground cover or revegetation of forest trees to a level of about 75 percent has resulted in restoration of erosion rates to levels similar to those before large area disturbance or land conversion (Bailey and Copeland 1961, Vanacker et al. 2007).

Considering that different studies involved different types of soil (e.g., texture and structure, bulk density, antecedent moisture conditions), different experimental settings (e.g., field versus laboratory, slope, size of plot, vegetative species), and different precipitation characteristics (e.g., frequency, intensity, duration, quantity, simulated versus actual rain events), the cover percentages that have been found to keep erosion low are in good agreement. Even gully erosion has been shown to be stopped with a vegetative cover of just over 50 percent (Rey 2003). Heavy fern growth (near 100-percent cover) in a rill in South China also was successful at reducing erosion by an order of magnitude during a single monitored storm compared to a rill with no vegetation due to the greater roughness, higher infiltration, and greater shear strength of soil in the vegetated rill (Woo et al. 1997). However, the effectiveness of vegetation at controlling rill and gully erosion may depend upon the source of concentrated flow, the frequency of flow, and discharge levels. Grasses and herbaceous plants typically are not well suited for controlling erosion via concentrated flow from road drainage features that can carry large amounts of water on a frequent recurring basis (see Chapter 7 for further discussion).

Just as higher percentages of ground cover show good erosion control, ground cover below 30 to 40 percent shows consistently poor erosion control. Ground cover less than 30 percent (including litter) on grazed lands in Rhodesia was associated with much greater soil losses than denser ground cover (Elwell and Stocking 1974, 1976). After a wildfire, cover of 39 percent on forested hillsides with slopes averaging about 15 percent had erosion rates that were similar to hillsides with no remaining cover (Groen and Woods 2008). In northern Mexico, mean sediment concentrations were significantly higher where grass cover was less than 30 percent than where cover was greater than 60 percent during the growing season (Gutierrez and Hernandez 1996). Grass cover of only 35 percent in the Lake Tahoe Basin along the border of California and Nevada was not effective at controlling erosion (Grismer and Hogan 2005). The increases in erosion that are associated with less-dense vegetation apparently occur when individual bare areas begin to come in contact with one another and allow unimpeded runoff to occur (Lang 1979).

There is broad agreement in the literature from research studies across many land types and ecosystems showing that vegetation provides a high degree of protection against soil erosion when coverage is adequate (Descheemaeker et al. 2006, Elwell and Stocking 1976, Gutierrez and Hernandez 1996, Lang 1979, Loch 2000, Moreno-de las Heras et al. 2009, Quinton et al. 1997, Rey 2003). At almost any scale, the importance of vegetation to erosion control cannot be overstated (Tibke 1988). This is particularly true in the long term as vegetation is usually self-sustaining if not subjected to extreme disturbances, which makes it extremely cost-effective.

Ironically, despite vegetation's effectiveness as a long-term approach, additional techniques are needed to control erosion in the short term immediately after seeding; otherwise erosion will not be appreciably different than from bare soil (e.g., Fig. 8). This is the period for which most mulches and rolled erosion control products are designed (Ingold and Thomson 1986, Rickson 2006, Smets et al. 2008).

Effectiveness of Mulch

A large variety of mulches have been tested for use, but in most applied field applications, the type of mulch that is used depends chiefly upon availability, cost, and the land use to which it is applied (Morgan and Rickson 1995). Grain and hay mulches are the most common mulches used where revegetation is the goal. Grain mulches can be derived from many plant species including oats (*Avena* spp.), rice (*Oryza* spp.), or barley (*Hordeum* spp.) (Jennings and Jarrett 1985, Robichaud 2005), but wheat (*Triticum* spp.) straw probably has been the dominant type for road applications (Foltz and Dooley 2003). Rice straw is being used increasingly because it is believed that its use will result in the spread of fewer invasive weed species. This is because rice straw is a wetland plant, so if the straw is used in areas away from water bodies and hydric soils, rice seeds and any associated weed seeds that also presumably require those wet conditions will not be able to survive (Beyers 2004). In contrast, hay is a mixture of all the plant species present in the field during cutting; consequently, it can contain much greater amounts of weed seeds than straw mulch (Cornell University Weed Ecology and Management Laboratory, n.d.). Old straw has been shown to be less effective at controlling erosion than new straw (Grismer and Hogan 2005), presumably because decay has already begun.

Other types of commonly tested mulches associated with forest roads include wood mulches (bark chips, wood strands, wood shreds, pulp wastes) and stone or rock. Because these are free of weed seeds, there has been some interest in shifting more toward some of these types of mulches (e.g., wood materials) and away from straws and hay (Yanosek et al. 2006). Many kinds of these mulches are available, but most of the available varieties are used for landscaping purposes. Therefore, large-scale erosion control with them has been fairly uncommon and limited to only a few major types. Asphalt emulsions alone or as tackifiers mixed with some types of mulches, and cement (Dudeck et al. 1967, 1970; King 1979, 1984; McKee et al. 1964; Meyer et al. 1972) also have been used as mulches, though these are expensive, and result in inconsistent erosion control compared to many other types of mulches (Table 19). Asphalt also can have a negative effect on vegetation establishment (Dudeck et al. 1970), which can be long-lasting, so little current research involves asphalt tackifiers.

All types of mulches have been shown to provide better erosion control than leaving the soil bare (Table 19). Even small applications of mulch measurably reduce soil losses (Mannering and Meyer 1963, Meyer et al. 1970). Mulch provides soil cover, so it is not surprising that most types of mulch typically are most effective in the same soil coverage ranges as vegetation (50 to 75 percent) (Foltz and Copeland 2009, Foltz and Wagenbrenner 2010). Additional soil coverage results in diminishing rates of erosion control. Given the propensity for rill formation on poorly protected soils, mulch applied more evenly across the entire area of exposed soil provides more effective erosion control than mulch applied in strips along the contour or along the lower one-third of plots (Bhatt and Khera 2006). However, other variables also come into play in determining its effectiveness, especially beyond the short term. These variables include the type and character of mulch used, slope steepness and length, soil texture, and pattern of application to the soil (Bhatt and Khera 2006, Foltz and Copeland 2009).

The type and characteristics of a mulch determine its shape, size, weight, and ability to absorb moisture. These characteristics influence its coverage (per unit weight), its ability to remain in place, and its ability to deter runoff, particularly rill flow. Chopped and unchopped straw provide a good example of how the shape and size of straw affect its coverage. About twice as much unchopped (long) straw was needed to achieve 90-percent soil cover as chopped straw (4.43 tonne ha^{-1} versus 2.16 tonne ha^{-1}) (Döring et al. 2005). Straw loses its structural integrity and tends to split longitudinally when chopped into shorter-length pieces; consequently, chopped straw approximately doubles its effective coverage for a given weight of material compared to whole, long straw. Similarly, a much greater mass of large wood chips (16.54 tonne ha^{-1}) was needed to achieve 80-percent soil cover than mixed-sized chips (10.25 tonne ha^{-1}), whereas only a comparatively small mass of small wood chips (3.31 tonne ha^{-1}) was needed for the same coverage (Buchanan et al. 2002).

Short-length or small-sized materials also can provide other benefits toward controlling erosion. Döring et al. (2005) found two classes of chopped straw (50 mm long and <35 mm long) reduced soil losses by 97.4 to 98.4 percent compared to untreated soil. Straw that was an average 75 mm long but included pieces >100 mm long reduced soil losses by 91.7 percent. Shorter pieces of mulch were more effective at filling the spaces between soil ridges and contacting the soil in the valleys. Chopped straw pieces also fitted more tightly into gaps between pieces of mulch, allowing a smoother, flat mat to be created that was not attainable with the longer pieces of straw (Döring et al. 2005). Long pieces of mulch tend to bridge across the ridges, leaving the soil underneath more susceptible to rill erosion (Döring et al. 2005, Foltz and Dooley 2003, Meyer et al. 1972). However, 6.5-inch-long ponderosa pine (*Pinus ponderosa*) needles were more able to weave together to form mini-dams and retard rill development than 1-inch-long Douglas-fir (*Pseudotsuga menziesii*) needles, though the latter had better overall soil contact and controlled interrill erosion (Pannkuk and Robichaud 2003). Similar mini-dams were formed by 60- and 120-mm-long wood strands, but not by 240-mm strands (Foltz and Dooley 2003). Thus, size, shape, flexibility, and strength all have a role in determining the effectiveness of a given type of mulch.

It should be noted that these studies were very short-term and performed in controlled conditions. Under ambient field conditions or in the long term, small-sized, lightweight mulch may exhibit erosion control rates that are much less than those in the studies cited earlier. This is because short-length lightweight mulch, such as straw or wood, is susceptible to movement by water and wind (Barnett et al. 1967, Buchanan et al. 2002, Meyer et al. 1972). Dudeck et al. (1970) found short strand lengths of chopped mulch (wood excelsior) became detached and transported off steep slopes, making it ineffective at controlling erosion. Even long straw has been shown to be susceptible to movement on slopes with high precipitation inputs (Harding 1988). The presence of relatively high percentages of lightweight fines in wood shreds was a dominant factor in undermining erosion control effectiveness on 40-percent burned slopes (Foltz and Wagenbrenner 2010). Wood shred blends containing 18 percent and 24 percent fines could not resist movement from concentrated flow and had poor erosion control. In contrast, when fines were limited to less than 2 percent, effective erosion control occurred with 50-percent cover. Likewise, mixed-sized (13- to 25-mm diameter) and large-sized (>25-mm diameter) wood chips provided superior erosion control on a 55-percent slope compared to small-sized chips (6.4- to 13-mm diameter) because the small-sized chips moved during natural rain events over 130 days (Buchanan et al. 2002; Table 19). After only three rain events, there was a reduction in soil cover from an original 80-percent cover to 37 percent for the small-sized chips, with only 10-percent and 5-percent reductions for the mixed-sized and large-sized chips, respectively. Most of the change in coverage for small-sized chips occurred in the bottom two-thirds of the 11.4-m-long slopes, whereas for the other two size classes, the average reduction in cover along the entire plot length never exceeded 10 percent. If mulch does not retain the level of soil cover that was intended due to mobilization, erosion control effectiveness can be reduced significantly over any timeframe.

Another potential problem with using small-sized or short mulch is that it can become partially or fully buried as soil is detached and redistributed over time. Foltz and Dooley (2003) observed rapid burial of short straw to 10 mm into the soil. In the short term, burial may help protect the soil against rill formation by providing organic matter to bind soil particles and contributing to roughness near the soil surface. But as burial progresses and deepens, mulch protection at the soil surface may be sacrificed. Loss of erosion protection has been observed when mulch was intentionally mixed into the soil compared to when it was applied at the soil surface (Barnett et al. 1967).

For wood mulch, fiber length or the degree of processing apparently influences its effectiveness. In general, short fibers (more processed) decrease effectiveness. Three short-fiber wood pulp products applied by using a hydroseeder provided significantly poorer erosion control than a long-fiber excelsior product (blown product, not rolled) (Kill and Foote 1971; Table 19). Soil losses from short-fiber products were more than twice as high per unit area as from the excelsior product. Most of the difference in sediment yields was from rill erosion on the short-fiber treatment plots; more grass became established on the long-fiber plots, which also contributed to controlling soil losses. Zellmer et al. (1991) also observed much greater soil losses from a processed wood fiber mulch than from straw mulches on a 28-percent-gradient pipeline right-of-way in Pennsylvania.

Paper mulches manufactured from wood pulps have a mixed record of effectiveness. Zellmer et al. (1991) found paper strips in netting performed comparably to straw mulch (Table 19), yet Swanson et al. (1967) found poor erosion control from Kraft paper netting on 3:1 (horizontal:vertical) construction slopes. Stedman (2008) did not measure erosion directly from mulched cutbanks, but he reported that hydroseeded cellulose mulch (small-sized shredded paper) provided essentially no soil cover because rainfall washed it off within a few days of application.

Stone and gravel are unlike most mulches in that they are heavy mineral materials. They are used less commonly than other mulches on road prisms off the driving surface, and their typical application is confined to drainage areas where flow is intentionally concentrated (see Chapter 7). However, stone and gravel mulches occasionally have been used more broadly, especially where a high risk of erosion is anticipated. In some instances, the natural occurrence of stones at the surface of some soils also has been exploited as a type of mulch to protect against raindrop impact, increase soil roughness, and decrease rill and interrill erosion (Poesen and Ingelmo-Sanchez 1992).

Stone and gravel have shown somewhat inconsistent results in controlling erosion compared to other types of more traditional mulches. Although rock fragment cover of at least 10 percent at the soil surface almost always reduces sediment yields compared to no soil cover (Poesen et al. 1994), the degree of effectiveness varies substantially. In some studies, stone and gravel have provided reasonable erosion control (Meyer et al. 1972), yet in others erosion control has been relatively poor (Jennings and Jarrett 1985) (Table 19). Based on reported observations, the inconsistencies apparently are related to whether the protection they provided to the soil was sufficient to control rill erosion. Because of its weight, stone can resist movement by surface runoff and rill formation. When stone cover rates exceeded 88 percent, runoff could not create rills even on a 20-percent slope, so stone provided more effective erosion control than lighter, mobile wood chips (Meyer et al. 1972; Table 19). At lower percentages of soil cover (60 to 70 percent), rills were able to form among the stones, so erosion rates exceeded that of wood chips (Meyer et al. 1972). Jennings and Jarrett (1985) similarly found that on relatively flat ground with soil cover between 70 and 80 percent, organic mulch (straw and bark) had better erosion control than stone; they attribute these results to organic mulch's capacity to absorb and retain moisture, which reduced runoff and erosion (Jennings and Jarrett 1985; Table 19).

The control of interrill and rill erosion resulting from stone cover is a function of the scale at which they are examined, and should be considered when interpreting erosion control effectiveness. At microscale (4×10^{-6} to 1 m^2) and macroscale (10 to 10,000 m^2) plot levels, coarse fragments will reduce erosion (relative to no soil cover); however, at the mesoscale (0.01 to 100 m^2), coarse fragment cover can result in increases or decreases in erosion, depending upon the porosity of the surface soil (Poesen et al. 1994). If the soil surface is dominated by what Childs (1969) called structural porosity (i.e., it contains voids and pores characteristic of newly excavated or tilled material), coarse fragments on or slightly embedded into the soil surface will reduce runoff and erosion (Poesen et al. 1994) due to the high infiltration capacity of macropores at the surface.

In contrast, a soil with textural porosity (essentially a sealed surface [Childs 1969]) will have lower infiltration rates. Coarse fragments on the surface of these soils may allow some infiltration, but it will be reduced compared to soils with structural porosity, and there will be little infiltration with embedded coarse fragments (Poesen et al. 1994). Consequently, overland flow can develop, and without 100-percent cover, erosion can increase with increasing cover (Poesen et al. 1999). Interrill and possibly rill erosion will result as flows are channeled into narrower pathways between more densely spaced fragments, thereby increasing the velocity, depth, and energy of the runoff (Poesen and Ingelmo-Sanchez 1992, Poesen et al. 1999). Although larger-scale processes ultimately overwhelm processes at smaller scales (Poesen and Ingelmo-Sanchez 1992), the processes at smaller scales contribute to the final efficacy of stone cover, and it is hypothesized that erosion increases at the mesoscale may help explain poorer erosion control seen in some studies (Poesen et al. 1994). This further suggests the importance of applying stone cover soon after soil disturbance so that structural porosity is maximized and soil sealing is minimized.

It would seem logical that the porosity of the soil surface also would affect the performance of other types of mulches and perhaps rolled erosion control products, but there are no similar investigations of this phenomenon in the literature for other soil cover products. It is possible in the moderate-to-long term that other types of mulches may be less prone to the effects of initial soil sealing. Less heavy materials may allow rapid re-creation of macropores after sealing through freeze/thaw and wet/dry cycles and through biological processes, such as the growth and extension of roots and churning by soil fauna.

The actual influence that scale has on erosion under mulched conditions is largely dependent upon whether erosion is dominated by interrill or rill erosion or a combination of both. Long plots present greater opportunity for severe rill or gully erosion to develop because of their length, and thus for greater overall erosion if mulch is insufficient to protect the soil (Snelder and Bryan 1995). King (1979) and Meyer et al. (1972) observed these outcomes. In contrast, when present in sufficient amounts mulch can become more effective on long plots because of the greater potential to buffer runoff and sediment; that is, there is greater potential for runoff to infiltrate and for sediment to be deposited on the plot due to the greater length (Smets et al. 2008). This was apparently the situation in a study Smets et al. (2008) performed using a worldwide dataset for plots that exceeded a length of 11 m. Mulch became more effective from 11 m to 50 m (the maximum plot length). Of course, even with greater mulch effectiveness on longer plots, the absolute amount of erosion increases with length (Hudson 1957, King 1979, Meyer et al. 1972, Smets et al. 2008), simply because of the greater surface area and length from which soil loss can originate (Smets et al. 2008).

On short plots there is almost no chance for infiltration or deposition to occur, so effects of simulated or real rain or inflow events are almost immediately measured (Smets et al. 2008). This explains the substantial variation in the effectiveness to control runoff or erosion by all of the mulches in the 65 plots that were less than 11 m long (Smets et al. 2008). Although rills can form on short plots, interrill erosion is typically more dominant, especially in the short term. But because both can exist and the buffering capacity of long plots is absent, it is much more difficult to make generalizations about whether mulch becomes increasingly effective with length on short plots.

Although slope length and scale are important site factors, slope gradient is sometimes the single most important factor in controlling erosion (Megahan et al. 2001). There are interactions between slope gradient and length, however, and the two together substantially affect the ability of mulch to remain in place and the likelihood of rill development (Foltz and Copeland 2009).

Flatter areas tend to be dominated by interrill erosion, and rill erosion dominates steeper gradients (Lattanzi et al. 1974, Meyer et al. 1972), but both interrill and rill erosion increase with increasing gradient (Lattanzi et al. 1974). The increase in both types of erosion occurs at least up to some point, after which soil armoring may form and thereby reduce erosion on some steep slopes (Descroix et al. 2001). Because soil detachment by raindrop impact—and not by runoff detachment—dominates interrill erosion, increasing slope increases the amount of soil detached (and therefore erosion) through the change in angle of impact. Gravity contributes to further downslope movement of detached particles (Lattanzi et al. 1974).

The generally direct relationship between slope and erosion, especially on newly disturbed soils, means that steeper slopes tend to require greater amounts of mulch than less steep slopes to provide the same level of protection (Kramer and Meyer 1969, Meyer et al. 1970). With a sufficient amount of mulch, interrill erosion is preventable on virtually any slope because raindrop impact can be nullified completely at 100-percent cover (Lattanzi et al. 1974). However, mulches that tend to be mobile (transported by water or wind) on steep slopes may require some type of anchoring, such as netting, to maintain their effectiveness (Megahan 1974a). Increasing the mass of a mobile mulch on increasing slopes without any anchoring may provide little additional benefit because exposed soil would be susceptible to raindrop splash and rill erosion (Bethlahmy and Kidd 1966, Lattanzi et al. 1974, Meyer et al. 1972). Similarly, if the type of mulch applied cannot control rill erosion because of its physical characteristics (e.g., Foltz and Wagenbrenner 2010, Meyer et al. 1972), the mulch may be largely ineffective at controlling erosion on sloped land, especially steep slopes (Lattanzi et al. 1974).

Few studies specifically compare a variety of types of mulches at equivalent soil covers over a range of slopes, so it impossible to recommend types of mulch that are best suited for specific situations. However, Lemly (1982) and Meyer et al. (1970), respectively, reported tacked straw and straw provided poor protection on steep slopes. Erosion control by tacked straw declined by 10 percent from a slope of 10 percent to a slope of 20 percent, and erosion control on a 50-percent slope was about half of what it was on a 10-percent slope (Lemly 1982).

On probably any slope, but especially on steep slopes, soil texture is a large factor in the absolute amount of erosion control that any mulch will have. The ability to control erosion of the finest sized particles is key to erosion control. Small-sized particles are not necessarily the easiest to detach, as structure has a big influence on shear strength, but they are the easiest to transport once detached (Sharma 1996). Eroded particles smaller than 0.04 mm from a red clay soil were much more poorly controlled by mulches than particles that exceeded 0.04 mm (Lemly 1982). Yanosek et al. (2006) similarly found that wood strand mulch was much more effective at controlling erosion of coarse-grained gravelly sand than of a fine-grained sandy loam. Kramer and Meyer (1969) used glass spheres to simulate two fine soil particle sizes (33-µm and 121-µm diameters) to examine the effect of steepness, plot length, and mulch rates on controlling erosion of the different sized particles. They simulated straw mulch with ⅛-inch-wide × 6-inch-long fiberglass strips. As slopes steepened for any given mulch rate and slope length, the erosion rate (kg min^{-1} m^{-1}) was greater for the smaller particles than larger particles (Table 19). The difference between the two particle sizes increased markedly with increasing slope gradient and length for all mulch rates.

Effectiveness of Rolled Erosion Control Products

Rolled erosion control products (RECPs), sometimes referred to as “geotextiles,” have been used for more than 50 yr (Bhattacharyya et al. 2008). They come in many forms and may be all natural, all synthetic, or a combination of the two. They may be made from jute, coconut fibers (also called coir), sisal, paper, wood chips, cereal straw, nylon, polypropylene, polyester, polyethylene, or other materials (Rickson 2006). The composition of an RECP determines its longevity in the field. Natural products typically last about 2 to 5 yr and synthetic products may last more than 25 yr (Rickson 2006). Jute-containing products, which are common RECPs, can deteriorate measurably in less than a year (Mitchell et al. 2003). Some products are woven; others are blankets or mats (Fig. 10). Some have netting; others are net-free (Smith 2007). There has been a shift toward products composed of all or greater amounts of natural

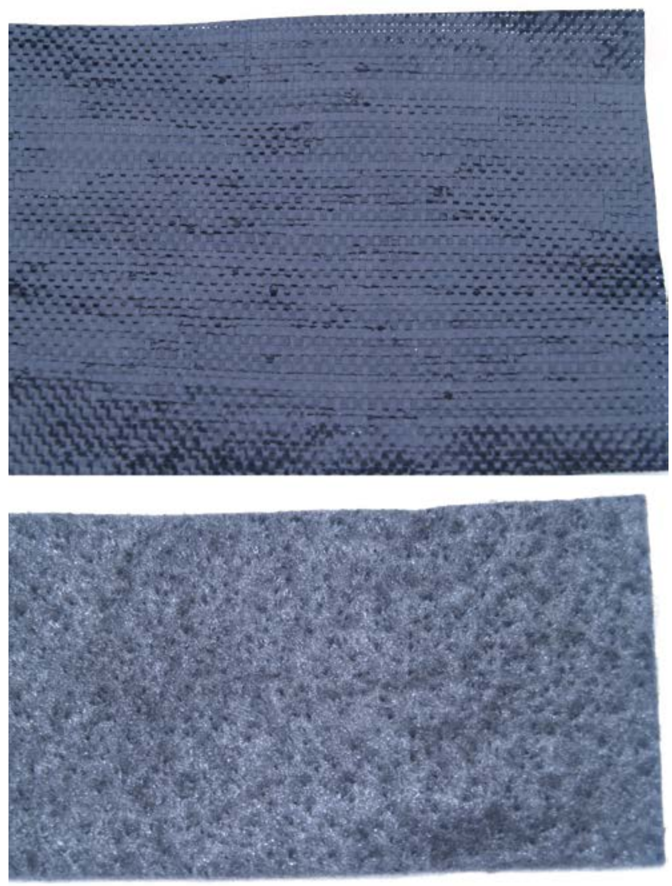

Figure 10.—Examples of woven (top) and nonwoven (bottom) geotextiles.

fibers because of their biodegradability (Smith 2007). However, as long as netting remains intact, it can trap some smaller wildlife species (Barton and Kinkead 2005, Kapfer and Paloski 2011, Walley et al. 2005), so net-free products may be a better choice in areas that contain species of interest.

RECPs differ from traditional mulches in that the latter are not associated with specific manufacturers, whereas each RECP is trademarked and manufactured according to a standard set of specifications. New products are placed on the market regularly and existing products are removed or their specifications changed. This can make it difficult for RECP users to keep pace with the available information about effectiveness and be familiar with the specifications of available products.

However, most of the published papers and available reports on the erosion control effectiveness of RECPs have been reviewed in detail in four comprehensive publications. Sutherland (1998b) and Sutherland (1998c) provide thorough reviews and critiques of studies available before 1990 and from 1990 to about 1997, respectively. Sutherland (1998a) assesses the research on RECPs done at the Hydraulics and Erosion Control Laboratory at Texas A&M University. Smith (2007) reviews many of the same studies evaluated by Sutherland (1998a, 1998b, 1998c) and also includes studies that were completed after Sutherland's papers were published. Rather than providing a lengthy review of individual product comparisons here that would be redundant with and probably less detailed than the information in those papers, we direct the reader to those publications for a thorough evaluation of available studies. The focus of this portion of the chapter will instead be a broader discussion of product characteristics that appear to be important in controlling their effectiveness. We use the word "appear" intentionally in the preceding sentence because there are limited studies of the processes that contribute to RECP effectiveness, and this is an identified weakness when improving the manufacture of RECPs and selecting the appropriate RECP for specific field conditions (Smith 2007).

Comparative tests with other types of erosion control products indicate that in virtually all instances, RECPs can reduce erosion of bare soil substantially (Benik et al. 2003, Grace et al. 1996, Mitchell et al. 2003, Rickson 2006), and their performance is comparable to and usually better than mulches (Dudeck et al. 1970, Harding 1988, Kill and Foote 1971, Swanson et al. 1967, Urroz and Israelsen 1995) (Table 19). Neither of these statements is surprising because RECPs are subject to market demand, and if they perform poorly they are unlikely to make it to market or to outcompete better-performing products. However, erosion control varies substantially among products or in different situations, so selecting the best product for erosion control is not straightforward or necessarily easy. There are no standard ways in which the erosion control by individual products is evaluated or in which different products are compared in a given situation. Most studies are empirical or "black box" studies (e.g., Krenitsky and Carroll 1994, Sutherland and Ziegler 1996, Thompson et al. 2001, Urroz and Israelsen 1995) in which RECPs are applied in a specific setting and the results are monitored with little if any replication or consideration of other experimental design components (Ziegler and Sutherland 1998). Further, little effort is given in most studies to understanding the specific processes contributing to or controlling erosion (Morgan and Rickson 1995, Smith 2007, Thompson et al. 2001, Ziegler and Sutherland 1998).

Standardized or uniform testing procedures for evaluating and comparing erosion control effectiveness of RECPs have long been identified as a need for the industry (Harding 1988, Rickson 2006, Sutherland 1998c), though the need has generally been ignored in practice (Sutherland 1998c). Despite some movement in developing standard approaches, industry-wide standardization still does not exist (Sutherland 1998c). Instead, major users, such as members of the American Association of State Highway and Transportation Officials (AASHTO) in their National Transportation Product Evaluation Program (NTPEP), use results from a collection of standard American Society for Testing and Materials (ASTM) tests developed by the Erosion Control Technology Council to standardize comparison of RECPs (AASHTO 2012, Smith 2007). However, manufacturer participation is voluntary (AASHTO 2012) and ASTM tests evaluate physical components of materials, such as strength and flexibility, that do not equate to erosion control effectiveness (Harding 1988).

In addition to the lack of uniform testing across the industry, there is also no way to know how well results can be extrapolated to other geographic situations or larger spatial scales (Sutherland and Ziegler 1997). Comparison of products across all types of field conditions would be an informative but extremely expensive, tedious, and lengthy process (Thompson et al. 2001) that might be unable to keep up with product development. Thompson et al. (2001) suggested an alternative approach to hastening those types of comparisons by focusing on understanding how each RECP or RECP attribute affects erosion, contributes to erosion control, or interacts with processes that influence erosion, such as shear stress. General processes and controls are understood somewhat. But a detailed understanding of processes would allow product selection on the basis of erosion control potential rather than other known, yet not useful, variables (from the perspective of erosion control effectiveness), even if a product has not been fully tested in a wide variety of situations. Understanding processes and interactions also would improve the development of more-effective erosion control products (Smith 2007). Without process-based studies, improvement of erosion control products will occur primarily through slow trial-and-error applications rather than through scientific study (Sutherland 1998c). Study duration would have to be considered in process studies to understand when, why, and how interrill erosion makes the transition to rill erosion and to understand when, why, and how erosion becomes self-healing through time (Sutherland et al. 1997).

Like other soil cover techniques discussed in this chapter, the effectiveness of an RECP depends on its ability 1) to control raindrop splash, runoff, and associated sediment transport; and 2) to allow, if not enhance the ability of, vegetation to become established (Benik et al. 2003, Lekha 2004). The second ability includes physically allowing developing vegetation to grow through the material and in some cases improving the microclimate for seed germination and plant growth, such as by providing moderating temperatures and improving soil moisture levels. And like mulches, the physical attributes of the RECP influence its effectiveness in preventing erosion and runoff processes and allowing vegetation establishment.

Few studies evaluate the effectiveness of RECPs in controlling raindrop splash (Bhattacharyya et al. 2010), but existing studies generally show that splash erosion is substantially less or statistically less compared to splash from exposed soil (e.g., Bhattacharyya et al. 2010; Sutherland and Ziegler 1996, 2007; Ziegler et al. 1997). The limited number of studies that have analyzed or discussed results in the context of the product characteristics suggest that controlling raindrop impact is most dependent upon two attributes: a low amount of open area in the material (i.e., high surface coverage) and a complex, thick, three-dimensional design (Harding 1988, Ingold and Thomson 1986, Smith 2007, Sutherland et al. 1997). The first characteristic is somewhat analogous to the need to have mulch soil coverage of 50 to 70 percent to achieve substantial benefit. However, there has been little analysis to identify the surface area coverage required by different types of RECPs to provide effective protection against raindrop splash (Sutherland et al. 1997).

Sutherland and Ziegler (2007) compared raindrop splash erosion of two open-weave coir RECPs. They found that the one with the higher mass per area and 30- to 40-percent open space had significantly lower splash erosion than the product with lower mass per area and 50- to 80-percent open space. Sediment concentrations and yields were 6.5 and 6 times, respectively, greater for the material with more open space than for the more closed coir product. A randomly oriented coir fiber RECP with the least open space (less than 10 percent) had the smallest splash erosion concentrations and yields; both were about one-third that of the material with high mass and low open space. Bhattacharyya et al. (2010) found borassus mats (black rhun palm; *Borassus aethiopum*) with about 23-percent open area significantly reduced splash erosion and splash height compared to bare soil. Splash erosion was reduced by about 89 percent, and splash height by about 54 percent. In contrast, splash erosion from buriti mats (buriti palm; *Mauritia flexuosa*) with about 56-percent open area was not significantly different from bare soil.

Product thickness generally contributes to the material's ability to absorb raindrop energy instead of transferring it to the soil surface (Bhattacharyya et al. 2010, Ziegler and Sutherland 1998, Ziegler et al. 1997). RECPs that are thick also may do a better job of catching and retaining detached soil particles in their matrix than RECPs that are relatively thin (i.e., more sheet-like), especially on steep slopes (Sutherland and Ziegler 1996). However, thickness alone does not guarantee energy absorption. Without a complex three-dimensional internal structure, raindrops can pass unimpeded to the soil, limiting energy dissipation by the product. Of 13 tested RECPs, Ziegler et al. (1997) found the only one that did not mitigate raindrop splash was the thickest one because it had the highest amount of open area, and thus, least soil cover. Because nonwoven blanket or mat products have thick, complex internal designs, they are often better at preventing raindrop impact than woven products, which tend to be thinner with holes extending through the entire product thickness (Smith 2007; Sutherland and Ziegler 1996, 1997).

If the RECP is effective at controlling raindrop splash, it necessarily absorbs much of the raindrop energy. As a result, soil sealing is prevented, infiltration is maintained, and the ability to transport sediment is reduced (Sutherland et al. 1997, Ziegler et al. 1997). However, the elimination of splash also can result in temporary ponding of water at the soil surface (Bhattacharyya et al. 2010) as water droplets make their way through the RECPs. With sufficient mass, these accumulations can be transformed into interrill and rill flow, especially on sloped land. Research suggests that in these situations, controlling runoff and sediment transport is dependent largely upon having good contact between the RECP and the soil surface, a characteristic termed "drapability" (Sutherland and Ziegler 1996, 1997).

Drapability is believed to be critical to reducing runoff velocity by impounding flow and increasing flow depth, and providing roughness for flowpaths even if the product has little effect on reducing overall runoff volume (Rickson 2006). Interrill and rill erosion and flow processes are poorly understood for RECPs, but there is a general belief that drapability creates conditions that favor interrill flow and discourage rill flow (Lekha 2004). RECPs that are most effective at controlling runoff and sediment loss apparently are those that prevent the transition from interrill flow to rill flow (Smith 2007, Sutherland and Ziegler 1997). Thus, even if sediment is available, little of it will be transported as long as the RECP remains highly drapable because runoff will occur primarily as interrill flow—which has lower velocity and energy than rill flow (Sutherland and Ziegler 1997).

Fiber flexibility is an important variable controlling drapability (Sutherland and Ziegler 1997), and flexibility can increase or decrease with moisture for some RECPs, particularly organic materials (Cazzuffi et al. 1994; Rustom 1993; Sutherland and Ziegler 1996, 1997). Consequently, declines in drapability can result in the initiation of rill erosion. Rustom (1993) observed these effects when stiff coconut fibers expanded and lost contact with the soil during wetting.

Although it may be possible to control rill erosion with RECPs, complete elimination of interrill erosion with a single RECP currently on the market is probably impossible on sloping hillsides because interrill runoff is extremely difficult to prevent. As with mulches, the soil particles detached by interrill erosion are dominated by small particles, and these can be transported with very little flow energy (Lemly 1982). Interrill erosion is related directly to runoff velocity and velocity increases with slope length, so on long slopes interrill erosion expands (Fox and Bryan 1999) and can be very challenging to control. Hence, stopping interrill erosion entirely probably is achievable only by immediate infiltration or by retaining water (through the formation of small dams by the product or absorption of surface water by the product) until complete infiltration can occur, or by treating the surface soil underlying the RECP with a soil conditioner to prevent detachment (see Soil Conditioners section in this chapter).

This bias for small-particle transport was illustrated by Lemly (1982), who examined erosion resulting from a variety of RECPs (and mulches) on clay soils at highway construction sites (Table 19). Consistently across five different slopes, the sediment in runoff at peak discharge was dominated by soil particles smaller than 0.038 mm (Table 20) for all types of erosion control products used. In most cases more than 90 percent of eroded soil was in this smallest size class and in all cases the majority was in this class. And while erosion control for size classes smaller than 0.038 mm improved with increasing slope for all RECP (and mulch) treatments, erosion control of particle sizes less than 0.04 mm was always much poorer than that of the larger particle sizes for slopes of 10 percent or greater (Table 21). For example, RECPs reduced sediment yields by 50 percent or more at 10-percent slopes, but silt- and clay-sized particles

Table 20.—Mean composition of sediment, as percentages, in runoff during peak discharge[a] (from Lemly 1982)

Treatment[b] and particle size (*mm*)[c]	Percent slope				
	10	20	30	40	50
No treatment					
0.50	99	96	93	90	85
0.25	98	91	88	83	79
0.10	97	89	86	80	73
0.038	96	87	83	77	67
Tacked straw					
0.50	99.5	99	97	95	94
0.25	99	98	96	93	90
0.10	98	97	94	91	88
0.038	97.5	96	93	88	85
Jute netting					
0.50	99.6	99	98	96	92
0.25	99	98.5	97	94	90
0.10	98.6	98	94	93	89
0.038	98	97	93	90	86
Mulch blanket					
0.50	99.7	99.2	98.8	97	95
0.25	99.2	99	98	96	94
0.10	99	98.5	97	95	93
0.038	98.8	98	96	93	91
Wood chips					
0.50	99.6	99	97	96	95
0.25	99.1	98.5	96	95	93
0.10	98.8	98	95	93	91
0.038	98	97	94	91	90
Excelsior blanket					
0.50	99.7	99.3	99	98	97
0.25	99.3	99	98.5	97	96
0.10	99	98.7	98	96	94
0.038	98	98	97	94	93
Multiple treatments					
0.50	99.9	99.8	99.5	99	99
0.25	99.8	99.6	99.2	98.7	98.5
0.10	99.6	99.4	99	98	98
0.038	99.3	99.1	98.7	97	97

[a] Data were pooled and averaged for 14 precipitation events.

[b] Rolled erosion control product treatments are jute netting, mulch blanket, and excelsior blanket; mulch treatments are tacked straw and wood chips; multiple treatments involve a chemical binder applied to the soil surface, a fiberglass erosion check dam at the base of the plot slope, and asphalt-tacked straw covered with secured jute netting.

[c] Values represent percent finer than the particle size indicated, based on dry weight.

Table 21.—Effect of treatment on sediment reductions[a] for different experimental construction plot slopes (from Lemly 1982)

Treatment[b] and sediment component	Percent slope				
	10	20	30	40	50
Tacked straw					
Total	42	39	36	29	23
>0.04 mm	98.5	91	90	89	82
<0.04 mm	1.5	9	10	11	18
Jute netting					
Total	55	43	32	31	30
>0.04 mm	98	90	90	87	81
<0.04 mm	2	10	10	13	19
Mulch blanket					
Total	76	70	61	55	52
>0.04 mm	96.2	89	87	84	76
<0.04 mm	3.8	11	13	16	24
Wood chips					
Total	78	74	63	58	51
>0.04 mm	98	90	89	86	77
<0.04 mm	2	10	11	14	23
Excelsior blanket					
Total	78	73	62	58	52
>0.04 mm	98	89	86	81	76
<0.04 mm	2	11	14	19	25

[a] "Total" values are percent reductions, dry weight, of average total sediment concentrations (mg L^{-1}) during peak discharge compared to untreated plots. Values for ">0.04 mm" and for "<0.04 mm" are the sediment concentration reductions as a percentage of the total for each particle-size class.

[b] Rolled erosion control product treatments are jute netting, mulch blanket, and excelsior blanket; mulch treatments are tacked straw and wood chips.

were the dominant particle size in runoff, which was primarily interrill flow.

Long-term effectiveness of an RECP also depends upon its ability to retain good drapability throughout its intended life. Wind, water, and, on steep slopes, gravity can cause the loss of contact of RECPs with the soil over time. Consequently, many products require the use of staples or pegs designed for the RECP to keep it attached to the ground properly (Ingold and Thomson 1986, Lekha 2004, Rimoldi and Zhao 1996). Thompson et al. (2001) found that the density of the placement of staples used to attach erosion control blankets was critical for retarding particle detachment because greater blanket contact reduced shear stress at the soil surface. Moisture absorbed by RECPs with high water-holding capacity can contribute to an RECP's drapability and help hold it in place due to the weight of the wet material (Rickson 2006).

Water absorption also reduces the amount of free water available for interrill or rill flow (Ziegler et al. 1997). Jute-containing RECPs particularly are known for their ability to retain moisture and increase their drapability (Cazzuffi et al. 1994, Sutherland and Ziegler 1996). Conversely, as some RECPs become wet, they expand and get heavier, which can result in their drifting down steep slopes. They will lose contact with the soil and lose their erosion control effectiveness if sufficient vegetation does not become established quickly to hold the product in place (Mitchell et al. 2003). If a product has high drapability because of high moisture absorbance but has low strength, it will be highly susceptible to tearing on slopes containing sharp rocks, debris, and perhaps even gravel (Sutherland and Ziegler 1997). This means that some RECPs are not suitable for all situations, particularly if large rocks, soil clods, root wads, and other debris are not removed or cannot be removed fully to achieve drapability and to avoid damaging or tearing the material (Rimoldi and Zhao 1996).

Even with good drapability, RECPs applied on a soil surface that is already sealed by fines (e.g., due to delayed application of the RECP) may not provide effective erosion control, and will be less effective than on a similar unsealed surface (Smets and Poesen 2009). Likewise, an RECP that has excellent drapability, moisture absorption characteristics, and other physical characteristics that encourage interrill flow can be challenged if there is a soil layer beneath the surface that impedes percolation. Saturation excess overland flow can result, creating rill flow and erosion even if the product is functioning at its best (Rustom 1993).

In addition to drapability, surface area coverage, and thickness, fiber orientation is important for controlling sediment transport. Three-dimensional, randomly oriented fibers are most effective at reducing rill erosion (Sutherland and Ziegler 1996, 1997). Sutherland and Ziegler (2007) compared two open-weave coir products. Random fiber orientation in one RECP resulted in significantly lower sediment concentrations and loads, as well as less rill formation, compared to the second RECP, which had a more regular fiber orientation. Further, if the dominant fiber orientation is across the slope (i.e., along the contour) rather than vertical, interrill erosion of detached soil particles can be retarded (Sutherland and Ziegler 1997).

Fiber length also may play an important role in sediment transport, at least for natural fibers. Smith (2007) found that RECPs containing short fibers and low amounts of synthetic materials were most effective at controlling erosion and sediment transport. Essentially all of the 13 products she tested were effective at controlling raindrop splash, but only natural materials with short fibers were similarly successful at stopping or largely controlling sediment movement. These materials all had high percent ground coverage, high water absorption capacities, high drapability (in part due to the short fiber length) especially with wetting, and adherence to the soil from product tackiness. Long-fiber products were less effective and sometimes fairly ineffective at controlling runoff because the fibers were too long to conform to irregularities of the soil surface. Often they were thin so as not to absorb water well; thus, their flexibility remained poor in wet conditions. A fully synthetic product constructed from polypropylene was the least effective product. Its structure was the most regular, and it had poor water absorption, which led to the most rill formation.

Sutherland and Ziegler (1997) also reported that a PVC product provided poor erosion control because of its poor drapability, even though it had randomly oriented fibers and high mass density (an index of ground cover). Thin nylon monofilament materials provide poor erosion control, in part due to their inability to trap and retain loose soil particles (Harding 1988). Even when they have good contact with the soil, structures can be so thin that water flows on top of the material (Smith 2007).

A few types of RECPs are designed to be laid on the soil surface and then covered or filled with topsoil (e.g., geoweb and geocells, respectively). The effectiveness of these types of products has been inconsistent, and in some cases erosion has been reported to be as high as or even exceeding that of bare soil (Cazzuffi et al. 1994, Ingold and Thomson 1986). Soil losses are elevated due to the combination of raindrop splash and increased runoff. The webbed nature of these products disrupts normal drainage patterns, causing soil saturation within the cells and greater surface flows.

The physical factors of RECPs that retard raindrop splash and sediment transport also often encourage the success of seed germination and vegetation establishment, and the converse is true as well. In open-weave or other products that permit substantial raindrop impact, seeds have become displaced from the soil surface or redistributed so that developing cover is not even (Rickson 2000). Redistribution or complete loss of seed also has resulted when RECP expansion during wetting has allowed rill flow and rill formation to occur. In some cases, the effects have resulted in seed germination that has been no better than bare soil with no protection (Krenitsky and Carroll 1994, Rickson 2000). In contrast, RECPs that provide good protection against raindrop impact, soil detachment, and sediment transport can result in very good seed germination. Dudeck et al. (1970) reported mulch and RECP effectiveness on fillslope plots, with the best grass seed establishment occurring on soil protected by excelsior or jute netting. However, there may be an antagonism between high surface area coverage and the complexity/thickness of products in that germinated seeds may not be able to penetrate the material easily (Krenitsky and Carroll 1994). Instead, germinating plants sometimes lift the product off the ground (Theisen 1992). Erosion rates can increase due to the loss of RECP contact with the soil surface and the reductions in roughness (Rickson 2006) and water absorption (Ziegler et al. 1997) that result from loss of drapability.

Only recently has research begun to focus on identifying specific RECP characteristics that provide erosion control benefits, including controlling raindrop splash, absorbing and temporarily detaining water, controlling the transition from interrill to rill erosion, limiting interrill flow, and encouraging successful germination of vegetation. Some of this work has required a new approach to analyses, such as understanding how cellular structures of organic materials behave temporally through short-term wetting and drying cycles, and through extended periods of use in which product degradation may occur (Smith 2007). As the conditions and interactions that affect RECP performance become better understood, new products are expected to be developed that exploit that information and improve their erosion control effectiveness; also anticipated is guidance that better defines situations and sites best suited for specific product application (Smith 2007).

Riprap applied to a forest road cutbank for stabilization after initial erosion control measures following road construction were not adequate during winter storms. (Photo by U.S. Forest Service, Umpqua National Forest.)

CHAPTER 7

Barriers and Biofilters

The focus in this chapter is on drainage, and hence, sediment control once water is diverted off the road surface or from a relief ditch. On forest roads, most road drainage typically is discharged onto roadside fillslopes or onto the original hillside adjacent to the area disturbed by road construction. The water's energy, erosive potential, and sediment delivery/deposition then are expected to be controlled by infiltration and sediment deposition, via the natural landscape attributes and features or humanmade structures, or a combination thereof. There are varying degrees of sophistication in the design of BMPs that rely on infiltration and deposition, but those that require substantial engineering, such as concrete energy dispersion ditches and settling ponds with designed outflow devices, are not included in this chapter or elsewhere in this document because they are rarely used in most forest road applications.

The barriers that are described in this chapter are non-engineered barriers—in other words, barriers that are installed with little or no design requirements other than standard installation guidelines (e.g., standard techniques given in BMP manuals for anchoring to the ground). Non-engineered barriers include such techniques as silt fence, straw bales, and fiber rolls or logs. In most cases, barriers are applied in two types of situations. One is when discharge is concentrated, including where concentrated discharge is cast onto adjacent land. This application typically is associated with roads, such as for water turnouts, ditches, and other similar features. The second is along the perimeter of, or as slope interrupters within, disturbed areas of construction sites (Faucette et al. 2009b). Barriers in both types of applications tend to be meant only for short-term use because the reestablishment of vegetation is the primary long-term erosion control technique.

The second section of this chapter is devoted to the class of BMPs known as "biofilters," which depend upon vegetation to reduce pollutant levels. These include buffer strips, such as vegetated filter strips and conventional forest buffers, and vegetated waterways and swales.

Non-engineered Barriers

Many types of non-engineered barriers are used to control erosion, but they fall into two general categories. One type will be called nonreactive barriers, which include the more commonly applied and more conventional types of barriers such as silt fence, straw bales, rock/stone berms, and fiber rolls or logs (also called wattles). These are in contrast to reactive barriers, the term given by Shipitalo et al. (2010) to the more recently developed techniques of filter berms and filter socks (Faucette et al. 2009b).

Nonreactive Barriers

Except for rare situations where barriers are used only to trap gravity-transported material (i.e., dry ravel), most nonreactive barriers involve water transport of pollutants and depend upon ponding and particle settling as the mechanisms for pollutant removal. The emphasis in this section is on this latter type of nonreactive barriers. The specific processes that lead to ponding and settling have been identified and described in detail only for silt fence, but all nonreactive barriers probably depend upon those same processes to one degree or another.

The initial key step for effective pollutant control is blinding and clogging of the openings in the barrier by large particles in runoff (Barrett et al. 1996, Kouwen 1990, Theisen 1992). Multiple small particles also can clog barriers if they reach an opening simultaneously and form bridges that clog barrier openings (Barrett et al. 1996). Blinding is a surface phenomenon where a crust forms across pores along the outer surface of the receiving side of the barrier, whereas clogging occurs when the interior pores become filled with solids and obstruct water flow (Holtz et al. 1998, Ossege 1993). Because it is impossible to distinguish between the two processes during testing or field use, the terms may be used interchangeably or no distinction may be made between the two in the literature (e.g., see Crebbin 1988, Kouwen 1990). Many people erroneously believe nonreactive barriers remove sediment by filtering (especially silt fence, which is often referred to as "filter fabric"), but particle filtering by nonreactive barriers happens only over the short time during which blinding and clogging occur (Barrett et al. 1996). Ponding and settling follow as a result of clogging and blinding.

Clogging and blinding reduce the permeability of the barrier material, which encourages temporary ponding, thereby allowing sediment to settle out during the much longer periods of inflow detention (Barrett et al. 1996, Kouwen 1990, Theisen 1992). Thus, barrier characteristics that contribute to ponding and increasing the time of detention are the ones that are most effective at retaining sediment (Barrett et al. 1998a). Such materials tend to be thick (three-dimensional) and have small pore sizes, as these characteristics reduce permeability and create slower, more tortuous flow paths that are conducive to greater particle interception and clogging, as well as longer ponding times (Barrett et al. 1996, Crebbin 1988).

Larger particles, particularly sands, dominate the settling process because settling velocities of smaller particles (silts and clays) are too low for deposition to occur during the time that water is ponded (Barrett et al. 1998a, Keener et al. 2007). Clays also are affected by Brownian forces that can keep them in suspension almost indefinitely (Smith 1920); thus, particles less than 0.02-mm diameter (i.e., medium-sized silt and smaller particles) are not removed effectively by ponding or by filtering/clogging with nonreactive barriers (Kouwen 1990). To illustrate, silt fence materials tend to remove 80 to 99 percent of sands compared to 50 to 80 percent of silt loams, and only up to 20 percent of silty clay loams (U.S. Environmental Protection Agency [EPA] 1993). Consequently, as the percentage of smaller particles in runoff increases, the trapping efficiency of nonreactive barriers decreases (Wishowski et al. 1998).

Silt fence is the industry standard for sediment control barriers during construction (Barrett et al. 1996, Faucette et al. 2008), so it is commonly used at road construction sites. Silt fence (and most nonreactive barriers) is meant to be used with sheet flow or small quantities of overland flow; it is not intended for use with concentrated flow or large amounts of overland flow, even if that flow occurs as sheet flow (Crebbin 1988, Island County Public Works 2003, Wyant 1980). Large volumes of inflow can overwhelm the ponding capacity and potentially compromise silt fence strength (Farias et al. 2006), which can result in overtopping, undercutting, circumventing, or tearing the fence material (Keener et al. 2007). Therefore, silt fence fabrics are not designed to control erosion at the outlet of most road drainage features (e.g., culverts, broad-based dips) even though they commonly are placed in those locations. Instead, silt fences are designed for use around the perimeters of disturbed areas, downslope of areas that contribute sheet flow, in small swales or ditches that carry very limited runoff, or upslope of points of concentrated flow to keep sediment from entering that flow (Barrett et al. 1996).

Silt fence is manufactured as either woven or nonwoven fabrics, using the same techniques described for woven and nonwoven rolled erosion control products in Chapter 6. Both woven and nonwoven materials are permeable to flow because the purpose of the fabric is to slow and temporarily pond water but not detain it indefinitely or until infiltration is complete. Consequently, fabric permittivity, as determined by the American Society for Testing and Materials (ASTM), is important for controlling flow-through rates and ponding (Barrett et al. 1998a, Farias et al. 2006).

Permittivity is a measure of the quantity of water that can pass through the fabric with a head of 50 mm of water (Chopra et al. 2010). The rate of ponding is inversely related to the permittivity of a fabric (Farias et al. 2006). However, ASTM-determined permittivity values for a given geotextile material are not equivalent to permittivity that would occur in field conditions (Denkler et al. 2000). This is because ASTM permittivity evaluates the quantity of clean water that can pass vertically through a specified cross-sectional area of a given fabric (subjected to 50-mm head of water; ASTM 2009), which is very different from field runoff that contains sediment and other contaminants, passes through the fence laterally, and has a variable and sometimes large head of water behind the fence.

Silt fence permittivity declines during use within and across storms as successive sediment-laden inflows increasingly clog the fabric (Wishowski et al. 1998). During flume studies, Barrett et al. (1998a) measured flow-through rates that were two orders of magnitude less than ASTM permittivity specifications. Farias et al. (2006) found permittivity was reduced to 62 percent and 85 percent of that for clean silt fence after two simulated rain events. For the four types of nonwoven geotextiles they tested, reductions in permittivity were greatest for thicker, less open geotextiles. Using woven silt fence fabrics, Denkler et al. (2000) measured initial flow-through rates that were four orders of magnitude less than the permittivity values specified for the material, and after long-term measurements of continuous flow-through (up to 2,000 h), they determined that flow-through decreased by as much as 50 percent compared to flow-through rates 1 h into the experiment.

Although permittivity affects water flow-through, sediment retention is not dependent upon flow-through rates (Wishowski et al. 1998). Instead, sediment retention has been found to be a function of the fabric's apparent opening size (AOS) (Theisen 1992) because the AOS provides an estimate of the size of the largest opening in a fabric through which soil particles can pass (ASTM 2004, Barrett et al. 1996, Chopra et al. 2010). The AOS is based on standard U.S. sieve sizes, and the AOS number and size of the opening are inversely related (Robichaud and Brown 2002). Some of the evidence linking AOS with sediment retention comes from findings that large particles are trapped more effectively than small ones; that is, fewer larger particles pass through material because the AOS estimates the largest size opening, but not all openings are that size and many are smaller. Wishowski et al. (1998) recorded trapping efficiencies of 99, 91, 74, 60, 54, and 45 percent, for the respective particle size classes >50 µm, >20 to 50 µm, >12.5 to 20 µm, >5 to 12.5 µm, ≥2 to 5 µm, and <2 µm. Similarly, retention of sand by four nonwoven silt fence fabrics over four replicate runs ranged from 84 to 99 percent (all but three runs for one fabric were 99 percent), whereas three nonwoven fabrics over four replicate runs all retained less than 7 percent of the fine silt/clay fractions (Fisher and Jarrett 1984). Coarse silt retention was between these two, ranging from 43 to 81 percent.

However, the relationship between retention and AOS is fairly weak and comparisons of fabrics with different AOS values have shown inconsistent results with respect to sediment retention. Britton et al. (2001) did find that the largest AOS corresponded to the fabric with the greatest particle transport (simulated with glass beads) through the material, but all three of the fabrics tested had similar particle retention (55, 65, and 66 percent) even though their AOS values spanned a fairly wide range relative to soil particle sizes (AOS=10, 30, and 40, which correspond to diameter openings of 0.80, 0.59, and 0.42 mm, respectively). Crebbin (1988) also reported that a fabric with a high AOS had markedly greater sediment retention than a fabric from the same manufacturer with a lower AOS.

Several factors contribute to the weak relationship between AOS and sediment retention. One is that AOS corresponds to the largest size opening, but not the frequency of those openings, which may be inconsistent across the fabric (Barrett et al. 1996, Ossege 1993). The frequency at which the manufactured AOS-sized opening occurs affects the probability that a given sized particle in inflow will encounter an opening of sufficient size to allow the particle to pass through the material. Second, the AOS is not uniform through the thickness of three-dimensional (i.e., nonwoven) fabrics, so particles transferred through larger openings on the surface may be captured in smaller openings within the interior of the material (Ossege 1993). Third, the AOS also decreases as clogging and blinding progress (Farias et al. 2006). Consequently, while permittivity and AOS have some control on flow-through and particle retention/loss, neither specification as determined through ASTM testing gives an accurate prediction of silt fence effectiveness in the field (Barrett et al. 1996, Martin 1985).

The temporal decreases in permittivity and AOS make geotextile materials increasingly susceptible to excessive ponding, and failure (Farias et al. 2006); therefore, although it is not discussed directly in the literature, it is probably important to anticipate and plan for changes in flow-through rates when selecting silt fence fabric. Selection of poorly suited materials for the soil texture, inflow rates, and erosion rates can result in structural failures that can substantially reduce silt fence effectiveness and yield large amounts of scour downslope from the fence when great volumes of ponded water are released suddenly (Storey et al. 2006, U.S. EPA 2002, Yeri et al. 2005). Broad guidance exists for selecting silt fence fabrics (e.g., California Stormwater Quality Association 2003, Wisconsin Department of Natural Resources 2006), but specific recommendations for material selection in different field conditions (e.g., soil, inflows, slopes, contributing areas) are not present in the literature. Denkler et al. (2000) described the appropriate silt fence for a given soil and situation as one that eventually reaches a steady rate of flow through the fence. Under steady state flow, clogging is no longer occurring and piping of fines through the fence is not increasing. However, steady state flow can take an excessively long time to achieve (for example, 360 to 2,000 h of inflow was required for loess soils in Denkler et al. [2000]), so this recommendation is probably not particularly valuable as a selection criterion.

Because the performance of silt fence cannot be accurately predicted from its simple physical characteristics, much of the effectiveness testing has defaulted to laboratory and flume studies that use controlled inflow and sediment (or simulated-sediment) composition. These studies have been useful in providing insight into the processes of clogging and deposition by which silt fence and other nonreactive barriers control pollutant transport (described previously). They suggest that some silt fence fabrics are capable of good to excellent sediment retention in controlled, short-term situations, whereas other fabrics result in very poor control (Table 22).

The variability in effectiveness is attributable to differences in the physical silt fence characteristics just described, as well as inflow rates, soil characteristics, and sediment slurry concentrations. Increasing inflow rates tend to result in lower sediment retention when all other variables are held constant (see Keener et al. [2007] in Table 22) because high inflow rates can cause overtopping or other failure of the fence (Keener et al. 2007). To avoid overtopping and failure, fabrics designed for high inflows have larger pore sizes and higher flow-through rates, which also contribute to poorer erosion control (Carroll et al. 1992). Retention increases when the soil suspended in inflow is dominated by larger particles (e.g., sands), as these can be trapped by the fabric or settled by ponding; in contrast, clays are more likely to stay suspended in water as it passes through the fence fabric (U.S. EPA 1993).

Elevated sediment concentrations tend to clog and blind silt fence fabric more quickly than inflow with low sediment concentrations (Henry et al. 1999), so fabrics with large pores can improve sediment trapping as long as the inflow rates are not so great that ponding results in overtopping or other silt fence failures. High-density silt fence fabrics, which tend to be nonwoven, typically trap more sediment than low-density fabric (Wishowski et al. 1998). This is because nonwoven silt fence materials have longer and more tortuous pores, so the probability of clogging, ponding, and sediment trapping are increased (Bell and Hicks 1980), and they are stronger so they can support greater volumes of ponded water (Wyant 1980). However, most geotextiles used for silt fence are woven (Crebbin 1988) so they often do not have optimal sediment retention or water detention characteristics (Wyant 1980).

Expectations of silt fence performance in the field should not be based on flume and laboratory experiments (Barrett et al. 1998a, U.S. EPA 2002). Flume and laboratory studies provide exaggerated estimates of silt fence performance compared to field conditions because the former often employ disproportionately large percentages of sand particles in inflows (Barrett et al. 1998a). In contrast, because fines are displaced by runoff and suspended easily, the percentages of clay and silt present in runoff in the field generally will exceed the percentages of clay and silt particles in the parent soil from which the eroded particles originated (Schueler and Lugbill 1990). Fines tend to pass through silt fence (Barrett et al. 1996, Yeri et al. 2005), so field performance can be poor under certain soil, fence, and inflow combinations. For example, when the median value of silt + clay in inflow was 96 percent in field studies, Barrett et al. (1998a) recorded a median retention efficiency of 0 percent with a standard deviation of just 26 percent.

Other field tests of silt fence performance show a wide range of effectiveness (Table 23), with some field effectiveness values being very low or even negative (i.e., sediment losses are increased with the presence of the silt

Table 22.—Trapping efficiencies of various types of silt fence fabric measured from flume or laboratory experiments

Reference	Trapping efficiencies	Description of study charactersistics
	percent	
Barrett et al. (1998a)	70, 90, 68	3 woven fabrics
	90	1 nonwoven fabric
Britton et al. (2001)	55, 65, 66	3 different woven fabrics; used glass beads to simulate soil particles
Crebbin (1988)	79, 87, 91, 91	4 woven silt fence fabrics
Farias et al. (2006)	62–85	4 types of needle-punched nonwoven fabric
Fisher and Jarrett (1984)	12–31	5 nonwoven fabrics using a sand slurry
	1–19	5 nonwoven fabrics using a coarse silt slurry
	0.7–2.6	5 nonwoven fabrics using a silt–clay slurry
	1.8	1 woven fabric using a sand slurry
	0.8	1 woven fabric using a coarse silt slurry
	0.1	1 woven fabric using a silt–clay slurry
Keener et al. (2007)	56	Average for 0.126 L s^{-1} inflow
	42	Average for 0.252 L s^{-1} inflow
	33	Average for 0.315 L s^{-1} inflow
		All with a silt loam slurry
Kouwen (1990)	>99	3 woven geotextiles using a 0.22-mm-diameter sand
Wishowski et al. (1998)	81	High density clean fabric
	69	Low density clean fabric
Wyant (1980)	97–99	9 nonwoven fabrics using a sandy soil slurry
	90–100	9 nonwoven fabrics using a silty soil slurry
	93–99	9 nonwoven fabrics using a clayey soil slurry
	92–98	6 woven fabrics using a sandy soil slurry
	49–99; only one was <84	6 woven fabrics using a silty soil slurry
	85–98	6 woven fabrics using a clayey soil slurry

Table 23.—Trapping efficiencies of various types of silt fence fabric measured from field experiments or field applications

Reference	Sediment retention	Comments
	percent	
Barrett et al. (1996)	-61 to 54[a]	One silt fence sample removed more than 26% TSS; 54% removal was a nonwoven fabric; measurements after 8 h following 1 storm; inflow contained mostly silt- and clay-sized particles
Ettlin and Stewart (1993)	16	From 5 storms on 34% slopes; as TSS
Grace (2003)	85	Silt fence installed in road turnout ditches; 3 replicates; 90 flow events; over 42 months; as TSS concentration
Horner et al. (1990)	85.7	Average from 2 plots with silt fence compared to 2 bare plots over 7 storms; as TSS
Yeri et al. (2005)	91	Sandy loam slurry at 10% slope
	51	Sandy loam slurry at 13% slope
	94	Loam slurry at 10% slope
	88	Loam slurry at 13% slope
	97	Silt loam slurry at 10% slope
	95	Silt loam slurry at 13% slope All 6 tests included new design of silt fence to improve effectiveness; as total sediment mass

[a]Negative values indicate that sediment in outflows exceeded that in inflows.

fence). In addition to the limitations of silt fence materials even under laboratory conditions, poor performance in field tests and field use often is associated with silt fence structural failures (e.g., Barrett et al. 1996, 1998a). Common reasons for silt fence failure include: improper installation, which can result in undercutting of the toe, erosion of poorly compacted fill in the toe trench, or flow around one or both ends of the fence; excessive stretching and tearing; and overtopping the fence at low points (Barrett et al. 1996, Harbor 1999, U.S. EPA 2002, Yeri et al. 2005). Supplemental support, such as the additional stakes (Robichaud and Brown 2002) or wire woven into the silt fence fabric during manufacturing (Wyant 1980), can help provide greater strength to silt fence, but even these extra precautions may not be sufficient to keep silt fence operational with high volumes of inflow or when installed improperly or in the wrong location. Lack of maintenance, even for short-term use, also is often cited as a reason for silt fence failure (Landphair et al. 1997, Stevens et al. 2007). Because the need for maintenance increases with duration of use, the effectiveness of silt fence for erosion control during field applications may be poorer than "effective" short-term field tests suggest (e.g., Horner et al. 1990, Yeri et al. 2005) (Table 23). However, if fences are maintained, Grace (2003) (Table 23) has shown them to provide reasonable levels of sediment retention over periods typical of field use.

Given the general inability of silt fence fabrics to retain fines (Yeri et al. 2005), it is not surprising that silt fence typically is ineffective at controlling turbidity (Barrett et al. 1996, 1998a). Turbidity is influenced primarily by fine particles with low settling velocities (Barrett et al. 1998a, Horner et al. 1990, Leytem and Bjorneberg 2005), which correspond to the particles most apt to be carried by flow through the fence fabric. Horner et al. (1990) reported only 3-percent reduction in turbidity. Barrett et al. (1998a) measured a median turbidity reduction of 2 percent, though removal during individual experimental tests ranged from -32 percent (i.e., an increase in turbidity) to 49 percent. Results from a study by Faucette et al. (2008) were unusual compared to other published literature. From four experiments of individual simulated rainfall-runoff events, silt fence consistently resulted in relatively high turbidity reductions of 45, 50, 54, and 76 percent.

The tendency of silt fence to allow fine particles to pass through the fabric also can influence other pollutant losses. Metals on roadways and in roadway runoff exist across the spectrum of sizes ranging from settleable particulates to dissolved colloidal forms; the size/form is related to the type of metal involved (Butler et al. 1992, Pitt 1979). Many metals attach to particles, of which the great majority are small diameter (Pitt 1979, Pitt and Amy 1973, Revitt and Ellis 1980, Sartor and Boyd 1972, Wilber and Hunter 1979), so it is likely that silt fence will have poor retention of metals. However, there is scant research about the effectiveness of silt fence on metal or other road-associated pollutant removal, perhaps because silt fence is designed primarily to remove settleable sediment. A single study by Büyüksönmez et al. (2012) examined "first flush" loss of metals from a single simulated runoff event. Silt fence removed 82 percent of total zinc and more than 50 percent of total cadmium, total lead, and total copper compared to a control. In contrast, for soluble forms of those metals only copper levels were reduced significantly. Loads of soluble forms of the other three metals increased compared to the control due to leaching from the silt fence material.

Far fewer studies have considered effectiveness for other types of nonreactive barriers (Table 24). Most available information about nonreactive barriers aside from silt fence involves straw or hay bales. But even these barriers have had very limited investigations of effectiveness (U.S. EPA 2002) despite their common use (Brown and Caraco 1997).

The effectiveness of straw and hay bales as barriers is greatest when they are positioned at the base of a slope to capture loose sediment (transported by gravity or small amounts of water), rather than erosion associated with overland or concentrated flow (U.S. EPA 2002). Collins and Johnston (1995) found straw bales positioned in rows downslope of urban road cuts were very effective at capturing the ubiquitous but manageable levels of sediment inputs. Results also suggested that straw bales installed to collect dispersed sources of sediment stay in place and function for longer periods than bales regularly affected by inflows of water.

In the presence of overland flow (even as sheet flow), straw and hay bale effectiveness depends largely on securely anchoring the bales to the ground (with stakes or rebar) and on keeping them wedged tightly together (U.S. EPA 2002) to resist failure and encourage sedimentation behind them. Models indicate that straw bales can be effective at sediment reduction in field use, but the results assume ideal conditions for installation and proper installation techniques (U.S. EPA 2002). Proper installation apparently is rare, however. In a visual review of field applications, Collins and Johnston (1995) found that straw bales regularly were not installed or maintained properly. As a result they were undercut or water flowed between bales, making them ineffective at controlling sediment losses. In a survey of experts on erosion and sediment control, Brown and Caraco (1997) reported inconsistencies between perceptions of straw bale effectiveness and use in the field. Only 27 percent of the surveyed

Table 24.—Sediment retention by nonreactive barriers other than silt fence

Reference	Type of erosion control barrier	Sediment retention	Comments
		percent	
Faucette et al. (2009b)	Straw bales	65.1	Total solids concentration
		71.3	Total solids load
		53.8	Total suspended solids concentration
		61.5	Total suspended solids load All replications: to simulate construction site perimeter erosion control
Kelsey et al. (2006)	6-inch excelsior fiber logs	55.2	All replications: 12.5% slope; to simulate slope interrupter and perimeter barriers; as sediment concentrations
	12-inch excelsior fiber logs	71.2	
	9-inch straw wattles	34.3	
	12-inch straw wattles	19.5	
Kouwen (1990)	Weighted or clamped-in-place straw bales	>95	Laboratory study to simulate check dams; sediment slurry composed of medium-sized sand
	Loose straw bales	70–80	
Line and White (2001)	Washed stone and rock check dam	69	Average for 34 storms; as TSS mass
McLaughlin (2003)	Large rock dam	77	Retention decreased through time
	Gravel check dam	90	
Wright (2010)	Riprap check dam	~80	As mass of sediment

experts considered straw bale barriers installed around the perimeter of construction sites to be an effective erosion control technique, yet straw bales were still allowed in more than half of the communities that were surveyed. The tradition of using straw bales for erosion control is linked to their affordability compared to other techniques (U.S. EPA 2002) rather than to proven effectiveness, as there are more data that show straw to be a far more effective BMP when applied as mulch (see Chapter 6) than when used as nonreactive barriers.

A controlled short-term study using test plots with flumes did suggest perimeter application of straw bales could remove relatively large amounts of solids when overland flows are limited if the bales were installed properly. From a single simulated storm, reductions in pollutants by straw bales around the perimeter of construction sites were 65.1 percent for total solids concentrations, 71.3 percent for total solids loads, 53.8 percent for total suspended solids (TSS) concentrations, 61.5 percent for TSS loads, and 11.8 percent for turbidity (Faucette et al. 2009b). However, performance typically is much poorer with increasing inflows. During the 1991–1992 wet season in southern California, Collins and Johnston (1995) examined straw bales installed as erosion barriers on alluvial fans (outside of channels) and at drop inlets following wildfires. Straw bales at drop inlets on roads increased erosion levels because the bales restricted water entry into the drains; water instead flowed downhill and eroded fillslopes. Decaying straw bales also clogged the inlets, resulting in the same type of consequences from overflow. On alluvial fans, straw bales were ineffective because they were displaced by overland flows that quickly overwhelmed the barriers. The bales were also susceptible to rapid decay in this situation.

Kelsey et al. (2006) studied straw-containing wattles and showed their effectiveness also was dependent upon good contact with the ground rather than the diameters of the barriers. Denser (72.62 kg m^{-3}), smaller diameter (22.9 cm) straw wattles were more effective at capturing sediment than less dense (61.2 kg m^{-3}), larger diameter (30.5 cm) wattles during 30-min applications of 10.2-cm h^{-1} simulated rain events. The greater density resulted in better contact of the barriers with the ground, making them less susceptible to rill formation and sediment transport than their larger diameter, but lighter weight, counterparts. Increased diameter of excelsior (wood fibers) logs became important only for barriers that had similar densities (Kelsey et al. 2006). Their more complex structure (curled and barbed) expanded and pieces linked to each other when wetted, such that even though the excelsior density (dry) was less than half of that of straw wattles it trapped substantially more sediment than the straw wattles (Table 24), as long as the excelsior logs were secured to the soil surface. In this test, stakes holding the excelsior logs in place were half as far apart (0.6 m) as for the straw wattles (1.2 m).

Effectiveness of barriers may be most challenged when they are employed as check dams (i.e., barriers applied to channels, gullies, ditches, or convergent topography where concentrated flow is present at least ephemerally). However, check dams generally are considered poor choices for overall erosion control because the primary role of check dams is to prevent or reduce erosion of the channel itself, and sediment capture is only a secondary objective (Collins and Johnston 1995).

Studies suggest that materials which are dislodged or displaced easily are most at risk for poor performance as check dams. In two watersheds in which 440 straw bale check dams were installed in ephemeral channels and gullies for postfire erosion control, 67 percent and 54 percent failed due to undercutting, side cutting, filling, or displacement, or a combination thereof, about 3.5 months after installation (Collins and Johnston 1995). In short-term experiments by Storey et al. (2006), bales were not displaced, but water moved under instead of through them. On sandy soils, this created substantial scouring immediately downstream of the bales, whereas on clay soils underflow remained dispersed and did not cause scouring. Kouwen (1990) found properly installed straw bales could provide high retention of sand in short-term flume studies with concentrated flows (Table 24), but the composition of the slurry mixture and duration of study do not represent realistic conditions of field applications of straw bale barriers.

Check dams constructed from rock materials, such as riprap, are more resistant to displacement and failure if the materials are the proper size. The limited data available suggest that rock check dams can perform surprisingly well, though the effectiveness probably depends primarily upon the type and grading of the coarse fragments. Over about 19 months of monitoring, a sediment trap with rock check dams in a drainage ditch of a construction site in the Coastal Plain of North Carolina had an average trapping efficiency of 69 percent, and a similarly constructed sediment trap installed in an intermittent channel draining a construction site in the Piedmont trapped an average of 59 percent of total suspended sediment (Line and White 2001). A third rock check dam installed at the outlet of a storm drain in the Piedmont was sampled for just over 2 months and had an overall efficiency of 58 percent. As with silt fence, all three sediment traps were more effective at retaining sand-sized particles compared to smaller silt- and clay-sized particles, as sands settle more easily. Silt- and clay-sized particles that were trapped tended to be associated with larger aggregates.

Wright (2010) examined 0.3-m-high trapezoidal rock (riprap) in 24:1 (horizontal:vertical) or 2.4-degree sloped field plots with natural rain events, as well as in the laboratory with simulated slopes of 6:1, 9:1, and 12:1 (horizontal:vertical) using artificial rain events. In the field study, about 80 percent of sediment was retained across three replicate rock check dams compared to losses from bare soil. The performance of the rock check dams in the laboratory was similar across all slopes and similar to the field results; they removed about 70 to 85 percent of sediment in inflow, with the best performance from the 9:1 slope (horizontal:vertical).

Standard state BMPs in North Carolina involving 1.5-ft-deep sediment traps in combination with rock check dams in road ditches were much less effective at sediment capture than combinations of stiff coir logs and malleable straw wattles (King and McLaughlin 2007, McLaughlin et al. 2009b). Average per storm sediment losses were 428 kg for the standard BMPs compared to 2.1 kg for the fiber check dams; respective average turbidity levels in outflow were 3,813 NTU and 202 NTU. When polyacrylamide (PAM) (see Soil Conditioners section in Chapter 6) was added to the coir log and straw wattle check dams, sediment retention increased even more: average per storm sediment and turbidity in outflow were 0.9 kg and 34 NTU, respectively. Similar results occurred at another site, where fiber check dams with PAM and standard state BMPs were used in road ditches, though the differences between the standard rock check dams and the fiber check dams were not nearly as great as at the other site (McLaughlin et al. 2009b). Average per storm sediment loss with the standard BMPs was 3.3 kg compared to 0.8 kg for the fiber check dam + PAM. Average per storm turbidity with the standard BMPs was 867 NTU compared to 115 NTU for the fiber check dams + PAM. Because the cost of the fiber check dams was comparable to the standard BMPs and water quality discharges were close to or below regulatory requirements, the authors recommended fiber check dams as an alternative to the sediment traps and rock check dams (McLaughlin et al. 2009b).

Check dams also are used in vegetated swales and grassed waterways to improve sediment deposition. Application in vegetated swales and waterways is described in subsequent sections of this chapter, where those biofilters are covered in greater detail.

Reactive Barriers

Filter berms and filter socks are the two types of reactive barriers currently on the market. Filter berms and filter socks physically differ from one another only in that the former are non-encased, and the latter are encased, accumulations of compost or mulch material. Thus, filter berms are simply piles or dikes of compost or mulch material used to control erosion (U.S. EPA 2010a). Filter socks are 8-, 12-, 18-, or 24-inch-diameter (Faucette et al. 2006) tubes made from mesh netting (e.g., polypropylene) that are filled with an organic compost or mulch, or both, and sealed on both ends (Faucette et al. 2009a, Shipitalo et al. 2010). During installation, filter berms and socks are laid on the ground along the perimeter of disturbed areas and along the contour at intervals (i.e., as slope interrupters) on the slope to detain sheet flow (Miller and Joaquin 2011). Filter berms are incompatible with concentrated flow (Alexander 2006, Storey et al. 2006, U.S. EPA 2012) because they are prone to washouts and failure in concentrated flow. Filter socks also may be used in vegetated waterways, swales, ditches, and similar (nonperennial) channels to control limited amounts of concentrated runoff (Alexander 2006, Shipitalo et al. 2012). Filter socks and berms are recommended for relatively steeply sloping surfaces, ranging from >4:1 to 2:1 grade (horizontal:vertical), where the likelihood for erosion is high and other types of surface erosion control may be less suited (Alexander

2006, Risse and Faucette 2001, Storey et al. 2006). Their ability to absorb substantial amounts of water allows them to remain effective on steep slopes (Risse and Faucette 2001), as long as inflows do not become concentrated.

The material in filter berms and socks may be nonmineralized (e.g., mulch), mineralized commercial or municipal organic wastes, or both (Alexander 2006). In most cases at least some portion is mineralized (i.e., humus), in which case these barriers are referred to as "compost filter berms" or "compost filter socks." However, the composts in filter berms and socks have substantially larger particle size distributions than that of composts used for planting and growing purposes (Faucette et al. 2006). The U.S. EPA provides recommendations for particle-size mixtures in its menu of BMPs for construction (U.S. EPA 2012). These recommendations essentially follow the specifications developed by the American Association of State Highway and Transportation Officials (AASHTO) (Alexander 2003).

Both filter berms and filter socks can be seeded to provide vegetative growth on the structure to help enhance stability (Storey et al. 2006). Seeding is usually done when at least part of the structure is made of compost rather than simply nondecomposed mulch, as the former provides a more suitable growing medium. Seeds can be mixed into the organic material as it is fed into the netting or before it is positioned on the ground, or they can be applied to the structure's surface after the material is positioned on the ground (Miller and Joaquin 2011; U.S. EPA 2010a, 2010b). In the case of filter socks, the mesh netting does not interfere with seed establishment because it has relatively large openings through which the seed can come into contact with the compost (Faucette et al. 2009a).

If filter socks are not seeded, they generally are used as temporary structures. When no longer needed, they are cut open, the mulch is spread around the area, and the synthetic netting is removed (U.S. EPA 2010b). When filter socks are left onsite, the netting should be biodegradable so it mineralizes in place (Miller and Joaquin 2011). Filter berms also may be left intact onsite following use, or the organic material may be dispersed around the area as a soil amendment (Storey et al. 2006).

In contrast to nonreactive barriers, reactive barriers depend only partially on ponding for sediment retention (Shipitalo et al. 2010; U.S. EPA 2010a, 2010b). As water passes through reactive barriers, chemical sorption onto the organic mulch or compost materials also is critical for sediment retention (Faucette et al. 2008)—it is this chemical adsorption that has led to characterizing these barriers as "reactive" (Shipitalo et al. 2010). The initial demonstration of the importance of filtration and sorption in compost barriers is largely credited to Keener et al. (2007). They documented slower and shallower ponding behind compost filter socks than behind silt fence even though the filter socks retained approximately the same amounts of sediment as silt fence (Faucette et al. 2008). It is believed that some small particles are filtered out when water passes by the sorption sites of larger sediment particles that previously have been adsorbed onto the filter media in the barrier (Demars and Long 2001). Faster flow-through rates with comparable or greater retention by reactive barriers means that they do not have to be as tall as silt fence to accommodate a given nonconcentrated inflow rate (Faucette et al. 2006, Keener et al. 2007).

Filtering and sorption occur by the combination of physical retention of solids among organic particles (i.e., blinding and clogging) and by chemical adsorption of charged particles onto the organic material in berms and socks (Faucette et al. 2009a). The humus fraction of compost provides both positively and negatively charged exchange sites (Faucette et al. 2006), both of which adsorb oppositely charged particles contained in runoff (Faucette et al. 2006). This ability is very important for retaining small particles that are not well retained by traditional nonreactive barriers. Clay- and silt-sized particles are difficult to retain without adsorption because they have very low settling velocities and can pass through porous barriers. Clay particles have net negative charges (Brady 1984), so they can be adsorbed by positive sites in filter media (Faucette and Tyler 2006). Silt particles do not possess a charge per se, but organic matter bound to them can provide positive or negative charges (Schafer 2008) for adsorption onto oppositely charged sites in filter berm or filter sock media. Faucette et al. (2009a) found filter socks removed 65 percent of both clay- and silt-sized particles (< 2-µm and 2- to 50-µm diameter, respectively); 60 percent of 0.01- to 5.75-µm-diameter particles, and 80 percent of 5.75- to 19.95-µm-diameter particles, were removed.

For both filter berms and filter socks, the three-dimensionality of the organic media and the mixture of organic particle sizes play important roles in filtration (Faucette et al. 2006, 2009b; U.S. EPA 2012). The open design of the sock mesh itself does not impede water entry into the organic material (Faucette et al. 2009a, 2009b). But too many small particles in a berm or filter sock hinder flow through the structure, reducing the sediment control effectiveness; conversely, too many large particles will not effectively trap sediment particles (Storey et al. 2006). Optimally, the three-dimensionality and the particle-size mixture allow runoff to flow through the media along tortuous pathways, thereby increasing the surface area and spatial area of compost contacted by inflow to encourage filtration and sorption (Faucette and Tyler 2006).

Ion exchange and filtration are a function of time of contact (Shipitalo et al. 2010), so it is not surprising that the amount of suspended solids retained by filter socks has been shown to be inversely related to flow-through rates (Faucette et al. 2006). Based on a review of particle sizes used in compost mixtures for sediment control, Faucette et al. (2006) concluded that smaller particle size distributions (i.e., <0.25 inch) of compost filter media remove more fines from runoff than larger filter-media particles. Thus, if pollutant removal by a given mixture is found to be inadequate, a greater portion of smaller particles can be added to improve retention; these particles can be added to the surface of the barrier (Demars and Long 2001). However, smaller particle sizes result in slower flow-through rates, which can lead to overtopping of filter socks and berms if inflow contains high concentrations of sediment that clog large amounts of voids and reduce flow-through rates (Faucette et al. 2006). Clogging and reductions in flow-through rates also can occur as increasingly more small particles from runoff are retained (Demars and Long 2001). Taller berms or larger diameter socks may compensate for the reduced hydraulic flow rate, so these factors should be considered in the design of the barriers (Faucette et al. 2006).

Most filter berm studies have been flume or laboratory studies, or short-term field studies on small plots. Reported removal rates of solids from these studies have been variable. About 92 percent of total settleable solids and 96 percent of TSS were removed by using filter berms constructed of mixed-material residential yard wastes on 32-ft-long slopes of 34-percent grade during five rain events (Ettlin and Stewart 1993). From a single simulated storm event, Faucette et al. (2009b) observed that a compost filter berm reduced total solids concentrations and loads by 54.8 percent and 63.5 percent, respectively, and reduced TSS concentrations and loads by 51.3 percent and 60.4 percent, respectively. With an inflow sediment concentration of 340 g L^{-1}, Demars et al. (2000) found that 99 percent of the total solids were removed with a compost medium having a particle size distribution adhering to U.S. EPA specifications. With higher inflow concentrations of sediment (500 g L^{-1}) two compost particle size distributions not meeting EPA specifications also removed 99 percent of total solids. In a separate study by Faucette et al. (2005), more than doubling the sediment concentration (to 1,200 g L^{-1}) did not reduce the removal efficiency (98-percent efficiency) of a filter berm containing compost medium with a particle size distribution not meeting EPA specifications. In contrast, another compost medium that did not meet EPA specifications subjected to an inflow sediment concentration of 500 g L^{-1} removed only 20 percent of total solids (Demars and Long 2001). The latter compost mixture had the lowest percentage (6 percent) of compost particles capable of passing through a 0.25-inch mesh compared to 18 to 93 percent of particles in the other studies; therefore, it could not effectively trap/sorb sediment particles (Storey et al. 2006).

Physical failure of filter berms results in runoff bypassing much of the organic medium, making them ineffective at removing pollutants. Storey et al. (2006) found 14 of 15 unseeded filter berms failed within 16 min of exposure to 0.25 ft^3 s^{-1} inflow rates on a 3-percent slope (i.e., simulating a shallow concentrated runoff). Each berm was constructed to Texas Department of Transportation specifications by using one of three different materials: dairy manure compost, yard waste compost, or biosolid compost. Longitudinal displacement of berm materials and berm breakthroughs were common on clay soil, whereas berms on sandy soil were susceptible to undercutting of the soil on which the berm was constructed after runoff infiltrated into the soil behind the berm. Seeding appeared to have increased stability and decreased failures; the filter berms had been seeded 45 days before the introduction of runoff although there was little evidence of vegetative growth on the composted yard waste berms at the time of experimentation. All of the berms were capable of structurally withstanding three runs with inflows of 0.35 $ft^3\ s^{-1}$ (Storey et al. 2006), even though these inflow rates resulted in overtopping the berm and were higher than inflow rates that caused failures of the unseeded berms. There are, however, few other data to suggest that seeding provides more stable berms or more effective pollutant removal.

The quality of compost and mulch in filter berms and socks also has consequences for the quality of the water flowing from these barriers. Leaching from the organic material composing the berm or sock can result in increased levels of TSS and total dissolved solids in water flowing from the barriers compared to inflow waters. This was the case during "first flush" effects for berms constructed of composted yard wastes, composted dairy manure, and composted biosolids (Raut Desai 2004, Storey et al. 2006). The yard waste compost resulted in the highest TSS in flow-through water, but leached the least amount of nutrients. The authors noted that the leachate concentrations were insufficient to have caused a problem in receiving waters, but in practice the outcome may be different with actual storm inputs. Consequently, the quality of compost materials is an important consideration and the reason that media for filter berms and socks should meet physical and chemical specifications (Alexander 2006). Criteria for compost quality are changing as research becomes available (Alexander 2006).

Filter socks provide a significant design improvement over filter berms because the encasement of organic materials makes the barriers less susceptible to failure (Raut Desai 2004). Almost all reports of sediment capture by filter socks originate from laboratory studies, but they consistently show good to excellent performance. Tests of 10 compost products from commercial and municipal composting operations in the United States, Canada, Japan, and New Zealand showed that 4 of the products removed 100 percent of total solids and the remaining 6 products each removed more than 95 percent of total solids in single test runs (Faucette and Tyler 2006). Through three consecutive runs, the average removal of TSS was 71 percent. The rate increased from 58 percent to 69 percent to 84 percent with each successive run, showing that the ability to retain particles was not diminished through the short term by repeated sediment-laden inflows.

A comparison of 12- and 18-inch-diameter filter socks containing identical mixtures of compost materials found that both removed an average of 70 percent of total suspended sediment (Faucette et al. 2006). Another comparison of 8- and 12-inch-diameter compost filter socks showed average TSS concentration removal efficiencies of 75.9 percent and 71.4 percent, respectively, for a single simulated runoff event (Faucette et al. 2009b). When expressed as loads, effectiveness was higher by about 10 percent, at 83.9 percent and 84.9 percent for the 8- and 12-inch socks, respectively. The percentages of total solids removed were similar to these values. Removal efficiencies for total solids concentrations were 76.3 percent and 72.7 percent and for loads, 84.3 percent and 85.0 percent, respectively, for 8- and 12-inch-diameter filter socks. In a bench study by Faucette et al. (2008) that used several different runs with different compost media (not all particle-size mixtures met AASHTO specifications), compost filter socks removed 62 to 87 percent of TSS. Sadeghi et al. (2006) found almost identical removal efficiencies during tests of six types of filter socks. Total suspended solid loads were reduced by 68.3 to 89.7 percent.

The ability of filter berms and filter socks to filter and sorb clay and silt particles can contribute to controlling turbidity, but the actual effectiveness is largely dependent upon the chemical constituency of the organic materials used in the barrier. Faucette et al. (2009b) observed turbidity reductions of 8.1, 28.6, and 19.1 percent, using a mulch filter berm, an 8-inch-diameter filter sock, and a 12-inch-diameter filter sock, respectively, after a single storm. Using bench studies, Sadeghi et al. (2006) reported turbidity reductions by compost filter socks of 52.5 to 77.8 percent. Turbidity was reduced an average of 24 percent across 45 different compost media in filter socks, and even with high inflow, a filter sock reduced turbidity by 21 percent (Faucette et al. 2006).

The addition of flocculating coagulants and polymers (including PAM; see the Soil Conditioners section in Chapter 6 for more information on these chemicals) can enhance retention of sediment by filter socks and berms. Flocculants and polymers can be added internally or externally to filter berms and socks. In a comparison of compost filter socks with and without flocculating and coagulating agents, Faucette et al. (2008) reported TSS removals in the range of 91 to 97 percent compared to 62 to 87 percent without the chemical agents. When identical particle-size mixtures were used in all tests, TSS removal remained at 91 to 97 percent with sediment-targeting flocculating agents but was 78 percent without polymers. Sadeghi et al. (2006) reported similar high levels of removals of TSS loads by compost filter socks with polymers, ranging from 94 to 98 percent.

However, the benefit provided by polymers is more apparent for turbidity. Sadeghi et al. (2006) reported turbidity reductions of 79 to 98 percent by compost filter socks with polymers. Faucette et al. (2006) found that 12- and 18-inch-diameter filter socks with PAM or polysaccharide polymers reduced turbidity by averages of 74 percent and 84 percent, respectively. The larger reductions with the larger diameter socks were attributable to the greater distance that runoff had to travel within the barrier, which resulted in greater contact with the compost medium within the sock.

Polymer additions may be most useful when the barrier is designed to handle large inflows for which some degree of sediment retention otherwise may be sacrificed (Faucette et al. 2006). Under high flow situations, turbidity was reduced by up to 90 percent with PAM and up to 77 percent when the polysaccharide polymer was applied to the media in an 18-inch-diameter filter sock, compared to 58 percent with no polymers (Faucette et al. 2006). Thus, compost filter socks, especially in combination with polymer flocculants or coagulants, may provide an effective alternative BMP in situations where turbidity-based water-quality standards are not attainable with barriers unable to retain suspended fines. The longevity of polymer effectiveness has not been studied with filter socks or berms (Faucette et al. 2006), and it is unknown whether the duration of efficacy is similar to that found when polymers are applied to the soil surface for erosion control (see Chapter 6).

The charged nature of compost also provides opportunities for sorption of other ions and chemicals by filter berms or filter socks. These constituents include nutrients (e.g., phosphorus), bacteria, herbicides, metals, and petroleum products, though removal only of petroleum compounds and metals is discussed here as these materials are the most often associated with road construction and use. Papers by Faucette and Tyler (2006), Faucette et al. (2006, 2008, 2009a), Shipitalo et al. (2010), and Storey et al. (2006) provide information on the efficacies associated with nonmetal and nonpetroleum types of chemicals and organisms.

Only a couple of studies have examined the efficacy of filter berms or filter socks in removing petroleum compounds or metals. The existing data suggest that both types of materials can be removed from runoff with berms and socks, but petroleum products are captured more effectively. In individual test runs, six of seven compost mixtures in filter socks removed at least 96 percent of motor oil and the seventh removed 85 percent (Faucette and Tyler 2006). With three consecutive runs, seven filter socks removed nearly 100 percent of the motor oil. Tests of 45 organic media in filter socks had a similar high average removal rate (89 percent) for motor oil (Faucette et al. 2006), and an average of 84 percent in another study using U.S. EPA-approved compost medium in 3 replicate filter socks (Faucette et al. 2009a). Average diesel fuel removal was 99 percent, whereas gasoline capture averaged only 43 percent (Faucette et al. 2009a). Additions of a petroleum flocculating agent had a negligible effect on the amounts of petroleum chemicals removed (Faucette et al. 2009a). The flocculating agent increased removal of the motor oil to 99 percent and removal of gasoline to 54 percent, but neither was significantly higher than without the flocculating agent. Diesel fuel capture with the flocculating agent stayed at 99 percent.

Faucette et al. (2009a) reported moderately effective capture of metals with filter socks, though there was little increase in effectiveness when a metal flocculating agent was added externally to the socks (Table 25). Additionally, specific forms of some metals were poorly retained; only 17 percent and 29 percent of soluble chromium was retained with and without the flocculating agent, respectively, and less than 50 percent of sediment-bound copper was captured regardless of use of the flocculating agent.

Table 25.—Percentage of metals in inflow removed by compost filter socks alone and compost filter socks in combination with a metal flocculating agent applied externally to the socks (data from Faucette et al. 2009a)

Erosion control	Constituent form[a]	Cadmium	Chromium	Copper	Nickel	Lead	Zinc
		percent					
Filter sock only	Soluble	64	17	68	61	72	53
	Sediment-bound	73	75	42	60	70	64
	Total	64	37	67	61	71	54
Filter sock + metal flocculant	Soluble	72	29	70	69	79	57
	Sediment-bound	77	78	45	63	61	47
	Total	73	47	70	69	73	53

[a] Inflow inputs included 500 ml of 15 ppm concentrations of each metal.

Comparative Studies of Nonreactive and Reactive Barriers

It is difficult to compare the effectiveness of nonreactive and reactive barriers as determined in different studies, even qualitatively, because of the myriad differences in conditions that could influence overall performance. However, a few papers have compared different types of reactive and nonreactive barriers, so the opportunity is taken here to review these studies. They tend to have little, if any replication, but they do provide insight into performance with identical inflows, sediment concentrations, slopes, and other variables.

Results from Demars et al. (2000) may be the most applicable to field use as the tests were longer term and involved natural rain events. They compared silt fence, a hay bale berm, and a compost (wood waste) filter berm during 11 events of varying intensity and magnitude over about 5 months. Compared to controls of bare ground plots with no erosion control, all three erosion control techniques performed well on 2:1 (horizontal:vertical) slopes; the compost filter berm performed best, followed by the silt fence and hay bale berm (Table 26).

Keener et al. (2007) reported nearly the same level of performance for both compost filter socks and silt fence (Table 26), but compost filter sock effectiveness was mediocre compared to most other filter sock studies (e.g., those described previously). Keener and colleagues found compost filter sock effectiveness to be consistently less than 50 percent, even for low inflows.

Faucette et al. (2009b) compared two different diameter filter socks with and without polymer additives, a mulch filter berm, and a straw bale barrier. All barriers significantly reduced both concentrations and loads of total solids and TSS (Table 26). The compost sock treatments retained significantly more total solids and TSS (concentration and load) than either the mulch filter berm or straw bale barrier. The filter berm and straw bale also were less effective than all other treatments at reducing turbidity (Table 26); the addition of the polymer to the filter socks significantly improved their ability to reduce turbidity (Faucette et al. 2009b). The TSS removal effectiveness reported by Faucette et al. (2005) for mulch filter berms was much better than the aforementioned study, and silt fence performance in the 2005 study also was much better than many other studies (Tables 26 and 23).

Ettlin and Stewart (1993) compared the effectiveness of filter barriers against silt fence placed at the base of 32-ft-long plots on 34-percent slopes on a closed landfill in Oregon. The filter barriers were composed of mixed types of yard debris compost. They removed substantially more settleable solids and TSS for five events (1.6 inches of rainfall) than the silt fence. Final results for settleable solids and TSS were 2.9 mg L^{-1} and 1,300 mg L^{-1}, respectively, for the filter barriers

Table 26.—Comparisons of different types of nonreactive and reactive barriers within individual studies

Reference/Comments	Type of erosion control barrier	Removal efficiency				Turbidity reduction
		Total solids concentration	Total solids load	TSS[a] concentration	TSS load	
		percent				
Demars et al. (2000)	Geosynthetic silt fence				98.4	
11 natural rain events of varying intensity and duration	Hay bale				98.0	
	Wood compost filter berm				99.8	
Faucette et al. (2009b)	8-inch-diameter compost filter sock	76.3	84.3	75.9	83.9	28.6
	12-inch-diameter compost filter sock	72.7	85.0	71.4	84.9	19.1
	8-inch-diameter compost filter sock + polymer	77.1	86.3	75.8	84.7	49.1
	12-inch-diameter compost filter sock + polymer	80.7	88.2	83.1	89.5	41.8
	Mulch filter berm	54.8	63.5	51.3	60.4	8.1
	Straw bale	65.1	71.3	53.8	61.5	11.8
Faucette et al. (2008)	8-inch-diameter compost filter socks with various particle size distributions			62–87		53–78
	8-inch-diameter compost filter sock + polyacrylamide			91		79
	8-inch-diameter compost filter sock + polymer			97		94
	8-inch-diameter compost filter sock + copolymer			97		98
	Silt fence			63–87		

(continued)

Table 26.—Comparisons of different types of nonreactive and reactive barriers within individual studies

Reference/Comments	Type of erosion control barrier	Removal efficiency				
		Total solids concentration	Total solids load	TSS[a] concentration	TSS load	Turbidity reduction
		percent				
Faucette et al. (2005)	Mulch filter berms		96			
	Silt fence		95			
Keener et al. (2007) Flume study with 30-min flows for each structure	8-inch-diameter compost filter sock with 0.126 L s^{-1} inflow	42.8[b]				
	8-inch-diameter compost filter sock with 0.252 L s^{-1} inflow	20.4[b]				
	8-inch-diameter compost filter sock with 0.315 L s^{-1} inflow	30.0[b]				
	12-inch-diameter compost filter sock with 0.126 L s^{-1} inflow	50.0[b]				
	12-inch-diameter compost filter sock with 0.252 L s^{-1} inflow	43.1[b]				
	12-inch-diameter compost filter sock with 0.315 L s^{-1} inflow	26.1[b]				
	24-inch-width silt fence with 0.126 L s^{-1} inflow	56.0[b]				
	24-inch-width silt fence with 0.252 L s^{-1} inflow	41.9[b]				
	24-inch-width silt fence with 0.315 L s^{-1} inflow	32.7[b]				
Sadeghi et al. (2006) Bench study	Compost filter socks				68.3–89.7	52.5–77.8
	Compost filter socks + polymer				94.0–98.2	79.2–98.0
	Silt fence				71.5–89.1	44.8–76.0

[a] TSS = total suspended solids.

[b] Estimated from Figure 10 in Keener et al. (2007).

Empty cells indicate variable was not measured.

compared to 32 mg L^{-1} and 26,000 mg L^{-1}, respectively, for silt fence. These silt fence values were similar to controls, where no erosion control devices were installed (34 mg L^{-1} and 31,000 mg L^{-1}, respectively).

Biofilters

Biofilters take many different forms, but are always composed of vegetation. While the vegetation itself contributes to erosion control through soil stabilization and soil cover (see Chapter 6), the primary role of biofilters is to stop sediment transported by water from reaching downslope water bodies. This is achieved through sediment deposition, by slowing water, encouraging water infiltration, or both. For some types of biofilters, such as forest buffer strips, other features that provide roughness at the soil surface are critical for limiting sediment transport.

There tends to be inconsistency in the terms "length" and "width" in biofilter literature. Consequently, to avoid confusion, the terms used in most of the literature are used here; that is, in this chapter the direction along the contour is referred to as the "width" and the downslope direction of flow is referred to as the "length." Note that this is opposite of the terminology typically used for buffer strips in forestry literature, where "width" denotes the downslope distance.

Buffer Strips

Buffer strips are a major category of vegetation-based pollution controls. They are designed to remove primarily sediment (Correll 1996, Dillaha et al. 1989, Hayes et al. 1984, Magette et al. 1989), though they also have some ability to remove nutrients and other chemical constituents (Dillaha et al. 1989, Magette et al. 1989). They differ from vegetated swales and waterways (described later) in two distinct ways. First, buffer strips are not channels, so they are not designed to be submerged; and second, their largest dimension (i.e., width) runs approximately along the contour, perpendicular to the direction of runoff that is cast onto them (Deletic and Fletcher 2006).

The two types of buffers described in this chapter are vegetated filter strips (VFS), and forested buffer strips. VFS have most commonly been associated with agricultural applications, but their use in urban areas and along or in the medians of roadways has been increasing. However, their effectiveness for these latter areas has not been studied as intensively as for agriculture (Deletic 2005). They typically are composed of herbaceous cover, primarily grasses, although in agricultural fields they may consist of agricultural crops (Hayes et al. 1984).

Forest buffers are used in forests for a variety of reasons in addition to sediment trapping, including aquatic habitat protection such as shade and water body bank stabilization. For sediment control, forest buffers differ from VFS in several ways. They predominantly depend upon overstory vegetation and resultant litter to provide infiltration and filtration mechanisms, and they tend to be longer than VFS. Typical VFS might be 5 to 15 m long (e.g., Abu-Zreig et al. 2003, Daniels and Gilliam 1996, Dillaha et al. 1987, Schmitt et al. 1999), whereas typical forest buffer length in the United States and Canada ranges from 15 to 30 m (Lee et al. 2004). Additionally, VFS are positioned immediately adjacent to or are interspersed within the potential pollutant source (e.g., a roadway or an agricultural field, respectively); a forest buffer is located adjacent to the water body it is intended to protect. Forest buffers and VFS positioned along water bodies commonly are called riparian buffers. However, when a forest road is located sufficiently close to a water body, the forest buffer also may be adjacent to the road and span the entire distance between it and the water body.

VFS and forest buffers very frequently are collocated along forest roads: the area immediately adjacent to the road (typically the fillslope) has grass or herbaceous vegetation that serves as a VFS, which leads into a forest buffer farther downslope. However, most forest buffer literature makes little distinction between the VFS and the forest buffer, and both features are collectively referred to as the "forest buffer." Only a few papers have examined the importance of the VFS separately from the adjoining forest buffer, so to ensure that these papers are described, they are discussed at the end of the VFS subsection. The Forest Buffers and Windrows subsection includes the more conventional review of forest buffers, and discusses them only in relation to water quality (primarily sediment). Even though forest buffers provide a myriad of other benefits (e.g., for wildlife), the attributes that make them effective for protecting water quality do not necessarily make them effective for other purposes (Richardson 2004). Readers interested in other facets of forest buffers are directed to papers by Barling and Moore (1994) and Belt et al. (1992).

Forest buffers used in an agricultural setting are a specialized, third type of buffer. They are composed of trees, sometimes in association with other herbaceous, grass, or shrub vegetation, positioned along waterways. Their purpose is to protect waterways from sediment, nutrients, pesticides, and other chemicals in runoff originating from upslope agricultural areas. Because the high nutrient and chemical loads present in agricultural runoff are not associated with forest road construction or use, these buffers are considered only from the context of sediment reduction. Readers who are interested primarily in forest buffers from the perspective of other nutrients and chemicals may review papers by Peterjohn and Correll (1984), Corley et al. (1999), Schmitt et al. (1999), Lee et al. (2003), and Schoonover et al. (2005) as a starting point for coverage of these topics.

Vegetated Filter Strips

There are two primary types of VFS used for erosion control: conventional strips that are composed of relatively common types of flexible vegetation, such as fescue (*Festuca* spp.), ryegrass (*Lolium* spp.), bluegrass (*Poa* spp.), and orchardgrass (*Dactylis glomerata*) (Abu-Zreig et al. 2004), and those composed of stiff, erect (and usually tall) vegetation (Blanco-Canqui et al. 2004). The term "VFS" often is used for both, but the stiff grasses are distinct from conventional VFS (Dabney et al. 1993) and are often specifically referred to as "grass hedges" or "grass barriers" (Blanco-Canqui et al. 2004, Meyer et al. 1995). Switchgrass (*Panicum virgatum*), which is native or has become naturalized in much of the United States, along with nonnative tropical grasses, especially vetiver (*Chrysopogon zizanioides*, formerly known as *Vetiveria zizanioides*), are commonly employed in stiff grass hedges (e.g., see Dabney et al. 1995, Dalton et al. 1996, Desbonnet et al. 1994, Owino and Gretzmacher 2002, Shariff 2000, Truong 2000).

Conventional VFS and grass hedges have some important differences but also share many similarities. The most important similarity is that they both depend upon the vegetation in the strip as the key for providing sediment control. The vegetation provides roughness, which reduces energy and transport capacity (Deletic and Fletcher 2006, Gumiere et al. 2011, Hösl et al. 2012) by decreasing the speed of flow to encourage sediment deposition, filtering, and infiltration (Prosser et al. 1995). The abatement of water's shear stress by vegetation also reduces soil scour within the filter strip (Prosser et al. 1995).

The ability to provide these benefits across a variety of inflows depends in large part upon the flexibility or stiffness of the plants in the filter strip. That is, energy control is derived primarily from the roughness of the vegetation, which is directly related to the flexibility or stiffness of the plant and stem density. Flexible grasses, even when dense, have less strength and lower moduli of elasticity (i.e., less ability to return to their original shape when deformed by a force) than stiff grasses. As a result they will bend and fail in the presence of high velocities, submergence, or sediment inundation (Dillaha et al. 1982, Kouwen et al. 1981) more easily than stiff grasses, which can withstand much greater inflow rates, water depths, and sediment weights (Dillaha et al. 1982, Dunn and Dabney 1996, Kouwen et al. 1981). In a prone position, grass blades provide substantially less roughness and energy dissipation than when upright (Fiener and Auerswald 2006, Kouwen and Unny 1973) and hence little ability to retain sediment. For example, Meyer et al. (1995) found that fescue grass failed under a range of tested inflow rates and was only marginally effective at trapping sediment. In contrast, stiff grasses can become fully inundated and overtopped with water and still withstand bending or breakage (Boubakari and Morgan 1999). Consequently, even though the water column above the top of the plants may have high velocity and shear stress, the shear velocities within the plant's (upright) height are much lower, allowing sedimentation to occur (Pethick et al. 1990, Prosser and Dietrich 1995).

Because flexible plants have limited strength and moduli of elasticity, conventional VFS are effective only when inflow occurs as sheet flow (Barfield et al. 1979; Blanco-Canqui et al. 2004; Dillaha et al. 1986, 1989; Hösl et al. 2012). Sediment removal rates exceeding 70 percent have been reported with conventional VFS for sheet flow (Dillaha et al. 1986, 1988, 1989; Neibling and Alberts 1979), but as inflow rates increase they tend to become less effective (Dabney et al. 1995, Gharabaghi et al. 2006, Meyer et al. 1995).

The strength and density of most grasses used for hedges (i.e., typically stiff, erect grasses) allow them to be employed in concentrated flow in rills, and even ephemeral gullies (Blanco-Canqui et al. 2004, Dabney et al. 1993, Van Dijk et al. 1996). However, even though grass hedges can retain sediment in concentrated flow, they still perform best when subjected only to sheet flow (Xiao et al. 2010) as deposition, filtering, and infiltration depend upon the ability to spread water out (Prosser et al. 1995).

Even though the means for energy dissipation and particle settling depend upon vegetation's roughness, a substantial portion of sediment deposition ironically occurs before (i.e., upslope of) the leading edge of the vegetated strip. In most applications, conventional VFS and grass hedges receive inflows from areas with little roughness (e.g., agricultural fields, feed lots, roads, urban areas), so when inflow reaches a strip, water spreads out along the contour and builds up in response to hitting the "plant wall" (Blanco-Canqui et al. 2004, Dillaha et al. 1987, Neibling and Alberts 1979). The wall slows the water and promotes sedimentation behind the leading edge of the strip (Boubakari and Morgan 1999, Spaan et al. 2005). For grass hedges this is typically referred to as "ponding" because the water may become deep. For conventional VFS the term "ponding" may be used, but the more subtle term "back water" is sometimes favored (e.g., Loch et al. 1999) to denote the generally lower amount and depth of water because incoming water should be as sheet flow.

Ponding and back water development are critical to sedimentation because settling is inversely related to flow velocity (Spaan et al. 2005), and slowed water results in greater sediment deposition than would occur from the presence of the vegetation alone, even if all inflow occurs as sheet flow. Detention of water behind the vegetative wall also helps encourage infiltration (Blanco-Canqui et al. 2004, Deletic 2001), which contributes to sediment settling.

Ponded water and back water are concentrated water, but they differ from concentrated flow in that ponded water or back water moves into the VFS as sheet flow (Loch et al. 1999) and not in discrete pathways or rills. Flow delivered as concentrated flow can become ponded and transformed to sheet flow if the density and stiffness of stems or grass blades are sufficient to back water up and disperse it laterally without plants failing (Blanco-Canqui et al. 2004).

Ponding, infiltration, and sediment trapping efficiencies of VFS are directly related to the density of vegetation (Boubakari and Morgan 1999, Polyakov et al. 2005, Spaan et al. 2005). Sparse or open vegetation may have little influence on slowing velocities, backing up water, and promoting sediment settling. Depending upon the characteristics of the plants involved, sparse vegetation can sometimes increase the velocity of water moving through the vegetated strip by concentrating water into the spaces between plant stems or bunched plants (Spaan et al. 2005). As the water is compressed through smaller openings that have little roughness, velocities increase, thereby negating the potential for settling while increasing the tendency for rill erosion (Boubakari and Morgan 1999, De Ploey et al. 1976).

Ponding or back water can be enhanced by positioning the VFS (including grass hedges) at or slightly above the elevation of the upslope contribution lands and installing the strip on a gentler slope or with an inverse slope compared to the contributing area. Slope control techniques allow water to slow and pile up more than if the strip simply was positioned adjacent to and on the same slope gradient as the contributing area (Deletic 2001), thereby encouraging sheet flow, settling, and infiltration to occur. Sediment capture typically is strongly related to the slope of a VFS (conventional or grass hedge). Dillaha et al. (1989) reported that sediment retention was inversely related to slope for VFS gradients ranging from 5 to 16 percent. Sediment concentrations in runoff from the outflow of a VFS on a 12-percent slope were twice as high as those from a 7-percent slope (Robinson et al. 1996). Xiao et al. (2010) found a slope of 20 percent resulted in a 54-percent reduction in runoff compared to a 77-percent reduction with a 5-percent slope. Even under concentrated flow, greater sediment retention has been documented on gentle slopes (5 percent) compared to sheet flow on steeper slopes (11 percent and 16 percent) (Dillaha et al. 1989).

However, care must be used when ponding is enhanced to encourage sediment deposition. Even if inflow occurs as sheet flow, there is a maximum degree of inundation that can occur before the vegetation becomes ineffective at controlling the factors that contribute to pollution reduction, especially when the VFS is dominated by flexible vegetation. When water becomes too fast or deep, hydrostatic, hydrodynamic, and sediment weight forces can cause herbaceous plants to bend and fail (Dunn and Dabney 1996). The modulus of elasticity and the strength of switchgrass are four and three times, respectively, that of fescue, which explains why switchgrass has greater resistance to bending or breaking than fescue (Dunn and Dabney 1996). But even stiff-bladed plants will reach a point of failure at some level of water velocity or depth (Dillaha et al. 1989, Kouwen et al. 1981). Failure of vegetation in a VFS not only substantially reduces sediment deposition, but can also lead to scour within the VFS because preferential flow paths and rills can develop where vegetation has failed (Blanco-Canqui et al. 2004).

Ponding and settling also have the potential to create sediment berms at the front edge of the vegetated strip. Berms can retard movement of sheet flow into conventional VFS and grass hedges and can cause breakthroughs at weak points by the water stored behind them. Concentrated flow paths develop at the breakthroughs (Barrett et al. 1998b, Parsons et al. 1994), which can create cascading problems of preferential flow and scour pathways (Barrett et al. 1998b). Routine removal of these berms (Pankau et al. 2012) or installation of longer berms to encourage dispersion of concentrated flow to sheet flow over the long term (Parsons et al. 1994) may be necessary. This may be difficult to do effectively, however, because berm breakthroughs with measurable erosion have been observed even where only very small berms exist (including those less than a few centimeters in height) (Pankau 2010, Pankau et al. 2012).

Studies suggest that sediment deposition within a conventional VFS or grass hedge does not cause the same problems as berm formation at the leading edge. Sediment deposited in either type of VFS does not simply lie on the soil surface, but instead tends to become incorporated with the soil surface (Barrett et al. 1998b, Dillaha et al. 1989). Vegetation grows through it, and the roots and stems help bind captured soil and reduce its potential for resuspension.

Sedimentation is a function of the velocity of surface flow, so it is not surprising that large particles are removed most effectively by vegetated strips, as these settle out most easily (Neibling and Alberts 1979, Zanders 2005). In a flume study performed by Meyer et al. (1995), vetiver and switchgrass hedges trapped more than 90 percent of sediment greater than 125-µm diameter. The percentages that were retained declined as sediment size decreased, with only about 20-percent capture of particles smaller than 32-µm diameter. Pan et al. (2010) found that grass strips removed 30 percent more 10- to 25-µm-diameter particles than smaller fines. Gharabaghi et al. (2006) observed more than 95-percent retention of sediment particles larger than 40-µm diameter in runoff compared to about 65 percent for particles smaller than 12-µm diameter. Deletic and Fletcher (2006) found that VFS removed almost all particles from 57- to 180-µm diameter, but there was almost no reduction in particles less than 5.8-µm diameter. It should be noted that the percentage of fine clay and silt particles captured can be significant if soils are well aggregated and may contribute to the high retention rates just reported for larger diameter classes.

The slope of a VFS also influences the retention of large particles; as the slope increases, a smaller percentage of large particles is captured. VFS retained about 92 percent of particles greater than 50-µm diameter on 3-percent slopes compared to 75 percent on 15-percent slopes (Pan et al. 2010). For particles less than 10-µm diameter, about 46 percent was retained on 3-percent slopes and about 25 percent was retained on 15-percent slopes. Although the difference between retention on the gentle and steep slopes was only slightly greater for the small particles, the influence of steepness is of particular concern for small particles because they have an overall lower potential to be retained at all gradients.

Increasing the grade of the VFS decreases the effective length of the VFS; that is, a greater flow length is required to achieve the same sediment trapping efficiency (Pan et al. 2010), especially for small particles (Barfield et al. 1979, Line 1991). Increasing sediment concentrations and inflow rates similarly decreases the effective flow length. In some situations, flow rates and sediment concentrations or loads can be controlled by reducing the overall size of the contributing area relative to the area of the VFS (Magette et al. 1989, Van Dijk et al. 1996). If variables that control the effective length cannot be altered or adjusted sufficiently, the length of the VFS may have to be increased (Deletic and Fletcher 2006), or additional BMPs, such as adding a mulch soil cover to all or part of the contributing area (Xiao et al. 2010), may be warranted. In all situations, techniques that encourage complete or nearly complete infiltration of inflow are required to retain very fine particles (Gharabaghi et al. 2006).

Although increasing the length of the VFS encourages greater sediment retention, a maximum length apparently exists for each field condition beyond which there is little advantage, in terms of sediment capture, to further increasing the length (Gharabaghi et al. 2006). The optimal flow length required to meet sediment retention goals may be fairly short for both conventional VFS and grass hedges because: 1) most sediment retention occurs in the back water area and first few meters of a VFS (Gharabaghi et al. 2006, Line 1991, Meyer et al. 1995, Neibling and Alberts 1979, Van Dijk et al. 1996), and 2) grass hedges are much shorter than conventional VFS (Dabney et al. 1993), yet they are reported to be more effective than VFS (Blanco-Canqui et al. 2004).

No rigorous, replicated experiments have been performed to identify VFS flow length guidelines for specific sets of environmental conditions, but 10 m may be sufficiently long to capture the majority of retainable sediment (Liu et al. 2008). Increasing the length from about 5 m to 10 m generally results in some improvement, though it can vary widely from almost no change in sediment retention to increases up to 30 to 40 percent (Dillaha et al. 1987, 1989; Magette et al. 1989; Van Dijk et al. 1996) (also see Table 27). Van Dijk et al. (1996) found that 90 to 99 percent of incoming sediment was removed within 10 m. Schmitt et al. (1999) found little improvement in sedimentation between 7.5 m and 15 m in three types of VFS composed of mixed grass, mixed grass and trees/shrubs, and sorghum (*Sorghum bicolor*), suggesting that extending VFS lengths up to 15 m provides little benefit to sediment retention.

The greatest advantage to using VFS lengths of a full 10 m may come in situations where sediment in runoff is dominated by fines (Gharabaghi et al. 2006), as capture of these particles (<0.45-µm diameter) is by infiltration into soil pores (Gumiere et al. 2011). Doubling VFS length from about 5 m to 10 m approximately doubled infiltration rates (Dillaha et al. 1987). Neibling and Alberts (1979) also showed that clay concentrations in outflow from bluegrass VFS were directly related to strip length. More than 90 percent of the particles larger than 20-µm diameter were captured by strips as short as 0.6 m long compared to only 37 percent of the clay fraction. Increasing the VFS length to 1.2, 2.5, and 4.8 m substantially increased the clay fraction retention rates to 78, 82, and 83 percent, respectively.

Most studies show conventional VFS and grass hedges are reasonably effective at reducing sediment in runoff (Table 27). However, almost all of these studies have been nonreplicated or poorly replicated, or the data are observations from simple monitoring efforts (Gumiere et al. 2011, Hayes and Hairston 1983) done in small-scale settings, such as plots and flume experiments (Gumiere et al. 2011). In most of these situations, contributing areas, inflow rates, sediment levels, study duration, and overall conditions do not mimic field situations; consequently, their reported effectiveness may bear little resemblance to effectiveness measured in actual field conditions (Daniels and Gilliam 1996, Dillaha et al. 1989, Dosskey et al. 2008). A case-in-point comes from Dillaha et al.'s (1989) examination of 24 km of conventional VFS on 18 working farms in Virginia. They found most VFS (they did not specify how many or the length) were not functioning effectively because runoff was collecting in natural drainages due to the hilly terrain, causing the runoff to reach the VFS as concentrated flow rather than sheet flow. In fewer situations with flat terrain, sediment accumulations in older VFS (1 to 3 yr old) became so great that they caused runoff to flow parallel to the VFS until a low point. At those low points, runoff was directed through the VFS as concentrated flow, rendering the VFS less effective.

Variables contributing to or detracting from VFS effectiveness have been identified primarily from observations or small-scale, short-term empirical studies, and these have yielded only a general understanding about the sediment retention processes occurring under VFS installation as a working BMP. Little effort has been put into rigorous and quantitative examinations of physical processes or interactions that would allow prediction of expected runoff responses in different

Table 27.—Studies documenting the effectiveness of vegetated filter strips (VFS) in reducing sediment

Reference/ Type of runoff	Type of VFS[a]	Slope	VFS length or distance through VFS	Sediment reduction Concen-tration	Sediment reduction Mass	Comments
				----- *percent* -----		
Ahearn and Tveten (2008)/Highway	Mixed grasses	~4%	2 m	59–82		As TSS[b]
			4 m	93–96		
Barrett et al. (2004)/ Highway	Nonnative flexible grasses	5–33%	4.2–13 m		77–97	Range represents results from 8 locations; as TSS
Chaubey et al. (1995)/Agriculture	Fescue	3%	3.1–21.4 m	35		Sediment reduction not significantly different at 3-, 6-, 9-, 15-, and 21-m sampling points within VFS; as TSS
Dabney et al. (1995)/Flume experiment	Fescue hedge	5%	280 mm		15–46	Multiple flow rates; Dubbs I soil; flexible grass VFS identified as a hedge; as sediment
	Vetiver hedge		200 mm		34–60	
	Wild switchgrass hedge		200 mm		35–61	
Dillaha et al. (1988)/ Agriculture	Orchardgrass	3 replicates each of 5, 11, and 16%	4.6 m		81	VFS received inflow from simulated feedlot plots; average from 3 simulated rainfall events; as TSS
			9.1 m		91	
Dillaha et al. (1989)/ Agriculture	Orchardgrass	3 replicates each of 5, 11, and 16%	4.6 m		74	VFS received inflow from cropland plots; average from 6 simulated rainfall events; as TSS
			9.1 m		87	
Gharabaghi et al. (2006)/Agriculture	A variety of flexible grasses	~5%	First 2.5 m	50		Sediment reduction in second 2.5 m dependent upon inflow rate; most >40-µm-diameter particles removed in 5 m
			Next 2.5 m	25–45		
Ghate et al. (1997)/ Aquaculture effluent	Bermuda grass and bahaigrass	3%	24 m	18–90		High effluent application rate
				14–82		Low effluent application rate
		1.5%	24 m	45–73		High effluent application rate
				27–84		Low effluent application rate
						As suspended sediment for all 4 tests
Hayes and Hairston (1983)/Agriculture	Kentucky-31 fescue	~2.4%	25.7 m	23–89		Average percentage of sediment trapped during individual storms over 16 months; 2 replicates
				38–87		
Lee et al. (1999)/ Agriculture	Switchgrass	3%	3 m		69	As sediment
			6 m		78	
	Cool season grasses		3 m		62	Cool season grasses were bromegrass, timothy, and fescue; as sediment
			6 m		75	
Lee et al. (2000)/ Agriculture	Switchgrass	5%	7.1 m		70	As sediment
	Switchgrass + woody		7.1 m + 9.2 m		92	VFS order: switchgrass upslope, mixed shrubs and trees downslope; as sediment
Lee et al. (2003)/ Agriculture	Switchgrass	5%	7.1 m		95	As sediment
	Switchgrass + shrubs + trees		7.1 m + 4.6 m + 4.6 m		97	VFS order: switchgrass upslope, shrubs in middle, and trees downslope; as sediment
Line (1991)/ Agriculture	Ryegrass + fescue	5–5.5%	1.5 m		40–80	Contributing area was tilled soil; VFS had ~100% ground cover; both 3 and 6.1 m VFS had approximately the same efficiency; as sediment
			3.0 m		72–95	
			6.1 m		72–95	

(continued)

Table 27.—Studies documenting the effectiveness of vegetated filter strips (VFS) in reducing sediment

Reference/ Type of runoff	Type of VFS[a]	Slope	VFS length or distance through VFS	Sediment reduction		Comments
				Concen-tration	Mass	
				----- *percent* -----		
Magette et al. (1989)/Agriculture	Kentucky-31 fescue	Not stated	4.6 m		66	As TSS mass
			9.2 m		82	
Meyer et al. (1995)/ Flume study	Vetiver	5%	200 mm		34–60	Sediment removal efficiency generally decreased with increasing flow rates
	Tall fescue		280 mm		15–46	---
	Wild switchgrass		200 mm		35–61	
	Fescue + wild switchgrass		350 mm		44–62	Fescue in front of wild switchgrass
	Kanlow switchgrass		760 mm		36–62	
Neibling and Alberts (1979)/ Agriculture	Commercially available bluegrass sod	7%	0.6 m		37	Retention for sediment particle sizes <0.002 mm; all tests with shallow sheet flow
			1.2 m		78	
			2.4 m		82	
			4.9 m		83	
			0.6 m		56	Retention for sediment particle sizes 0.002–0.01 mm; all tests with shallow sheet flow
			1.2 m		70	
			2.4 m		94	
			4.9 m		95	
Paterson et al. (1980)/Agriculture	Fescue	3.4%	35 m	71		Dairy waste used for inflow; as suspended solids
Robinson et al. (1996)/Agriculture	Bromegrass	7%	3 m	70		As sediment concentrations
			9.1 m	85		
		12%	3 m	80		
			9.1 m	85		
Schmitt et al. (1999)/Agriculture	Sorghum	6–7%	7.5 m	63	79	As TSS
			15 m	65	93	
	2-yr-old grass		7.5 m	76	84	Predominantly switchgrass and fescue: as TSS
			15 m	87	96	
	2-yr-old grass + shrubs + trees		7.5 m	79	89	Grass upslope, shrubs in middle, trees downslope; as TSS
			15 m	88	94	
	25-yr-old grass		7.5 m	89	95	Mixed-grass hay field; as TSS
			15 m	93	99	
Van Dijk et al. (1996)/Agriculture	Grass strips	2–9%	1 m	50–60		Species not given; results from 2 sites with differing soil porosity; as sediment concentrations
			4–5 m	60–90		
			10 m	90–99		
Yonge (2000)/ Highway	Native grass mix	Not stated	4.6 m	72		Period 1: 18 months; as sediment concentration
				95		Period 2: 3 months; as sediment concentrations; as sediment concentration

[a] Scientific names of VFS species may be provided in reference.
[b] TSS = total suspended solids.
Empty cells indicate variable was not measured.

types of field conditions with a reasonable degree of confidence (Deletic 2001). Consequently, even though a variety of design recommendations and suitability checklists for VFS installation have been developed (e.g., Dillaha and Hayes 1992), they are not well supported by a breadth of scientific or process-based studies (Dabney et al. 1993, Deletic 2001). Mathematical models, such as the Vegetative Filter Strip Model (Muñoz-Carpena and Parsons 2012), are available to assist with or evaluate VFS design (Dosskey et al. 2008, 2011), but most were developed for agricultural fields.

Applying results from VFS for mostly agricultural land uses to forests has even more unknowns because there have been so few studies of VFS in forests, particularly in association with roads. Roadside vegetation is different from traditional VFS vegetation. Most notably, roadside vegetated strips are not designed to intentionally back up and pond runoff due to the steepness of most forest roadside areas (e.g., fillslopes), the majority of flow received by the roadside vegetation is concentrated flow from road cross-drain features, and the attributes of areas that contribute to VFS from roads are very different from the attributes in agricultural applications. These differences could have a major influence on the specific processes at play in road-applied VFS performance and the consequent effectiveness of roadside vegetated strips.

Barrett et al. (2004) studied the effectiveness of grass strips adjacent to eight freeways in northern and southern California. Using VFS lengths ranging from 4.2 m to 13 m at 31 sites, they measured TSS reductions ranging from 77 to 97 percent. Ahearn and Tveten (2008) observed moderate to high TSS removals, with the effectiveness improving substantially with short distances. At 2 m from the edge of a road pavement, 59 to 89 percent of TSS was removed compared to 93 to 96 percent at 4 m from the edge. Yonge (2000) found that VFS along highways removed an average of 72 percent of TSS. The favorable BMP effectiveness of VFS in many agricultural settings and along highways suggests that VFS should be at least moderately effective in the application to both paved and unpaved forest roads.

As noted earlier, there are few studies associated with forest roads that have examined VFS effectiveness separately from the total vegetated strip/forest buffer combination. Those that exist frequently show the VFS provides a greater contribution to sediment reduction than does the forest buffer. This is the case for sediment transport associated with nonconcentrated as well as concentrated runoff, though effectiveness is greater with nonconcentrated flow (e.g., with outsloped roads rather than cross-drain discharge).

Swift (1986) found that grass on the fillslope of newly constructed roads was capable of reducing sediment transport more than bare soils and more than mulched, ungrassed fillslopes. Maximum and minimum sediment transport distances (based on visual evidence) from roads were twice as long through bare soils and ungrassed fills compared to grassed fillslopes; these included measurements associated with concentrated culvert discharge and nonconcentrated drainage. Even on 60-percent slopes, sediment deposits did not extend more than 45 m on grassed fillslopes.

Hairsine (1997) found VFS to be more effective at retaining sediment than forest buffers of the same length, especially as runoff velocities increased. The combination of VFS and forest buffer within a riparian forest did little to increase sediment trapping compared to the VFS alone. Grass filter strips positioned downslope of the fillslopes and in forest plantations also had greater sediment retention and were more resistant to erosion than the forest floor (i.e., forest litter) on both steep and gentle slopes (29.1-percent and 6.3-percent grade, respectively) (Loch et al. 1999). Some litter scour was visible on the steep slope even when litter was thick, but where litter was thin all litter was scoured from the area. The grass filter strips captured sediment on both types of slopes, but effectiveness was much greater in the strips on the gentle slopes, especially with increasing grass strip length. Because these grass filters were positioned downhill from the fillslope on a gentler grade, they were capable of ponding water at the leading edge, which improved sediment retention. In the North Carolina Piedmont, most of the sediment reduction originating from two agricultural plots occurred as the result of the grass strip portion of a grass/riparian forest buffer because there was not enough roughness to slow flow and settle particles in the forest buffer (Daniels and Gilliam 1996).

Not surprisingly, in the studies by Swift (1986), Daniels and Gilliam (1996) and Hairsine (1997), larger sediment particles were more easily retained than small ones. Hairsine (1997) and Loch et al. (1999) reported that fines tended to move through the entire buffer. Hairsine (1997) did not clarify the diameters of fines, but Loch et al. (1999) defined fines for that study as < 0.125- to 0.05-mm diameter. Daniels and Gilliam (1996) found greater removal of sand-sized particles than silt + clay-sized particles, and the finer particles moved farther through the buffers. Where roadside mowing may take place, the height of the residual grass may be an important consideration during management for influencing both the overall effectiveness and the ability to retain finer particles, though this subject has been only minimally investigated. Overall sediment capture by montane riparian grasses was not significantly different for unclipped grass and grass clipped to 10-cm height; however, much greater sediment transport resulted when the grasses were clipped to the soil surface (Pearce et al. 1998). In contrast, significantly more sediment was captured more quickly by uncut sedges (*Carex* spp.) than by sedges cut to 10 cm. Sedges cut to the ground trapped significantly less sediment than natural or 10-cm-tall vegetation. The differences in all of these comparisons were most pronounced for a fine soil compared to a coarse soil. Overall, larger particles did not move as far as fine particles.

The retention or lack of retention of fines by deposition, filtration, and infiltration in a VFS can have a direct influence on the capture of or transport of metals to receiving waters as metal sorption tends to be dominated by small clay particles (Zanders 2005). Barrett et al. (2004) measured substantial reductions in copper (77 to 97 percent), lead (84 to 99 percent), and zinc (87 to 99 percent) from freeway VFS that were 4.2 to 13 m in length. Metal capture is probably a more important issue for filter strips applied to roads or in urban areas than for agricultural land, where VFS use is more traditional, because metal concentrations in runoff from the former two can greatly exceed those from agricultural land. Even if clays are retained in a VFS it is possible that metals and other clay-bound pollutants eventually may leach into receiving surface water or groundwater. But there has been little research into the fate of these constituents (Barrett et al. 1998b).

There have been no substantive efforts to identify effective flow lengths for forest road VFS. Based on Swift's (1986) findings, the 10-m flow length that generally serves as an effective guideline for agricultural applications may not be sufficient to control erosion associated with roads, where cutslopes and fillslopes usually are steeper. Sediment in runoff that moved through paired upslope VFS and riparian forest buffers could be traced visually (during nonstorm periods) a maximum of 148 ft. This maximum was applicable to situations with and without brush barriers, as well as dispersed runoff and runoff that originated from cross-drain culverts. The lack of visually traceable sediment suggested little or no sediment movement farther downslope. Given the extremely limited data from VFS applications in forests, it is impossible to provide recommendations for VFS lengths appropriate for forest roads.

Forest Buffers and Windrows

Forest buffers are subjected to both diffuse discharge from along the overall road length and to highly concentrated discharge from road drainage features (Swift 1986). Diffuse discharge may be composed of sheet flow or small amounts of concentrated flow, but for the latter the volume of water is usually small enough that concentrated flow paths do not develop or their development is limited in extent and severity. Points of planned discharge from drainage features make up only a small percentage of the road length, but the volumes and velocities of water flowing from those outlets are much greater. Such concentrated flows are the greatest challenge to a buffer's effectiveness in controlling associated erosion and sediment transport (Megahan and Ketcheson 1996, Polyakov et al. 2005, Swift 1986) because infiltration and sediment deposition are difficult to induce with concentrated flow.

The inherent differences in the delivery mechanisms for diffuse and concentrated discharge result in considerable variation in sediment transport distances. Sediment transport by concentrated discharge from road drainage structures typically results in much longer travel distances than by diffuse flow alone (Belt et al. 1992, Burroughs and King 1989), and thus, would require substantially greater buffer lengths for infiltration and sediment retention. Ketcheson and Megahan (1996) measured the length of visibly identifiable sediment deposits originating from roads in Idaho. From 4 yr of monitoring during and after road construction, they found that downslope sediment movement from road cross-drains averaged 49.6 m, which was 3.5 times and 5.7 times the average distance associated with berm drains and rock drains, respectively. The maximum transport distance from cross drains was 275 m, whereas maximum distances for the other sources barely exceeded 100 m. They concluded there was a 15-percent probability that sediment associated with concentrated flow could travel more than 100 m, and for some large cross-drain discharges it could travel as far as 450 m downslope. Eighty-five percent of the individual observations of sediment movement were associated with fillslope erosion and not cross drains, and had an average travel distance of only 3.8 m. A similar result was observed in northern Idaho (Burroughs and King 1989, Croke and Hairsine 2006), where about 90 percent of sediment movement was attributable to fillslopes not influenced by road drains. Sediment associated with fillslopes moved less than 88 ft, whereas sediment transport associated with cross-drain discharge was up to 200 ft.

High levels of erosion and the consequent formation of gullies below drainage features often contribute to the longer sediment transport distances (Mockler and Croke 1999, Takken et al. 2008, Wemple et al. 1996). Gullies or other convergent topographic features that keep flow concentrated substantially increase transport distances (Rivenbark and Jackson 2004), thereby increasing the probability that road-to-stream connections will develop (Croke and Hairsine 2006). Once gullies form, they are difficult to rehabilitate (Barfield et al. 1979), making it challenging to reduce sediment transport distances. On the other hand, divergent topographic features disperse runoff, promote infiltration, and result in short transport distances (Swift 1986). In North Carolina, soil movement never exceeded 20 ft below roads on the hillsides on the outside of curves despite substantial disturbance from road construction on the noses of ridges (Swift 1986).

Because sediment can be transported far downslope from forest roads, recommended forest buffer strip lengths are fairly long (Table 28) compared to recommended VFS lengths (Table 27). Commonly used forest buffer lengths are 30 m, or 60 to 100 m (Barling and Moore 1994, Belt et al. 1992, Clinnick 1985, Davies and Nelson 1994). Different criteria are used to estimate sediment trapping efficiencies for VFS and forest buffers. Sediment trapping efficiencies for VFS are determined from comparisons of incoming sediment and sediment at some point within or at the downslope end of the strip. Studies from which forest buffer lengths are derived usually involve visual observations (Clinnick 1985) of whether sediment is transported through the full length of one or a few buffers (Wenger 1999). There is no comparison between sediment inputs and outputs, so efficiency calculations (percent removal) cannot be made. Consequently, the resulting recommendations from forest buffer studies tend to be by-products of the sampling design rather than experimentally determined (Fennessy and Cronk 1997).

Many of the most basic questions about buffer effectiveness remain unanswered, in part due to the limitations of this type of study design and lack of information about basic site factors that may substantially influence sediment transport distances (Clinnick 1985). These questions include what effective buffer lengths are and how acceptable forest buffer lengths should be established given the widely varying conditions within and among forested watersheds (Correll 1996, Croke and Hairsine 2006, Norris 1993). Despite the lack of this basic knowledge, there is popular acceptance about the effectiveness of forest buffers for protecting water quality (Norris 1993). This acceptance may be attributable to anecdotal observations or monitoring results following forest operations (primarily harvesting or harvesting combined with road construction) with buffers that have shown no or only small turbidity or sediment increases in streamwater during or soon after ground disturbance (e.g., Kochenderfer et al. 1997, Lynch and Corbett 1990). Although these studies typically do not measure or provide visual observations of hillside sediment movement through or retention within the buffer, the lack of evidence in the water column at some point downstream (usually the watershed outlet) is interpreted as evidence of buffer effectiveness. The danger in this indirect approach of determining buffer effectiveness is that sediment may be transported through the buffer at some locations upstream, but storage within the channel

Table 28.—Forest buffer strip lengths recommended for use or stated as being effective within the associated study conditions or review paper

Reference	Geographic location	Hillside gradient	Buffer length	Comments
Aubertin and Patric (1974)	West Virginia	10–65%	10–20 m	Clearcut
Balmer et al. (1976)	Georgia	Level ground and stable soils	9 m	Minimum length required
		Up to 60% and erodible soils	Up to 97 m	
Belt et al. (1992)	Literature review	Not stated	200–300 ft	Lengths that are “generally effective at controlling sediment that is not channelized” (p. 16)
Bren and Turner (1980)	Northeastern Victoria, Australia	Not stated	20 m	Length of undisturbed forest buffer on either side of the channel
Broadmeadow and Nisbet (2004)	Literature review	Not stated	20–30 m	---
Burroughs and King (1989)	Idaho	Various	88 ft	Length needed to capture 90% of sediment flows below road fillslopes
			200 ft	Length needed to capture 90% of sediment flows below road fillslopes where road drains influence runoff
Castelle and Johnson (2000)	Literature review	Various	5–30 m	Lengths were at least 50% and often greater than 75% effective at protecting streams
Castelle et al. (1994)	Literature review	Not stated	15–30 m	Minimum buffer length to protect streams and wetlands in most circumstances
Chalmers (1979)	New South Wales, Australia	Various	30 m	Length is maximum sediment travel distance
Clinnick (1985)	Literature review	Various	30 m	Length is the most commonly recommended, but length should increase with site limitations (e.g., slope, impermeable soil)
Corbett et al. (1978)	Literature review for eastern United States	Not stated	11–22 m	Length needed to prevent water quality deterioration
			20–40 m	Length required to maintain stream ecosystem
Cornish (1975)	Literature review	Not stated	20 m	---
Curry et al. (2002)	Newfoundland	Not stated	20 m	Length “successful in reducing the magnitude of sedimentation following a major storm event (in a stream) subject to clear felling”
Erman et al. (1977)	California	17–22°	≥30 m	---
Graynoth (1979)	New Zealand	17–22°	30 m	---

(continued)

Table 28.—Forest buffer strip lengths recommended for use or stated as being effective within the associated study conditions or review paper

Reference	Geographic location	Hillside gradient	Buffer length	Comments
Haupt (1959a, 1959b)	Idaho	>56%	185 ft	Length needed to trap 83.5% of sediment flows
			230 ft	Length needed to trap 97.5% of sediment flows
Haupt and Kidd (1965)	Idaho	35–55%	3–10 m	Proximity of streams to roads was directly related to frequency of sediment delivery
Haussman and Pruett (1978)	Technical guide		8–50 m	8 m needed for 0% slope; 0.6-m increase in buffer length needed for each additional 1% slope, with a maximum of 50 m for slopes of 70%
Ketcheson and Megahan (1996)	Idaho	15–40°	450 m	Length is maximum sediment travel distance originating at cross drains with large water supply; only a 15% probability that it will exceed 100 m
			60 m	Length needed to capture sediment from all other road sources (e.g., fillslopes, berm drains, and rock drains)
Lynch et al. (1985)	Pennsylvania	Not stated	30 m	Length needed to remove about 75 to 80% of suspended sediment
Packer (1967)	Northern Rocky Mountain Region	Various	35–127 ft	Length depends on types and spacing of obstruction; length needed to capture 83.5% of sediment flows
			95–187 ft	Length depends on types and spacing of obstruction; length needed to capture 97.5% of sediment flows
Plamondon (1982)	Canada	5–30%	10–15 m	---
Rashin et al. (2006)	Washington state	Average near-stream hillslope gradient ranges from 4 to 75%	10 m	Length needed for ground disturbances; can be expected to reduce sediment delivery to streams from harvest-related erosion features
Swift (1986)	North Carolina	47%	65 ft	Lengths represent average sediment travel; maximum travel lengths were 314 ft and 198 ft, respectively
		42% (also burned)	96 ft	
Trimble and Sartz (1957)	New Hampshire	0%	25 ft	Lengths needed to trap 90% of sediment flows
		70%	165 ft	For 100% efficiency, they recommend doubling the distance
van Groenewoud (1977)	Canada	Relatively flat	15 m	---
		Moderately sloping	65 m	
Wong and McCuen (1982)	Maryland	2°	30 m	Length needed for 90% sediment removal
			60 m	Length needed for 95% sediment removal

is sufficient to mask that delivery downslope where the water column is monitored or sampled (Edwards 2003).

Only a small number of published papers have used large datasets or rigorous approaches to define recommended buffer lengths in specific physiographic areas. The most notable are Haupt (1959a, 1959b) and Packer (1967). Haupt (1959a, 1959b) mathematically related sediment movement to site and road drainage factors for 75 sections of haul road in southwestern Idaho; variables included aspect, cross-drain spacing, road gradient, fillslope length, and number and types of obstructions on the hillside. Packer (1967) performed a similar analysis on 720 study sites in the northern Rocky Mountains. He examined sediment travel distances as a function of soil type, road age, cross-drain spacing, distance to first obstruction on the hillside, and fillslope cover density.

Haupt's (1959b) analysis showed that numbers, kinds, and spacings of obstructions on the hillside were the most important variables in determining sediment transport distances. The cross-drain interval was the next most important variable: transport distance increased with the square of cross-drain spacing. Packer (1967) used variables that were statistically significant in explaining sediment movement to develop recommended buffer lengths suitable for 5-yr-old logging roads built in basalt geology with 9-m cross-drain spacings and various obstruction spacings. He also provided adjustments for a variety of conditions, such as different geologic materials, increasing cross-drain spacings, and decreasing fillslope cover. From these analyses, he determined that to retain about 85 percent of sediment flows from cross drains installed on 9-m spacings, the required buffer lengths ranged from 11 to 46 m, with the distance depending upon obstructions, soil characteristics, road width, and vegetative cover associated with the area from which and onto which the structure drained.

It is not surprising that these studies reported the presence of hillside obstructions, or roughness features, as critical to controlling sediment movement in forest buffers. Obstructions, including vegetation, forest floor litter, downed wood, and rocks, along with hillside depressions, provide means to slow or temporarily trap water and allow sediment to settle and be stored (Barling and Moore 1994, Belt et al. 1992, Croke and Hairsine 2006, Ohlander 1976). Obstructions restrict travel distances, as shown in two locations in Idaho where sediment transport lengths were strongly inversely related to the density of hillside obstructions (Burroughs and King 1989, Megahan and Ketcheson 1996). Beasley et al. (1984) found hillside depressions allowed water to accumulate or infiltrate, and dense vegetation and other debris on the soil surface slowed runoff and allowed sediment to settle out during events in which there were small to moderate amounts of water discharged onto the forest floor.

In forests, trees play a much more limited role in controlling sediment delivery through the presence of their stems compared to grasses and herbaceous vegetation along the roadside (i.e., VFS) at the upslope edge of forest buffers (Dorman et al. 1996). This is because overstory stems are not dense enough to sufficiently slow water and promote sedimentation. But tree root growth creates high soil infiltration rates that can substantially contribute to controlling sediment transport through deposition (Lyons et al. 2000). Litter also provides limited roughness because it is not anchored to the soil surface. Even thick litter layers have been shown to be susceptible to scour by concentrated flow (Loch et al. 1999).

The presence of obstructions and depressions varies by site and within sites, making dependence on natural features for energy dissipation, infiltration, and sediment deposition unpredictable and inconsistent. Consequently, filter windrows (or brush barriers) are commonly created during road construction. Windrows are composed of woody slash from right-of-way clearing placed along the contour at or near the base of the fillslope (Burroughs and King 1989). The debris provides a barrier, albeit a porous one, to slow and spread concentrated flow, thereby encouraging water infiltration and sediment deposition (Burroughs and King 1989, Cook and King 1983). Slash presumably is most useful when it is in contact with the ground, so road contracts often include provisions requiring slash to be cut into short lengths to improve contact with the ground and contact with other pieces of debris. However, no studies could be found to compare the effectiveness of simply placing the slash on the hillside to this practice, so the degree of improvement is unknown.

Experiments with filter windrows involve a variety of data collection techniques, so the results are not entirely comparable, but they do tend to show that these barriers are generally effective at reducing sediment delivery. The first year after road construction in Idaho, filter windrows retained all but an average of 0.22 ft^3 of eroded material per 100 ft of road length. Fillslope slumping did result in some larger amounts of soil passing through or over the windrows during spring snowmelt (King 1979). Machine-constructed filter windrows resulted in 75 to 85 percent lower sediment losses from fillslopes than hydromulched fillslopes (Cook and King 1983). Hand-constructed windrows resulted in 88-percent reductions in fillslope sediment losses in the Intermountain Region compared to 99 percent for Curlex® (American Excelsior Co., Arlington, TX) mulch, but the creation of filter windrows is much less expensive than is procuring and applying Curlex mulch (Burroughs and King 1989). Rill formation also was less common on fillslopes that had windrows, and even when rills formed, the average and maximum sediment transport distances (3.8 ft and 33 ft, respectively) were much less than in a variety of permutations of conditions without windrows (average = 25.8 to 80.4 ft and maximum = 85 to 125 ft) (Burroughs and King 1989). In North Carolina, sediment movement on vegetated fillslopes with brush barriers at the toe of fillslopes never exceeded 75 ft (Swift 1985).

Log barriers or log + brush barriers have been used as an alternative to conventional windrows in a variety of situations. Rothwell (1983) experimented with close placement of logging debris barriers on roads. He placed log/brush barriers along the contour 60 and 120 cm apart on road shoulders, ditches, and cutslopes at three stream crossings. These resulted in an average reduction of 75 percent of total suspended sediment production during a summer season. Log erosion barriers installed on two hillsides in California after a wildfire reduced sediment yields by 66 percent compared to a hillside with no log barriers (Wohlgemuth and Robichaud 2007). As with the windrows on fillslopes, site characteristics, including differences in runoff and sediment loads, influenced the effectiveness of log barriers (Wohlgemuth and Robichaud 2007). In some cases the areas behind the logs were completely filled with sediment within only a few years.

These findings by Wohlgemuth and Robichaud (2007) show that vegetative roughness features have a finite capacity for sediment accumulation. They also have a limited life expectancy due to decay and mineralization (Ohlander 1976), but longevity will vary depending upon the size and species of the material. No information was found in the literature concerning how long windrows remain effective in different climates, nor was information found about whether road-to-water body connectivity increases as windrows lose functionality or whether previously deposited sediment becomes remobilized. Other barrier techniques also exist that can be applied in buffers, including silt fence. But because these tools have application other than just within buffers, they are covered elsewhere in this chapter (see the subsection on Vegetated Waterways and Swales, and the section on Non-engineered Barriers).

In addition to roughness, buffer gradient is the other landscape variable that most influences forest buffer effectiveness. Slope is important primarily with respect to sediment in concentrated runoff because velocity and hence energy are directly related to slope. From a review of the effective buffer lengths in the literature, Clinnick (1985) concluded that buffer lengths should increase with increasing slope, and increases in length become most critical on slopes over 30-percent gradient where the topography is concave. In these situations, spreading flows out, and encouraging infiltration on the hillside is very difficult. Trimble and Sartz (1957) recommended increasing forest buffer flow lengths as the hillside slope between roads and streams increased because they observed substantial differences in sediment transport across the range of slopes present in the White Mountains of New Hampshire. Only 8 m of buffer length was needed to retain 90 percent of incoming sediment from roads on fairly level ground, but 51 m of length was necessary to trap that amount on 70-percent slopes.

There is relatively little discussion in the literature to indicate at what gradient a forest buffer becomes too steep to be effective and should be replaced with other types of BMPs for erosion control or sediment retention. That so many other important factors, such as roughness, road discharge rates, sediment concentrations, and soil permeability, vary among sites and through time may make simple generalizations impossible.

Buffers can be fixed or variable lengths (Polyakov et al. 2005). Fixed-length buffers have a pre-determined minimum length that is applied to all or part of the area adjacent to a water body (Lee et al. 2004). Fixed lengths are most commonly used in forest buffer application because they typically are specified in governmental regulations or guidelines, and they are simple to apply. All of the field-based studies on VFS and forest buffer effectiveness reported previously in this chapter were derived from fixed-length buffers.

Variable-length buffers are used less commonly. As their name indicates, their length along a water body varies, depending upon physical characteristics present in the catchment near that location. These characteristics include contributing area, hillside slope, soil characteristics, and pollution sources (Polyakov et al. 2005). Little field-based research has been devoted to developing techniques to define variable-length forest buffers or to validate their effectiveness. Most research in this area has taken the form of heuristic or optimization models (e.g., Dosskey et al. 2002, 2011; Polyakov et al. 2005; Weller et al. 1998).

Based on heuristic modeling, Weller et al. (1998) found limitations to using variable-length buffers because they may capture lower pollutant loads than fixed-length buffers (i.e., when the average buffer lengths of the two types are equal). This is because pollutant transport through the buffer occurs primarily where the buffer length is narrow. Therefore, the average length of a variable-length buffer must be greater than the average length of a fixed-length buffer to attain a specified level of pollutant removal. The Weller modeling used uniform sheet flow from the uphill source areas whereas Dosskey et al. (2002) modeled filter strip pollutant trapping efficiencies using nonuniform runoff and the buffer area ratio (i.e., the ratio of filter strip area to upslope contributing area) from agricultural fields with variable-length buffers. Sediment trapping efficiency was predicted to be 7 to 56 percent less across four sites for nonuniform flow conditions compared to model results for uniform flow, which agreed with the Weller et al. (1998) results.

Establishing variable-length buffer boundaries on the ground can be difficult because buffer lengths that have been defined spatially from models can result in complex patterns that are impractical for field staff to implement (Polyakov et al. 2005). The theoretical advantage of variable buffers is that if properly designed and implemented, they should result in less land within the buffer. This situation can have economic benefits as most buffers have defined limits on disturbance, which can take some land or material out of production.

In highly dissected landscapes with a high density of nonperennial channels, there is a constant tension between protection and production with respect to buffer application. A type of compromise between fixed- and variable-length buffers has been the application of different (usually fixed-length) buffer designations to different types of water bodies (e.g., perennial versus intermittent versus ephemeral streams) (Norris 1993) or to specially designated water bodies. This arrangement recognizes the need to protect water quality and the connectedness of water bodies, while trying to alleviate some of the economic impacts associated with applying buffer protection. However, often the assignment of the specific buffer lengths to a given type of water body is arbitrary (Phillips 1989). At best, research-based lengths are applied to perennial channels, while nonperennial channels, particularly ephemeral channels, typically receive less protection. Nonperennial channels typically have buffers based on lengths that are palatable to users but not defined or supported by scientific data.

The economic impacts of applying buffers to headwater channels are probably the main reason that headwater channel reaches are not consistently buffered, but inconsistent application of buffers to these channels leads to the question posed by Cornish (1975: 10): "Which watercourses require filter strips?" He argued that neither permanence nor frequency of flow should be used to identify water body segments that should receive buffer protection, and instead suggested that any length of stream that has high peakflows and is susceptible to receiving pollutants should be accorded protection from a forest buffer. Norris (1993) went further and stated that ensuring water quality protection requires that buffers extend along all tributaries to the end of the headwaters where flow initiation begins.

The modeling results by Weller et al. (1998) support the need for providing buffers along the entire channel. They showed that eliminating gaps in buffer widths would yield greater protection than providing longer buffers along only parts of a channel. Their work provides further support for avoiding stream crossings and keeping roads far from streams if possible, because stream crossings necessarily create gaps in buffers and provide direct conduits for sediment delivery (along the road). Road impingement in streamside areas (including in the approaches to crossings) necessarily creates short buffer lengths and high potential for sediment delivery in those areas, even if the rest of the buffer has long fixed lengths (i.e., akin to the problems with narrow buffers in variable-length buffers).

Qiu and Prato (1998) noted that riparian buffers in agriculture have a positive economic value, but the prices of agricultural products do not include the value of maintaining water quality, so there is little incentive for buffer installation. Similar challenges exist in forests because of the low profit margins associated with most forest products (Blinn et al. 2000, Timberharvesting.com 2011). To improve the acceptability of applying forest buffers to the entire channel length, nonmarket incentives need to be considered (Polyakov et al. 2005), particularly because headwater forests already have more buffer length and width than downstream lands dominated by multiple-use (Norris 1993). Acceptance of implementing buffers along the full channel length in forested watersheds might increase if the improvement to water quality attained by buffering headwater forests could be shown to influence water quality downstream in lands that do not include buffers. Currently there is a lack of data demonstrating that local forest buffer implementation provides any measurable improvements to water quality at the landscape scale (Norris 1993, Polyakov et al. 2005).

Vegetated Waterways and Swales

Vegetated waterways and vegetated swales are open channels that are lined with low-growing, flood-tolerant vegetation (usually grass) (Dorman et al. 1996). They are oriented in the direction of the slope, and stormwater runoff (both sheet flow and concentrated flow) from upslope contributing areas is collected and transmitted through them (Deletic and Fletcher 2006) with the objective of reducing pollutants in the runoff (Mazer et al. 2001). A properly designed vegetated waterway or swale transmits runoff slowly to allow complete or nearly complete pollutant retention, if not also complete water infiltration (Burkhard et al. 2000, Mazer et al. 2001). Runoff that does not infiltrate fully within the swale or waterway's length is transported with outflow and discharged into additional drainage or treatment systems, or receiving waters (Daniels and Gilliam 1996, Deletic and Fletcher 2006). Some vegetated channels are used only to convey drainage water to another pollution control structure (usually some type of detention pond) and there is no pollution-reduction objective for the waterway itself (Novotny and Olem 1994). However, in this chapter, only the pollution control aspects of vegetated waterways are considered.

Although the terms "vegetated waterways" and "vegetated swales" often are used interchangeably, for some practitioners the appropriate term depends upon the shape of the feature, with the shape being a function of available space (especially width). When a distinction is made, vegetated waterways usually are narrower and V-shaped (e.g., see Resource Conservation District of Monterey County 2005), or trapezoidal (e.g., see Natural Resources Conservation Service 2003). Swales tend to be wider and shallower (Iowa State University 2009). Typical locations for vegetated waterways and swales are in medians of divided highways, along roadways or parking areas, or adjacent to or within commercial or residential developments (Burkhard et al. 2000, Donaldson 2009, Iowa State University 2009, Yu et al. 2001, Zanders 2005). Their application in urban areas also is increasing substantially (Deletic 2005).

Pollution reduction in vegetated waterways and swales occurs by biochemical and physical processes (Deletic and Fletcher 2006) that are influenced by standing vegetation, organic matter, and soil (Mazer et al. 2001). Biochemical processes control retention of dissolved forms of pollutants; these processes involve uptake by vegetation, adsorption onto soil and organic matter (Deletic 2005, Iowa State University 2009, Yu et al. 2001), and transformation into less harmful substances by microbial decomposition (Mazer et al. 2001). Physical processes dominate pollutant removal in vegetated waterways and swales; these include particulate deposition, vegetative filtration, and infiltration of chemicals into the soil with infiltrating water (Deletic 2005, Deletic and Fletcher 2006, Iowa State University 2009, Yu et al. 2001). Particulate deposition, or settling, is the most important means of pollution retention (Bäckström 2002, Claytor and Schueler 1996, Deletic and Fletcher 2006, Dorman et al. 1996). Aboveground plant parts induce sedimentation of particulates by providing roughness and slowing velocity of inflow, while plant roots stabilize sediment deposits and discourage sediment resuspension (Claytor and Schueler 1996, Kadlec and Knight 1996). Filtration of particulates by grass blades or other vegetative parts is much less important for sediment capture (Claytor and Schueler 1996, Deletic and Fletcher 2006, Kadlec and Knight 1996).

The interrelationships between vegetative characteristics and inflow are important to the effectiveness of vegetated waterways and swales. Because vegetated waterways and swales are channels (Barling and Moore 1994, Dorman et al. 1996, U.S. EPA 1999), the depth of water in them changes through time with the size of storm and runoff events (Dorman et al. 1996). Vegetation can be fully submerged (submerged flow) or only partially submerged (nonsubmerged flow) (Tollner et al. 1976). The length of time that plants in a vegetated waterway or swale are subjected to submergence can affect their condition. Greenhouse experiments showed biomass and leaf blade density of grasses were affected by growing them in inundated pots. Biomass and the number of leaf blades were least for pots with 2 to 4 cm of water inundation above the soil surface for two 14-day cycles (Mazer et al. 2001). In contrast, plants grown without inundation had the greatest number of leaf blade and biomass accumulations, regardless of species tested. Biomass averaged across the four inundated turf grass species studied was only about 11 percent of biomass for the same species kept moist but not inundated.

Even establishment of plants well adapted to limited inundation can be reduced by extended inundation (Crawford 1992, 1996; Ernst 1990; Ewing 1996; Kozlowski 1984). In field experiments, swales that were inundated with water for more than 35 percent of the time during summer had significantly less vegetative growth and biomass accumulations in litter (Mazer et al. 2001). Water stagnation also should be avoided as it has similar effects (Burkhard et al. 2000), though standing water in the short term (1 to 14 days) may have slightly less effect on vegetative growth than flowing water over the same duration (Temple 1991).

To be effective, vegetation must have characteristics that allow it to accommodate the range of expected flows without failing, or inflow must be controlled in a way to ensure vegetation can be effective. Substantial reduction in settling potential results during submergence if vegetation becomes prone, breaks (i.e., is shortened) (see Vegetated Filter Strips subsection in this chapter for details on how roughness is affected when vegetation is prone or broken), or is scoured from the feature. Scouring of vegetation from swales is predominantly a problem when flows or inundation persists for long periods (Mazer et al. 2001) or when the feature is too narrow and water becomes too concentrated (Barrett et al. 1998b).

Research by Bäckström (2002) illustrated the importance of vegetative condition in controlling roughness and influencing sedimentation for swales. Using nine grassed swales with a standardized runoff event simulation process, he found short, thin grass had the lowest removal of TSS, at 80 percent. In contrast, swales with well-developed thick turf had the greatest TSS removal (90 to 100 percent). Colwell et al. (2000) examined roadside ditches in Washington state and found that those that were well vegetated and lacked rocks or other coarse roughness features appeared to perform like vegetated swales based upon indirect estimates of vegetative cover, siltation, scour, and energy profiles. Though pollutant removal was not measured directly, their survey data suggested that these ditches resulted in a net removal of pollutants compared to those that had less vegetation or rocky bottoms, or were created in rocky soils. The latter ditches appeared to serve as sources of pollutants, especially sediment.

The removal of particulates by vegetated swales or waterways ultimately depends upon achieving some balance between flow rate and particle settling velocity (Deletic and Fletcher 2006). Although vegetation abundance contributes to controlling velocity and particle retention, it is not the only controlling feature, because even nonsubmerged flows can have high flow rates that are not conducive to particle settling (Mazer et al. 2001). Altering the longitudinal slope, length, and overall shape of vegetated waterways and swales can change the inflow rate and depth of water; these in turn influence the degree of infiltration and deposition (Deletic and Fletcher 2006). Reducing the longitudinal slope and increasing the length of the vegetated waterway or swale both help to increase retention times and potential sediment deposition (Deletic and Fletcher 2006, Yu et al. 2001).

Mazer et al. (2001) observed that vegetated swale performance was most effective when inflow was controlled in such a way that it was always shallow (<37 mm) and hydrologic retention time within the feature was long (>9 min). They did not recommend specific slopes because they found that a slope of even 1.5 percent was too much for some swales to handle the high flow rates that developed for even small storms. Yu et al. (2001) tested swale performance in Virginia, comparing a swale with a 1-percent grade to one with a 3-percent grade. Based on sediment reductions that they measured and results of eight other studies from the literature, they concluded that a 3-percent longitudinal slope performed better and was optimal for swale construction. Similarly, Colwell et al. (2000) examined roadside ditches in Washington state; those that behaved like vegetated swales had slopes of 2 to 3 percent in combination with gentle side slopes of no more than 3:1 (horizontal:vertical). Those that were flatter longitudinally would retain too much water, whereas those that were steeper had high energy and vegetative and soil scouring. These studies suggest that the maximum longitudinal slope for swales and waterways is probably no more than 3 percent. However, slopes of swales generally range from 0.5 to 6.0 percent (Mazer et al. 2001), so many existing swales may be challenged by their associated inflow velocities.

The actual effect that any change in vegetated swale or waterway shape has on sediment capture depends in some part on the particle size distribution of incoming sediment. Most pollutant reduction by vegetation follows exponential decay (Deletic and Fletcher 2006); that is, most occurs in the first few meters of length (Bäckström 2002, Kaighn and Yu 1996) and then retention falls off exponentially. This is because the largest/heaviest particles settle out most easily throughout the entire range of flow velocities, and these particles compose most of what is captured (Deletic and Fletcher 2006, Yu et al. 2001). For this reason Bäckström (2002) found no relationship between pollutant removal and swale length; the heaviest particles settled out in the first few meters of the feature, so extending the flow length contributed little to sedimentation.

Increasing swale length is important for retaining fine particles (e.g., clays) and dissolved constituents (Bäckström 2002, Wang et al. 1981), but only as long as infiltration rates remain high (Bäckström 2002). This is because fines and dissolved constituents are not captured easily through sedimentation, but instead depend upon having sufficient flow-through or residence time in the swale or waterway to allow infiltration to occur (Kaighn and Yu 1996, Yousef et al. 1985). Bäckström (2002) found the advantage of increasing swale length was associated primarily with particles less than 25-µm diameter. In Washington state, about 90 percent of lead levels were removed by a 60-m-long swale compared to 60-percent removal by a 20-m-long swale (Wang et al. 1981). However, even when infiltration capacity is high, infiltration may become inadequate to allow for retention of small particles (<50 µm diameter) when velocities become too great or flow too deep (Mazer et al. 2001). For this reason, swales work best when they are exposed to frequent light rainfalls—even those of extended duration—rather than large intense events (Yu et al. 2001).

Some studies have shown that the presence of a strip of vegetation located between the edge of the pavement and the beginning of the swale or waterway is an important physical attribute in their design (Barrett et al. 1998b, Wu et al. 1998). This strip acts as a vegetated filter strip (VFS; described previously in this chapter), which "pretreats" runoff (Stagge et al. 2012) before it enters the swale or waterway. However, other studies have shown the swale or waterway to be more important than the VFS (Bäckström 2003, Schueler 1994).

Low concentrations of incoming pollutants to vegetated waterways or swales, whether due to initial removal by VFS or by naturally low inflow pollutant levels, generally result in poor performance (Welborn and Veenhuis 1987). When influent TSS concentrations were less than 30 mg L^{-1}, no sediment reduction occurred in Maryland swales (Dorman et al. 1996). Similarly, Bäckström (2003) found no significant reduction in TSS when inflow concentrations were less than 40 mg L^{-1}. In contrast, when TSS concentrations have been in ranges normally associated with highway runoff (i.e., where swales and vegetated waterways normally are installed), substantially improved TSS removal usually results. For example, 93 and 94 percent of TSS was removed during two storms when influent concentrations were between 100 and 200 mg L^{-1} in Florida (Dorman et al. 1996). Bäckström (2003) summarized three studies and showed that 79 to 89 percent of TSS was removed during simulated storm events and the parameter that best explained TSS removal was TSS concentrations in inflow.

Because so many factors influence pollutant retention in swales and vegetated waterways, it is not surprising that a wide range of pollutant removal efficiencies is reported in the literature (Table 29). Total suspended solids (i.e., primarily sediment) tend to be removed more efficiently than other constituents (Horner et al. 1994, Municipality of Metropolitan Seattle 1992), but even suspended solids removals have ranged from being ineffective to completely effective (Weiss et al. 2010). Metals (Table 29) tend to be removed more effectively than dissolved nutrients because metals commonly adsorb to sediment (Dorman et al. 1996, Schueler et al. 1992).

The inconsistency in retention efficiencies among studies has resulted in a general lack of confidence in the effectiveness of vegetated waterways and swales (Bäckström 2002, Barrett et al. 1998b). Many stormwater handbooks recommend these techniques primarily as pretreatment techniques (Barrett et al. 1998b) before releasing water into other filtration or stormwater controls.

Structural barriers, known as check dams, can be installed in vegetated waterways and swales (Fig. 11) to increase detention times, thereby helping to compensate for poor infiltration or poor vegetative cover (Mazer et al. 2001) by temporarily ponding and storing water (Yu et al. 2001). Ponding water slows its velocity, and storing water increases the time of detention; both of these factors increase the opportunity for deposition and infiltration (Kaighn and Yu 1996, Yu et al. 2001). During low runoff events, check dams in swales have been found to double the detention times of water compared to where there was no check dam in place (Yu et al. 2001). Check dams can be constructed from a variety of materials, including stone berms, railroad ties, riprap, gabions (large wire cages usually containing rocks or broken concrete), pressure-treated wood, and natural wood that is resistant to decay (U.S. EPA 2004). Specific details about pollutant retention processes associated with barriers are provided in the previous section of this chapter (Non-engineered Barriers). The effectiveness of using barriers aside from those in vegetated swales and waterways also is covered in the previous section.

A few studies have demonstrated that swales and waterways containing check dams tend to be more effective than those without check dams. Not surprisingly, the greatest improvement in pollutant retention by check dams typically results for fine particles and dissolved chemicals (Kaighn and Yu 1996)

Figure 11.—A series of rock check dams in a grassed swale to create storage potential within the swale. Photo by Virginia Department of Conservation and Recreation (1999), used with permission.

Table 29.—Sediment, metal, and other chemical removal efficiencies reported for vegetated waterways and swales

Reference	Type of feature	Sediment removed (as TSS[a])	Metals and other constituents removed		Comments
		percent	*percent*		
Bäckström (2003)	Grassed swale	79–98			Simulated runoff events
Barrett (2008)	Grassed swales	Mean = 60 Range 6–70			Data extracted from 14 swales in international BMP database
Goldberg et al. (1993)	Grassed swale	67.8			8 storm events
Kercher et al. (1983)	Grassed swale	97.9			Residential area
Mazer et al. (2001)	Grassed swales	60–99			Non-roadside swales
Municipality of Metropolitan Seattle (1992)	Grassed swale	83	72	Total iron	All values from 200-ft-long swale; THP = total petroleum hydrocarbons
			67	Total lead	
			46	Total copper	
			63	Total zinc + total aluminum	
			30	Dissolved zinc	
			75	Oil + grease/THP	
Oakland (1983)	Grassed waterway	33			11 storm events
Walsh et al. (1998)	Grassed swale	35–59			At 10 m through swale
		54–77			At 20 m through swale
		50–76			At 30 m through swale
		51–75			At 40 m through swale
Wang et al. (1981)	Grassed swale	90.4			At 21 m length
		93.2			At 43 m length
		94.5			At 67 m length
Yousef et al. (1985)	Grassed swale		90	Total zinc	Reductions of metal concentrations in highway runoff through swales over 8 months
			82	Dissolved zinc	
			91	Total lead	
			50	Dissolved lead	
			41	Total copper	
			19	Dissolved copper	
			71	Total iron	
			44	Dissolved iron	
			44	Total chromium	
			13	Dissolved chromium	
			29	Total cadmium	
			18	Dissolved cadmium	
			86	Total nickel	
			47	Dissolved nickel	

[a] TSS = total suspended solids.
Empty cells indicate variable was not measured.

as fines and dissolved species have low settling velocities (which require lengthy detention times) or depend upon infiltration, or both (see Non-engineered Barriers section). Kaighn and Yu (1996) compared pollutant retention from highway runoff by a swale without a check dam to one with a weir-type check dam during eight rain events. Removal of TSS ranged from 73 to 100 percent with a single check dam compared to a range from -4.1 percent (i.e., there was a net increase in TSS through the swale) to 57.4 percent for a swale of the same slope and length with no check dam (Table 30). Likewise, reductions in zinc, chemical oxygen demand, and total phosphorus were much greater with the check dam (Table 30). The 100-percent pollutant reductions observed in this study correspond to storms in which 100 percent of inflow was infiltrated into the soil, which occurred due to the check dams. Yu et al. (2001) examined the effects of length, longitudinal slope, and the presence of check dams in swales and also found the most important feature for improving retention of TSS and total phosphorus was a check dam.

Shipitalo et al. (2010) examined the efficacy of check dams constructed from compost filter socks (see Non-engineered Barriers section in this chapter for more description of compost filter socks) placed in grassed waterways that received drainage from tilled or no-till corn (*Zea mays*) fields. Sediment concentrations from the no-till fields were not reduced significantly by filter socks, presumably because sediment concentrations in inflow were very low. However, from tilled fields, which had much higher inflow sediment concentrations, the filter socks decreased the sediment concentrations by an average of 49 percent. The filter socks worked well even with high runoff volumes in the waterway, where many other types of BMPs may have been challenged. The composition of compost media (e.g., particle-size mixture) may be designed specifically to handle large inflow volumes, but these designs usually require higher flow-through rates to avoid overtopping, which translates to less-effective control of suspended sediment and turbidity (Faucette et al. 2006).

Table 30.—Percent reductions in various constituents without and with a weir-type check dam in a swale during eight storm events in Virginia (data from Kaighn and Yu 1996)

Constituent[a]	Range removed: Without check dam	Range removed: With check dam	Average removed: Without check dam	Average removed: With check dam
	------------ percent			------------
TSS	-4.1–57.4	73–100	23.3	87
Total zinc	5.1–35.4	58–100	17.8	83.8
COD	-27.8–54.9	67–100	29.8	84
TP	-14.9–55.6	80–100	11	91.5

[a] TSS = total suspended solids; COD = chemical oxygen demand; TP = total phosphorus.

Straw or hay bales are sometimes used as check dams in swales and waterways, but there is no information in the literature on their application to these types of biofilters. However, the U.S. EPA (2002) recommends against their use as check dams with flowing water because they are easily undercut or overtopped due to their impermeability and proclivity to dislodge and collapse when flow concentrates in channels (Indiana Department of Natural Resources 1992, Robichaud 2010, U.S. EPA 2002). In addition, hay and straw bales decay easily, so even if they are installed properly and are effective in the short term (<3 months) (Harbor 1999, U.S. Department of Transportation 1995), failure in the long term is likely despite intensive maintenance. Additional information about straw bales as check dams in ditches or as general erosion control barriers to overland flow is provided in the section on Non-engineered Barriers.

Additives show promise for increasing the effectiveness of vegetated waterways and swales. A wide variety of chemical and organic material additives have been tested, primarily in laboratory studies, for their potential use for increasing pollutant retention in vegetated waterways or swales. These include activated charcoal, compost, metal compounds, clay compounds, and other chemical additives. However, because many of these materials have undergone little if any field testing—and they probably would not be used much in swales or waterways associated with forest roads—they are not described here. Interested readers are directed to Weiss et al. (2010), who have synthesized this literature.

There has been little research on the moderate- to long-term fate of particulates or dissolved chemicals that have been removed by vegetated waterways and swales (Davis et al. 2003, Yu et al. 1993). Intense storms create the potential for resuspension of settled particles in the short term (Yu et al. 2001), but little information is available about whether particles become part of the general soil substrate through time, as they can with VFS (Barrett et al. 1998b, Dillaha et al. 1989). Chemicals that were retained through plant uptake or adsorption onto plant or organic surfaces have the potential for remobilization during mineralization (Bäckström 2003). Consequently, clippings from mowed waterways and swales should be removed to reduce the potential for transport of metals and other pollutants released by decomposition (Colwell et al. 2000, Dorman et al. 1996, Schultz 1998).

Straw bales installed in a ditch line to slow drainage and capture sediment, and straw mulch and silt fence on a cutbank used to control erosion after construction of an access road to a natural gas well pad. (Photo by U.S. Forest Service, Northern Research Station.)

CHAPTER 8

Research Needs, Potential Direction, and Concluding Thoughts

Research Gaps and Potential Future Direction

There is a common perception that the effectiveness of forestry BMPs, including forest road BMPs, is well supported by scientific research (e.g., see Hornbeck and Kochenderfer 2001 or Ice, n.d.). Many road BMP effectiveness studies do exist; however, the effectiveness of most forest road BMPs has not been investigated rigorously (including replicated and quantitative studies) under a wide variety of geologic, topographic, physiographic, and climatic conditions since their development decades ago. Much more quantification of effectiveness is needed (Anderson and Lockaby 2011a, Moore and Wondzell 2005, Stafford et al. 1996) to understand the site characteristics for which each BMP is most suitable and for proper selection of the most effective BMP techniques (Carroll et al. 1992, Weggel and Rustom 1992).

The divergence between the general belief of well-supported science and lack of critical analysis may result from several factors:

1. Many of the most commonly cited studies that provide the basis for illustrating forestry BMP effectiveness are paired watershed tests in which a treated watershed (e.g., harvesting with road construction) is compared to a control or reference watershed in which no land-disturbing activities were undertaken (Hewlett 1971). Some of these studies were designed and analyzed following classic paired watershed regression techniques (e.g., Arthur et al. 1998, Brown and Krygier 1971, Lynch and Corbett 1990, Rice et al. 1979); others simply compared water quality results between watersheds (e.g., Reinhart et al. 1963). These studies interpreted small changes in water quality (usually sediment loads or turbidity) at the mouth of a treated watershed compared to a control as "proof of effectiveness."

   Although paired watershed studies are common, they are limited in their ability to demonstrate and quantify BMP effectiveness for water quality protection. Most paired watershed studies have less than 5 yr of pretreatment measurements, which limits the amount of variation that can be captured for the comparisons. Therefore, it is difficult to accurately interpret post-treatment data that fall outside pretreatment ranges (Lewis et al. 2001). Sediment measurements at the mouth of a watershed do not account for in-channel storage of delivered sediment and the associated lag times in sediment delivery to the mouth of the watershed (Edwards 2003, Rice et al. 1979). Finally, in paired watershed studies BMPs are considered en masse, making it impossible to quantify the effectiveness of individual BMPs or to identify the individual BMPs that were most or least effective.

2. Studies have investigated the effectiveness of individual road BMPs, but the lack of replication and broad-scale testing across different physiographies, climates, soil types, and other factors for most BMPs weakens the argument that their effectiveness is scientifically well proven. As a result, a single study or just a few studies typically are cited in the literature to support a BMP's effectiveness or the overall general effectiveness of BMPs.

3. The similarity of forest road BMPs used in many different states' forestry BMP manuals and handbooks suggests a degree of confidence in their

validation that may not be justified. For example, many eastern state BMPs employ recommendations for waterbar spacings on skid roads developed from a single study (Elliot et al. 2014) performed in West Virginia (Trimble and Weitzman 1953, Weitzman 1952).

However, the lack of replicated testing of individual BMPs under different conditions should not dismiss them from consideration as tools for protecting water quality. Because most BMPs are based on the laws of physics and chemistry, they typically contribute some degree of nonpoint source pollution control if installed and maintained properly (Anderson and Lockaby 2011b, Stuart and Edwards 2006). For example, it is well understood that controlling the mass or velocity of water reduces its energy available to do work, so it is logical that incorporating that principle into BMPs will reduce erosion. But the dearth of information about most individual BMPs leaves many important questions unanswered, such as: How effective (i.e., quantified) is the BMP in different situations or conditions? and Does its level of effectiveness warrant its implementation or would another technique be more effective and possibly more cost-effective? These questions are particularly applicable to skid road BMPs, as skid roads are included in only a small minority of road BMP research, and those that do involve skid roads focus on BMPs applied once they are "put to bed" (i.e., removed from service). There is scant information on the effectiveness of BMPs applied during skid road use when the potential for water quality impact is high.

As indicated in the previous chapters of this document, most BMP effectiveness studies have focused on the initiation of erosion and quantifying erosion rates. Historically, this information was critical for promoting the acceptance of BMPs and their implementation. But in the future it will no longer be sufficient for BMP effectiveness studies to measure only erosion rates, as these do not provide the necessary information to quantify sediment delivery to water bodies (Anderson and Lockaby 2011b, Dickinson and Wall 1977, Grace 2005, Walling 1983). In conjunction with sediment delivery measurements, there is a pressing need to identify the locations or sources where sediment originates, understand why/how sediment delivery is controlled and explain the mechanisms by which BMPs protect water quality (Anderson and Lockaby 2011b, Dickinson and Wall 1977, Sutherland 1998c). These questions generally have been ignored in forest road research (Anderson and Lockaby 2011b, Rivenbark and Jackson 2004). Most BMP studies report only outcomes, such as erosion rates or captured volumes of sediment, and they were not designed to provide information about the processes or environmental variables that contributed to or resulted in those outcomes. Being able to predict sources of sediment and understand the mechanisms and processes that control or contribute to sediment losses and mobility are critical for improving BMP effectiveness (Anderson and Lockaby 2011b, Rivenbark and Jackson 2004).

Understanding the sources, mechanisms, and processes also will improve models used to predict sediment delivery (Anderson and Lockaby 2011b). Commonly used contemporary road erosion models (e.g., the Water Erosion Prediction Project; U.S. Forest Service, n.d.) typically are driven by physical parameters, including road geometry or contributing area. These models exclude important hillslope hydrology mechanisms and processes, including subsurface flow interception by the road prism and infiltration of water discharged off the road. They also exclude the conditions that influence those processes, which may be the most critical factors controlling sediment delivery (Thompson et al. 2010). Improving prediction models by incorporating source and mechanism/process information should help validate BMP effectiveness and illustrate BMP robustness across many different conditions when replicated rigorous field testing has not been performed in a specific region or condition.

BMP studies tend to extend over only one or a few years, which is much shorter than the life of most forest roads. Due to the potential lag in sediment delivery that can occur, long-term studies of road BMP effectiveness are warranted (Anderson and Lockaby 2011b, Daigle 2010). The conditions to which a road is subjected over the long term also are likely to include more interannual extremes than would occur during an individual study.

Many of the initial studies of the effectiveness of individual road BMPs (e.g., surfacing material, cross-drain spacing) were conducted at U.S. Forest Service experimental forests or similar institutional research locations. Thus, for some studies it may be possible to return to the location of the initial study and resume data collection after reinstrumentation of the site. Current (i.e., long-term) results can be compared to data collected when the road was newer. If current circumstances do not allow studies to be resumed or would confound interpretation, or if it is impossible to pinpoint past research locations, space-for-time (SFT) studies of long-term effectiveness may produce useful findings. For example, studies of the effectiveness of road BMPs that were implemented several years or decades ago provide the opportunity to evaluate responses to years of wetting and drying cycles, extreme storm and runoff events, and long-term use and maintenance (e.g., Bold et al. 2007).

SFT studies also allow comparison of effectiveness for BMPs that have undergone refinement through time (e.g., cross-drain spacings may have changed with revisions to state BMPs over time) (e.g., Henderson 2001). The effectiveness from road segments containing a range of the prescription for that BMP can be compared. Of course, SFT studies must account for differences in road and landscape conditions (e.g., soil erodibility, road design) when comparisons are made across sites. Whenever possible, long-term studies should be coupled with validated models or used to improve models so the spatial and temporal variability of sediment delivery can be better understood and predicted (Anderson and Lockaby 2011b).

Understanding the mechanisms and processes that affect whether sediment delivery to water bodies will occur and the differences between short- and long-term effectiveness will move the science toward the ability to develop the most effective site-specific BMP prescriptions, a need identified for many years (Aust and Blinn 2004). Site-specific BMP prescriptions involve selecting the most effective BMPs for the site conditions and locating them where they provide the greatest benefit. This approach requires acknowledgment that some BMPs do not perform as well as others in a given situation (Sutherland 1998a).

Site-specific BMPs would result in much different BMP implementation strategies than current approaches, in which blanket applications of state recommendations are employed. Blanket application typically allows the user to identify the BMPs that will be applied. Without the advantage of scientifically based studies to guide the decision, however, the individual BMPs may not be the most effective. Furthermore, blanket application typically allows little flexibility in where BMPs are applied. Blanket application of BMPs within a watershed or managed area presupposes that all areas have more or less equal potential for being the source of delivered sediment. For forest roads, where sediment tends to originate from discrete segments or features on only a small percentage of the corridor length (Cafferata et al. 2007, Mills 2006, Rice and Lewis 1991, Skaugset et al. 2007), this assumption is false. Consequently, although blanket application is probably the easiest and most efficient way to ensure BMP implementation, it can be economically inefficient and suboptimally effective (Thompson et al. 2010). Blanket BMP implementation also can create disincentive to apply extra dollars where they are most needed to protect water quality; available funds, which typically are limited, are expended across the entire project area/road length, including where they provide little benefit to water protection. The lack of flexibility in BMP application may cause some to question the environmental benefits they provide (Brynn and Clausen 1991), thereby reducing the likelihood of their implementation.

As suggested above, BMP flexibility inherently includes an economic component— that is, being able to define and apply funds where they are most needed to address current or potential impacts. As such, application of BMP treatments is increasingly being viewed as inefficient if it does not include an analysis of the overall cost-effectiveness of the outcome (Thompson et al. 2010). This approach requires an analysis of environmental performance measures (e.g., what, if any, BMPs can provide the desired environmental outcome, including consideration of their location) for forest roads (Mills 2006) and a technique to simultaneously achieve economic and environmental goals. The latter includes determining the point at which increasing benefits end.

Optimization modeling is one approach to meeting these outcomes (Veith et al. 2003), but field measurements of the effectiveness of specific BMPs are necessary to validate optimization models. Optimization modeling requires that incommensurate objectives (e.g., cost of road construction and mass of sediment delivered, or cost of BMPs and cost of environmental impacts) be developed and compared to determine the optimal outcomes (Rackley and Chung 2008, Thompson et al. 2010). However, research into the valuation of many environmental factors, such as maintenance of water quality or soil loss, is in its infancy.

Different people value environmental consequences differently, so there is no unique answer within optimization modeling (Thompson et al. 2010). This factor impedes the broad use of optimization, particularly as an alternative to current blanket applications of BMPs. If optimization modeling becomes widely used for helping to more efficiently prescribe BMPs, states (to whom the U.S. EPA has delegated BMP authority) could set sediment reduction objectives (e.g., on a watershed scale) to achieve environmental outcomes and water quality protection.

All the previous issues and needs presented in this chapter will be complicated in the future by potentially significant changes to climate. Although the specific changes and their magnitude will vary by region, more-extreme weather conditions are expected to increase the frequency of flooding, soil surface and in-channel erosion, and annual variability in streamflow (e.g., greater peakflows and lower baseflows) (Furniss et al. 2010, Marshall and Randhir 2008). Maintaining forest road integrity and protecting water quality may become more difficult during more-extreme events (i.e., changing precipitation timing, intensity, and volume) as these may create conditions that challenge BMP performance. Even under the conditions in which BMPs were developed, intense events have been observed to severely test road BMPs, sometimes resulting in failure or suboptimal performance. More-intense events, more frequent events, and longer duration events that accompany climate change may demonstrate that BMPs perform even more poorly in those situations. Research is urgently needed to identify BMP weaknesses under extreme events so that refinements, modifications, and development of BMPs do not lag behind the need.

The focus of road BMPs is controlling erosion and sediment transport on the road prism and on hillsides where road runoff is influential; in other words, the ultimate objective of BMPs is to control nonpoint source pollutant delivery to water bodies (Blinn and Kilgore 2001). Under climate change, hydrologic changes are predicted to increase on-the-ground erosion/sediment transport as well as to increase in-stream erosion and alter channel characteristics (Furniss et al. 2010). Although some road BMPs do include aspects of hydrology (e.g., turning water off roads in small parcels), they are not designed to control in-stream erosion or channel modifications (with the exceptions of controlling substrate changes from sedimentation and channel morphology alterations from road crossings). However, because roads have such a significant effect on watershed hydrology, under climate change the onus of controlling in-stream hydrologic changes attributable to roads (e.g., increased discharge attributable to cross drains connected to the stream) may fall to BMPs. Tempering road-induced hydrologic changes may become necessary for maintaining healthy and resilient aquatic habitats as climate-induced changes in streamflow regimes and resultant changes to habitat conditions are expected to stress some aquatic systems (Marshall and Randhir 2008). If controlling road-induced hydrologic changes becomes a BMP objective, substantive changes to existing BMP prescriptions and development of entirely new BMP techniques may be necessary.

Concluding Thoughts

At the opening of this chapter, we indicated that the effectiveness of most BMPs has not been well quantified from the perspective of statistical rigor and replicated studies across many different types of conditions. But the sheer number of papers included in this document shows there are many studies of road BMP effectiveness as well as studies with results that may be applicable to roads. Based on the results of most of these studies, the case can be made that most BMPs result in some level of effectiveness in terms of reduced sediment generation or transport. Until more extensive and rigorous comparisons of effectiveness become available for specific types or categories of BMPs, the information and tabulated data herein provide the reader a starting place for selecting BMPs for local use.

During the compilation of this document we decided to include some techniques that are rarely used for forest roads, but that we believe have application to them. For some BMPs this was somewhat of a controversial position, even during the manuscript review period. In the end, we retained all the sections we originally included so as to provide a breadth of BMP possibilities for individuals wanting to extend their options beyond those that they have traditionally used. Considering and perhaps trying new techniques when those typically employed do not perform to expectations is key to adaptive management. Especially as new stressors are placed on the ecosystem, adaptive management is essential for continued watershed protection and improvement.

Wide ranges of BMP effectiveness were reported in many studies in which different techniques were tested under identical conditions. Comparisons of specific types of BMPs across studies also show variability. This variability is evidence that use of the phrases "BMPs minimize erosion" or "BMPs minimize nonpoint source pollution," which are often found in BMP literature, should be discontinued. If one BMP performs more poorly than another in the same or similar situations, both cannot be minimizing pollution. Furthermore, if the intention is to minimize pollution, the actions taken are likely to be much more intensive and involve much greater costs than simply implementing BMPs. In many cases the only action that would truly minimize pollution would arguably be the decision not to implement any management or cause any disturbance. The Society of American Foresters' dictionary of forestry (Society of American Foresters 2008) defines best management practices as "a practice or usually a combination of practices that are determined by a state or a designated planning agency to be the most effective and practicable means (including technological, economical, and institutional considerations) of controlling point and nonpoint source pollutants at levels compatible with environmental quality goals." The ideas of "practicable means" and "compatible with environmental goals" describe the essence of BMPs: Neither the most complex/costly techniques nor the total elimination of pollutants is required or expected with BMP use. Using statements such as "BMPs minimize pollution" can create a false impression about the degree of pollutant generation and transport to be expected with BMP implementation. This false impression in turn provides fodder for individuals and groups who argue for a shift to regulatory means of pollution control for roads. It is best to portray BMPs for what they are because they play such an important role in protecting watersheds and water quality.

ACKNOWLEDGMENTS

The authors thank the U.S. Forest Service's San Dimas Technology and Development Center, and in particular Kim Clarkin, Mohammad Nash, and Ed Messerlie, and the U.S. Environmental Protection Agency for their input and support for publishing this synthesis. Special thanks go to the National Agricultural Library staff who expeditiously fulfilled our hundreds of requests for publications and reports that we could not have otherwise obtained. We also thank Kim Clarkin, Mohammad Nash, Peter Adams, Johnny Boggs, Mike Dosskey, Bill Hansen, and Julianne Thompson of the U.S. Forest Service, Jon Schoonover of Southern Illinois University, and two anonymous reviewers for contributing comments to improve previous drafts of this manuscript. Finally, we extend our sincere gratitude to Cynthia Moser, who expeditiously and expertly edited this document.

REFERENCES

Aase, J.K.; Bjorneberg, D.L.; Sojka, R.E. 1998. **Sprinkler irrigation runoff and erosion control with polyacrylamide – laboratory tests.** Soil Science Society of America Journal. 62(6): 1681-1687.

Abdelmagid, H.M.; Tabatabai, M.A. 1982. **Decomposition of acrylamide in soils.** Journal of Environmental Quality. 11(4): 701-704.

Abu-Hamdeh, N.H.; Abu-Ashour, J.S.; Al-Jalil, H.F.; Khdair, A.I.; Reeder, R.C. 2000. **Soil physical properties and infiltration rate as affected by tire dynamic load and inflation pressure.** Transactions of the American Society of Agricultural Engineers. 43(4): 785-792.

Abu-Zreig, M.; Rudra, R.P.; Lalonde, M.N.; Whiteley, H.R.; Kaushik, N.K. 2004. **Experimental investigation of runoff reduction and sediment removal by vegetated filter strips.** Hydrological Processes. 18: 2029-2037.

Abu-Zreig, M.; Rudra, R.P.; Whiteley, H.R.; Lalonde, M.N.; Kaushik, N.K. 2003. **Phosphorus removal in vegetated filter strips.** Journal of Environmental Quality. 32: 613-619.

Adams, J.W. 1988. **Environmental effects of applying lignosulfonate to roads.** [Internal report]. Rothschild, WI: Daishowa Chemicals Inc. 11 p.

Addo, J.Q.; Sanders, T.G. 1995. **Effectiveness and environmental impact of road dust suppressants.** MPC Report No. 95-28A. Fargo, ND: Mountain-Plains Consortium. 132 p.

Agassi, M.; Ben-Hur, M. 1991. **Effect of slope length, aspect, and phosphogypsum on runoff and erosion from steep slopes.** Australian Journal of Soil Research. 29: 197-207.

Agassi, M.; Ben-Hur, M. 1992. **Stabilizing steep slopes with soil conditioners and plants.** Soil Technology. 5: 249-256.

Ahearn, D.; Tveten, R. 2008. **Legacy LID: stormwater treatment in unimproved embankments along highway shoulders in western Washington.** In: Proceedings, ASCE 2008 international low impact development conference. Reston, VA: American Society of Civil Engineers. 10 p.

Ajwa, H.A.; Trout, T.J. 2006. **Polyacrylamide and water quality effects on infiltration in sandy loam soils.** Soil Science Society of America Journal. 70: 643-650.

Alaska Department of Environmental Conservation [DEC]. 2010. **Air non-point and mobile sources.** Anchorage, AK: State of Alaska Department of Environmental Conservation, Division of Air Quality. 3 p. http://www.dec.state.ak.us/air/anpms/pm/pm10_ctrl.htm (accessed February 23, 2011).

Alexander, R. 2003. **Standard specifications for compost for erosion/sediment control.** Prepared for: University of New Hampshire, Recycled Materials Resource Center. Apex, NC: R. Alexander Associates, Inc. 9 p. Available at http://www.alexassoc.net (accessed October 25, 2012).

Alexander, R. 2006. **Filter berms and filter socks: standard specifications for compost for erosion/sediment control.** Apex, NC: R. Alexander Associates, Inc. 8 p.

Amaranthus, M.P.; Rice, R.M.; Barr, N.R.; Ziemer, R.R. 1985. **Logging and forest roads related to increased debris slides in southwestern Oregon.** Journal of Forestry. 83(4): 229-233.

American Association of State Highway and Transportation Officials [AASHTO]. 2012. **Standard practice for NTPEP evaluation of erosion control products [draft].** Washington, DC. 7 p. http://www.ntpep.org/Documents/Technical_Committee/ECP/Documents/ECP%20-%203-6-12%20FINAL.pdf (accessed September 5, 2012).

American Society of Civil Engineers. 1992. **Local low volume roads and streets.** New York: American Society of Civil Engineers: Chapter II: 17-22.

American Society for Testing and Materials [ASTM]. 2004. **ASTM standard D4751-04 standard test method for determining apparent opening size of a geotextile.** West Conshohocken, PA: ASTM International.

American Society for Testing and Materials [ASTM]. 2009. **ASTM standard D4491-99a(2009) standard test methods for water permeability of geotextiles by permittivity.** West Conshohocken, PA: ASTM International.

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

Anderson, D.F.; Garisto, M.; Bourrut, J.; Schonbeck, M.W.; Jaye, R.; Wurzberger, A.; DeGregorio, R. 1996. **Evaluation of a paper mulch made from recycled materials as an alternative to plastic film mulch for vegetables.** Journal of Sustainable Agriculture. 7(1): 39-61.

Anderson, L.; Bryant, M. 1980. **Fish passage at road crossings: an annotated bibliography.** Gen. Tech. Rep. PNW-117. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 10 p.

Anderson, M.G. 1983. **Road-cut slope topography and stability relationships in St Lucia, West Indies.** Applied Geography. 3(2): 105-114.

Appelboom, T.W.; Chescheir, G.M.; Skaggs, R.W.; Hesterberg, D.L. 2002. **Management practices for sediment reduction from forest roads in the coastal plains.** Transactions of the American Society of Agricultural Engineers. 45(2): 337-344.

Arnáez, J.; Larrea, V. 1995. **Erosion processes and rates on road-sides of hill-roads (Iberian System, La Rioja, Spain).** Physics and Chemistry of the Earth. 20(3-4): 395-401.

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

Arnold, J. 1957. **Engineering aspects of forest soil.** In: An introduction to forest soils of the Douglas-fir region of the Pacific Northwest. Portland, OR: Western Forestry and Conservation Association. Chapter XIII. 15 p.

Arthur, M.A.; Coltharp, G.B.; Brown, D.L. 1998. **Effects of best management practices on forest streamwater quality in eastern Kentucky.** Journal of the American Water Resources Association. 34(3): 481-495.

Asaria, P.; Chisholm, D.; Mathers, C.; Ezzati, M.; Beaglehole, R. 2007. **Chronic disease prevention: health effects and financial costs of strategies to reduce salt intake and control tobacco use.** The Lancet. 370: 2044-2053.

Aubertin, G.M.; Patric, J.H. 1974. **Water quality after clearcutting a small watershed in West Virginia.** Journal of Environmental Quality. 3(3): 243-249.

Aust, W.M.; Blinn, C.R. 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: 5-36.

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.** Southern Journal of Applied Forestry. 35(3): 123-130.

Aycock, E. 2009. **The hydrology of road defects.** Alpharetta, GA: Atlanta Engineering Services Inc. http://161.58.137.120/Images/The%20Hydrology%20of%20Road%20Defects%20Submitted%2007%2006%2009%20to%20AIH.pdf (accessed December 29, 2011).

Bäckström, M. 2002. **Sediment transport in grassed swales during simulated runoff events.** Water Science and Technology. 45(7): 41-49.

Bäckström, M. 2003. **Grassed swales for stormwater pollution control during rain and snowmelt.** Water Science and Technology. 48(9): 123-132.

Bäckström, M.; Karlsson, S.; Bäckman, L.; Folkeson, L.; Lind, B. 2004. **Mobilisation of heavy metals by deicing salts in a roadside environment.** Water Research. 38: 720-732.

Bader, C.D. 1997. **Controlling dust.** Erosion Control, Journal of the International Erosion Control Association. 4(5): 20-32.

Bailey, R.W.; Copeland, O.L. 1961. **Vegetation and engineering structures in flood and erosion control.** In: Proceedings, 13th congress. Vienna, Austria: International Union of Forest Research Organizations. 23 p.

Balmer, W.E.; Williston, H.L.; Dissmeyer, G.E.; Pierce, C. 1976. **Site preparation – why and how.** Forest Management Bulletin SA-FB2. Atlanta, GA: U.S. Department of Agriculture, Forest Service, State and Private Forestry, Southeastern Area. 8 p.

Barber, M.E.; Downs, R.C. 1995. **Investigation of culvert hydraulics related to juvenile fish passage: research report.** Olympia, WA: Washington State Department of Transportation. 268 p.

Barfield, B.J.; Tollner, E.W.; Hayes, J.C. 1979. **Filtration of sediment by simulated vegetation: I. Steady-state flow with homogeneous sediment.** Transactions of the American Society of Agricultural Engineers. 22(3): 540-545, 548.

Barkley, D.G.; Blaser, R.E.; Schmidt, R.E. 1965. **Effect of mulches on microclimate and turf establishment.** Agronomy Journal. 57: 189-192.

Barling, R.D.; Moore, I.D. 1994. **Role of buffer strips in management of waterway pollution: a review.** Environmental Management. 18: 543-558.

Barnett, A.P.; Diseker, E.G.; Richardson, E.C. 1967. **Evaluation of mulching methods for erosion control on newly prepared and seeded highway backslopes.** Agronomy Journal. 59: 83-85.

Barraclough, D.; Nye, P.H. 1979. **The effect of molecular size on diffusion characteristics in soil.** Journal of Soil Science. 30: 29-42.

Barrett, M.; Lantin, A.; Austrheim-Smith, S. 2004. **Storm water pollutant removal in roadside vegetated buffer strips.** Transportation Research Record. 1890: 129-140.

Barrett, M.E. 2008. **Comparison of BMP performance using the international BMP database.** Journal of Irrigation and Drainage Engineering. 134(5): 556-561.

Barrett, M.E.; Kearney, J.E.; McCoy, T.G.; Malina, J.F., Jr.; Charbeneau, R.J. 1996. **An evaluation of the use and effectiveness of temporary sediment controls.** Report No. FHWA/TX-96/1943-2. Prepared for: Texas Department of Transportation. Austin, TX: University of Texas, Center for Transportation Research, Bureau of Engineering Research. 149 p.

Barrett, M.E.; Malina, J.F., Jr.; Charbeneau, R.J. 1998a. **An evaluation of geotextiles for temporary sediment control.** Water Environment Research. 70(3): 283-290.

Barrett, M.E.; Walsh, P.M.; Malina, J.F., Jr.; Charbeneau, R.J. 1998b. **Performance of vegetative controls for treating highway runoff.** Journal of Environmental Engineering. 124(11): 1121-1128.

Barton, C.; Kinkead, K. 2005. **Do erosion control and snakes mesh?** Journal of Soil and Water Conservation. 60(2): 33A-35A.

Barvenik, F.W. 1994. **Polyacrylamide characteristics related to soil applications.** Soil Science. 158(4): 235-243.

Batista, J.; Piechota, T.; James, D.; Stave, K. 2002. **Literature review: dust suppression and its environmental impacts.** In: Piechota, T.; Van Ee, J.; Batista, J.; Stave, K.; James, D., eds. Potential environmental impacts of dust suppressants: "avoiding another Times Beach." EPA/600/R-04/031. Appendix A. Washington, DC: U.S. Environmental Protection Agency; Las Vegas, NV: University of Nevada: 45-58.

Baver, L.D. 1956. **Soil physics.** New York: John Wiley and Sons, Inc. 489 p.

Beasley, R.S.; Miller, E.L.; Gough, S.C. 1984. **Forest road erosion in the Ouachita Mountains.** In: Peters, P.A.; Luchok, J., eds. Proceedings, mountain logging symposium. Morgantown, WV: West Virginia University: 202-213.

Beaty, A.N.S.; Raymond, G.P. 1995. **Concrete block road paving.** Concrete International. 17(5): 36-41.

Bell, J.R.; Hicks, R.G. 1980. **Evaluation of test methods and use criteria for geotechnical fabrics in highway applications.** Corvallis, OR: Oregon State University, Department of Engineering. 202 p.

Belt, G.H.; O'Laughlin, J.; Merrill, T. 1992. **Design of forest riparian buffer strips for the protection of water quality: analysis of scientific literature.** Idaho Forest, Wildlife, and Range Policy Analysis Group Report No. 8. Moscow, ID: University of Idaho. 35 p.

Ben-Hur, M. 1994. **Runoff, erosion, and polymer application in moving-sprinkler irrigation.** Soil Science. 158(4): 283-290.

Ben-Hur, M.; Malik, M.; Letey, J.; Mingelgrin, U. 1992. **Adsorption of polymers on clays as affected by clay charge and structure, polymer properties, and water quality.** Soil Science. 153(5): 349-356.

Benik, S.R.; Wilson, B.N.; Biesboer, D.D.; Hansen, B.; Stenlund, D. 2003. **Evaluation of erosion control products using natural rainfall events.** Journal of Soil and Water Conservation. 58(2): 98-105.

Bennett, F.L. 1994. **Gravel roadway maintenance in cold regions.** Report No. INE/TRC-94.05. Fairbanks, AK: University of Alaska Fairbanks, Institute of Northern Engineering, Transportation Research Center. 74 p.

Benoit, G.R.; Mostaghimi, S.; Young, R.A.; Lindstrom, M.J. 1986. **Tillage-residue effects on snow cover, soil water, temperature and frost.** Transactions of the American Society of Agricultural Engineers. 29(2): 473-479.

Bergeson, K.L.; Brocka, S.G. 1996. **Bentonite treatment for fugitive dust control.** In: 1996 semi-sesquicentennial transportation conference proceedings. Ames, IA: Iowa Department of Transportation; Iowa State University, Center for Transportation Research and Education. 3 p.

Berglund, E.R. 1976. **Seeding to control erosion along forest roads.** Extension Circular 885. Corvallis, OR: Oregon State University Extension Service. 19 p.

Berkshire Regional Planning Commission. 2001. **Unpaved roads BMP manual: a guidebook on how to improve water quality while addressing common problems.** Guidebook, Project 98-06/319. Pittsfield, MA: Berkshire Regional Planning Commission: 15-16.

Beschta, R.L. 1978. **Long-term patterns of sediment production following road construction and logging in the Oregon Coast Range.** Water Resources Research. 14(6): 1011-1016.

Bethlahmy, N.; Kidd, W.J., Jr. 1966. **Controlling soil movement from steep road fills.** Res. Note INT-45. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 4 p.

Beyers, J.L. 2004. **Postfire seeding for erosion control: effectiveness and impacts on native plant communities.** Conservation Biology. 18(4): 947-956.

Bhatt, R.; Khera, K.L. 2006. **Effect of tillage and mode of straw mulch application on soil erosion in the submontaneous tract of Punjab, India.** Soil and Tillage Research. 88: 107-115.

Bhattacharyya, R.; Davies, K.; Fullen, M.A.; Booth, C.A. 2008. **Effects of palm-mat geotextiles on the conservation of loamy sand soils in east Shropshire, UK.** Advances in GeoEcology. 39: 527-538.

Bhattacharyya, R.; Davies, K.; Fullen, M.A.; Booth, C.A.; Subedi, M. 2007. **Effects of palm-mat geotextiles on conservation of temperate agricultural soils in the United Kingdom.** Wolverhampton, UK: University of Wolverhampton, School of Applied Sciences. 1 p.

Bhattacharyya, R.; Fullen, M.A.; Davies, K.; Booth, C.A. 2010. **Use of palm-mat geotextiles for rainsplash erosion control.** Geomorphology. 119: 52-61.

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

Bill, M.D. 2005. **Channel morphology and substrate responses to high flows and road construction in forested mid-Appalachian watersheds.** Carbondale, IL: Southern Illinois University. 64 p. M.S. thesis.

Bjorneberg, D.L.; Santos, F.L.; Castanheira, N.S.; Martins, O.C.; Reis, J.L.; Aase, J.K.; Sojka, R.E. 2003. **Using polyacrylamide with sprinkler irrigation to improve infiltration.** Journal of Soil and Water Conservation. 58(5): 283-289.

Blanco-Canqui, H.; Gantzer, C.J.; Anderson, S.H.; Alberts, E.E. 2004. **Grass barriers for reduced concentrated flow induced soil and nutrient loss.** Soil Science Society of America Journal. 68: 1963-1972.

Blinn, C.R.; Dahlman, R.; Hislop, L.; Thompson, M.A. 1998. **Temporary stream and wetland crossing options for forest management.** Gen. Tech. Rep. NC-202. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Research Station. 61 p.

Blinn, C.R.; Kilgore, M.A. 2001. **Riparian management practices: a summary of state guidelines.** Journal of Forestry. 99(8): 11-17.

Blinn, C.R.; Taff, S.J.; Thompson, M.J.; Mlinar, M.; Townsend, N. 2000. **Assessing the financial effects associated with implementing Minnesota's timber harvesting and forest management guidelines.** Report #RR-0900. St. Paul, MN: Minnesota Forest Resources Council. 164 p.

Bloser, S.M.; Scheetz, B.E. 2012. **Sediment production from unpaved oil well access roads in the Allegheny National Forest. Final report.** University Park, PA: Pennsylvania State University, Center for Dirt and Gravel Road Studies. 62 p.

Bochet, E.; García-Fayos, P. 2004. **Factors controlling vegetation establishment and water erosion on motorway slopes in Valencia, Spain.** Restoration Ecology. 12(2): 166-174.

Bohn, H.L.; McNeal, B.L.; O'Connor, G.A. 1985. **Soil chemistry.** 2nd ed. New York: John Wiley and Sons, Inc. 341 p.

Boise State University. 1984. **Project completion report: sediment yield from cut and fill slopes, Silver Creek research road, Boise National Forest, Idaho.** Cooperative Agreement INT-80-003-CA. Boise, ID: Boise State University, Department of Geology and Geophysics. 97 p.

Bolander, P. 1997. **Chemical additives for dust control: what we have used and what we have learned.** Transportation Research Record. 1589: 42-49.

Bolander, P. 1999. **Laboratory testing of nontraditional additives for stabilization of roads and trail surfaces.** Transportation Research Record. 1652(2): 24-31.

Bolander, P.; Yamada, A. 1999. **Dust palliative selection and application guide.** Project Report 9977 1207-SDTDC. San Dimas, CA: U.S. Department of Agriculture, Forest Service, San Dimas Technology and Development Center. 20 p.

Bold, K.; Edwards, P.; Williard, K. 2007. **A case study of an erosion control practice: the broad-based dip.** In: Sessions, J.; Havill, Y., eds. Proceedings, international mountain logging and 13th Pacific Northwest skyline symposium. Corvallis, OR: Oregon State University: 65-71.

Bold, K.C. 2007. **Erosion control practices in managed forested watershed: vegetation establishment and broad-based dips.** Carbondale, IL: Southern Illinois University. 115 p. M.S. thesis.

Boubakari, M.; Morgan, R.P.C. 1999. **Contour grass strips for soil erosion control on steep lands: a laboratory evaluation.** Soil Use and Management. 15(1): 21-26.

Bouska, W.W. 2008. **Road crossing designs and their impact on fish assemblages and geomorphology of Great Plains streams.** Manhattan, KS: Kansas State University, College of Arts and Sciences. 158 p. M.S. thesis.

Bouska, W.W.; Keane, T.; Paukert, C.P. 2010. **The effects of road crossings on prairie stream habitat and function.** Journal of Freshwater Ecology. 25(4): 499-506.

Brabb, E.E. 1995. **The San Mateo County California GIS project for predicting the consequences of hazardous geologic processes.** In: Carrara, A.; Guzzetti, F., eds. Geographical information systems in assessing natural hazards. Advances in Natural and Technological Hazards Research. Norwell, MA: Kluwer Academic Publishers: 299-334.

Bracken, L.J.; Croke, J. 2007. **The concept of hydrological connectivity and its contribution to understanding runoff-dominated geomorphic systems.** Hydrologic Processes. 21: 1749-1763.

Brady, N.C. 1984. **The nature and properties of soils.** 9th ed. New York: MacMillan Publishing Company. 750 p.

Bren, L.J.; Turner, A.K. 1980. **Hydrologic output of small forested catchments: implications for management.** Australian Forestry. 43(2): 111-117.

Britton, S.L.; Robinson, K.M.; Barfield, B.J. 2001. **Modeling the effectiveness of silt fence.** In: Proceedings, seventh federal interagency sedimentation conference. [N.p.]: Advisory Committee on Water Information, Subcommittee on Sedimentation: V75-V82.

Broadmeadow, S.; Nisbet, T.R. 2004. **The effects of riparian forest management on the freshwater environment: a literature review of best management practice.** Hydrology and Earth System Sciences. 8(3): 286-305.

Brown, G.W.; Krygier, J.T. 1971. **Clear-cut logging and sediment production in the Oregon Coast Range.** Water Resources Research. 7(5): 1189-1198.

Brown, K. 2010. **Effectiveness of forestry best management practices in minimizing harvesting impacts on streamflow and sediment loading in low-gradient headwaters of the Gulf Coastal Plain.** Baton Rouge, LA: Louisiana State University. 75 p. M.S. thesis. http://etd.lsu.edu/docs/available/etd-09222010-164535/ (accessed September 12, 2012).

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.** Forest Ecology and Management. 310: 836-846.

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

Brown, W.E.; Caraco, D.S. 1997. **Muddy water in – muddy water out.** Watershed Protection Techniques. 2(3): 393-403.

Bryan, R.B. 1992. **The influence of some soil conditioners on soil properties: laboratory tests on Kenyan soil samples.** Soil Technology. 5: 225-247.

Brynn, D.J.; Clausen, J.C. 1991. **Postharvest assessment of Vermont's acceptable silvicultural management practices and water quality impacts.** Northern Journal of Applied Forestry. 8(4): 140-144.

Buchanan, J.R.; Yoder, D.C.; Denton, H.P.; Smoot, J.L. 2002. **Wood chips as a soil cover for construction sites with steep slopes.** Applied Engineering in Agriculture. 18(6): 679-683.

Burkhard, R.; Deletic, A.; Craig, A. 2000. **Techniques for water and wastewater management: a review of techniques and their integration in planning.** Urban Water. 2: 197-221.

Burroughs, E.R., Jr.; Chalfant, G.R.; Townsend, M.A. 1976. **Slope stability in road construction.** Portland, OR: U.S. Department of the Interior, Bureau of Land Management, Oregon State Office. 102 p.

Burroughs, E.R., Jr.; King, J.G. 1989. **Reduction of soil erosion on forest roads.** Gen. Tech. Rep. INT-264. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 21 p.

Burroughs, E.R., Jr.; Watts, F.J.; Haber, D.F. 1984a. **Surfacing to reduce erosion of forest roads built in granitic soils.** In: O'Loughlin, C.L.; Pearce, A.J., eds. Proceedings, symposium on effects of forest land use on erosion and slope stability. Honolulu, HI: University of Hawaii, East-West Center: 255-264.

Burroughs, E.R., Jr.; Watts, F.J.; King, J.G.; Haber, D.F.; Hansen, D.; Flerchinger, G. 1984b. **Relative effectiveness of rocked roads and ditches in reducing surface erosion.** In: Proceedings, 21st annual engineering, geology, and soils engineering symposium. Moscow, ID: University of Idaho: 251-263.

Burroughs, E.R., Jr.; Watts, F.J.; King, J.G.; Hansen, D. 1984c. **Relative effectiveness of fillslope treatments in reducing surface erosion, Horse Creek Road, Nezperce National Forest, Idaho.** Unpublished report on file with: U.S. Department of Agriculture, Forest Service, Moscow, ID. 34 p.

Butler, D.; Thedchanamoorthy, S.; Payne, J.A. 1992. **Aspects of surface sediment characteristics on an urban catchment in London.** Water Science and Technology. 25(8): 13-19.

Butzke, M.R. 1974. **Organic cationic and sodium chloride soil stabilization.** Ames, IA: Iowa State University. 155 p. M.S. thesis.

Büyüksönmez, F.; Yang, L.; Beighley, R.E. 2012. **Evaluation of best management practice products in preventing discharge of metals: a laboratory evaluation.** Journal of Environmental Quality. 41: 800-806.

Cafferata, P.H.; Coe, D.B.R.; Harris, R.R. 2007. **Water resource issues and solutions for forest roads in California.** In: Proceedings, American Institute of Hydrology 2007 annual meeting and international conference. Hydrological Science and Technology. 23(1-4): 39-56.

California Stormwater Quality Association. 2003. **Silt fence.** California Stormwater BMP Handbook. Menlo Park, CA: California Stormwater Quality Association. 8 p.

Cameron, A.L.; Henderson, L.E., eds. 1979. **Logging roads.** In: Environmental considerations for forest harvesting. Canberra, Australia: CSIRO: 27-37. Chapter 4.

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

Carrara, A.; Cardinali, M.; Detti, R.; Guzzetti, F.; Pasqui, V.; Reichenbach, P. 1991. **GIS techniques and statistical models in evaluating landslide hazard.** Earth Surface Processes and Landforms. 16: 427-445.

Carroll, C.; Merton, L.; Burger, P. 2000. **Impact of vegetative cover and slope on runoff, erosion, and water quality for field plots on a range of soil and spoil materials on central Queensland coal mines.** Australian Journal of Soil Research. 38(2): 313-327.

Carroll, R.G., Jr.; Rodencal, J.; Collin, J.G. 1992. **Geosynthetics in erosion control – the principles.** Geotextiles and Geomembranes. 11: 523-534.

Castelle, A.J.; Johnson, A.W. 2000. **Riparian vegetation effectiveness.** Tech. Bull. No. 799. Research Triangle Park, NC: National Council for Air and Stream Improvement. 26 p.

Castelle, A.J.; Johnson, A.W.; Conolly, C. 1994. **Wetland and stream buffer size requirements – a review.** Journal of Environmental Quality. 23: 878-882.

Cazzuffi, D.; Monferino, F.; Monti, R.; Rimoldi, P. 1994. **Experimental evaluation of the erosion on bare and geosynthetically protected slopes.** In: Rickson, R.J., ed. Conserving soil resources: European perspectives. Cambridge, UK: University Press: 413-421. Chapter 38.

Cederholm, C.J.; Reid, L.M.; Salo, E.O. 1981. **Cumulative effects of logging road sediment on salmonid populations in the Clearwater River, Jefferson County, Washington.** In: Proceedings of the conference, Salmon-spawning gravel: A renewable resource in the Pacific Northwest? Pullman, WA: Washington Water Research Center: 38-74.

Cerdà, A. 2007. **Soil water erosion on road embankments in eastern Spain.** Science of the Total Environment. 378: 151-155.

Chalmers, R.W. 1979. **Erosion from snig-tracks on granite/adamellite-derived soils.** Australian Forest Research Newsletter. 6: 142-143.

Chatterjea, K. 1994. **Dynamics of fluvial and slope processes in the changing geomorphic environment of Singapore.** Earth Surface Processes and Landforms. 19: 585-607.

Chatwin, S.C.; Howes, D.E.; Schwab, J.W.; Swanston, D.N. 1994. **A guide for management of landslide-prone terrain in the Pacific Northwest.** Victoria, BC: British Columbia Ministry of Forests. 229 p.

Chaubey, I.; Edwards, D.R.; Daniel, T.C.; Moore, P.A., Jr.; Nichols, D.J. 1995. **Effectiveness of vegetative filter strips in controlling losses of surface-applied poultry litter constituents.** Transactions of the American Society of Agricultural Engineers. 38(6): 1687-1692.

Chaudhari, K.; Flanagan, D.C. 1998. **Polyacrylamide effect on sediment yield, runoff, and seedling emergence on a steep slope.** In: Proceedings, ASAE annual international meeting. Paper No. 98-2155. St. Joseph, MI: American Society of Agricultural Engineers. 18 p.

Childs, E.C. 1969. **An introduction to the physical basis of soil water phenomena.** London, England: John Wiley and Sons, Ltd. 439 p.

Chopra, M.; Wanielista, M.; Gogo-Abite, I.; Hardin, M. 2010. **Index testing to support the stormwater management erosion and sediment control laboratory.** FDOT Project No. BD521, RPWO #5. Prepared for: Florida Department of Transportation. Orlando, FL: University of Central Florida, Stormwater Management Academy. 101 p.

Christ, M.J.; Driscoll, C.T.; Likens, G.E. 1999. **Watershed- and plot-scale tests of the mobile anion concept.** Biogeochemistry. 47: 335-353.

Claiborn, C.; Mitra, A.; Adams, G.; Bamesberger, L.; Allwine, G.; Kantamaneni, R.; Lamb, B.; Westberg, H. 1995. **Evaluation of PM10 emission rates from paved and unpaved roads using tracer techniques.** Atmospheric Environment. 29(10): 1075-1089.

Clarkin, K.; Keller, G.; Warhol, T.; Hixson, S. 2006. **Low-water crossings: geomorphic, biological, and engineering design considerations.** San Dimas, CA: U.S. Department of Agriculture, Forest Service, San Dimas Technology and Development Center. 366 p.

Claytor, R.A.; Schueler, T.R. 1996. **Design of stormwater filtering systems.** Silver Spring, MD: Center for Watershed Protection. 220 p.

Clean Water Act of 1972; 33 U.S.C. s/s 1251 et seq.

Clinnick, P.F. 1985. **Buffer strip management in forest operations: a review.** Australian Forestry. 48(1): 34-45.

Clinton, B.D.; Vose, J.M. 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.** Fort Collins, CO: Colorado State University. 110 p. M.S. thesis.

Coker, R.J.; Fahey, B.D.; Payne, J.J. 1993. **Fine sediment production from truck traffic, Queen Charlotte Forest, Marlborough Sounds, New Zealand.** Journal of Hydrology (N.Z.). 31(1): 56-64.

Collins, L.M.; Johnston, C.E. 1995. **Effectiveness of straw bale dams for erosion control in the Oakland Hills following the fire of 1991.** In: Keeley, J.E.; Scott, T., eds. Brushfires in California wildlands: ecology and resource management. Fairfield, WA: International Association of Wildland Fire: 171-183.

Colorado State Forest Service. 2010. **Forestry best management practices to protect water quality in Colorado.** Fort Collins, CO: Colorado State University. 28 p.

Colwell, S.; Horner, R.R.; Booth, D.B.; Gilvydis, D. 2000. **A survey of ditches along county roads for their potential to affect storm runoff water quality.** General Report. Seattle, WA: University of Washington, Center for Urban Water Resources Management. 26 p. + appendices.

Cook, D.F.; Nelson, S.D. 1986. **Effect of polyacrylamide on seedling emergence in crust-forming soils.** Soil Science. 141(5): 328-333.

Cook, M.J.; King, J.G. 1983. **Construction cost and erosion control effectiveness of filter windrows on fill slopes.** Res. Note INT-335. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 5 p.

Cook, W.L., Jr.; Hewlett, J.D. 1979. **The broad-based dip on Piedmont woods roads.** Southern Journal of Applied Forestry. 3(3): 77-81.

Copstead, R.L.; Johansen, D.K.; Moll, J. 1998. **Water/road interaction: introduction to surface cross drains.** Report 9877 1806-SDTDC. San Dimas, CA: U.S. Department of Agriculture, Forest Service, Technology and Development Program. 15 p.

Corbett, E.S.; Lynch, J.A.; Sopper, W.E. 1978. **Timber harvesting practices and water quality in the eastern United States.** Journal of Forestry. 76(8): 484-488.

Corley, C.J.; Frasier, G.W.; Trlica, M.J.; Smith, F.M.; Taylor, E.M., Jr. 1999. **Technical note: nitrogen and phosphorus in runoff from 2 montane riparian communities.** Journal of Range Management. 52(6): 600-605.

Cornell University Weed Ecology and Management Laboratory. [N.d.]. **Organic weed management.** http://weedecology.css.cornell.edu/manage/manage.php?id=23 (accessed September 5, 2012).

Cornish, P.M. 1975. **The impact of forest operations on water quality.** Tech. Paper No. 24. Sydney, Australia: Forestry Commission of New South Wales. 12 p.

Cornish, P.M. 2001. **The effects of roading, harvesting, and forest regeneration on streamwater turbidity levels in a moist eucalypt forest.** Forest Ecology and Management. 152: 293-312.

Correll, D.L. 1996. **Buffer zones and water quality protection: general principles.** In: Naycock, N.E.; Burt, T.P.; Goulding K.W.T.; Pinay, G., eds. Buffer zones: their processes and potential in water protection. Proceedings of the international conference on buffer zones. Hertfordshire, UK: Quest Environmental: 7-20.

Countess, R. 2001. **Methodology for estimating fugitive windblown and mechanically resuspended road dust emissions applicable for regional air quality modeling.** Prepared for: Western Governors' Association. Westlake Village, CA: Countess Environmental. 14 p.

Crawford, R.M.M. 1992. **Oxygen availability as an ecological limit to plant distribution.** In: Begon, M.; Fitter, A.H., eds. Advances in Ecological Research, vol. 23. Cambridge: Academic Press: 93-185.

Crawford, R.M.M. 1996. **Whole plant adaptations to fluctuating water tables.** Folia Geobotanica & Phytotaxonomica. 31: 7-24.

Crebbin, C. 1988. **Laboratory evaluation of geotextile performance in silt fence applications using a subsoil of glacial origin.** Seattle, WA: University of Washington. 90 p. M.S. thesis.

Croke, J.; Mockler, S. 2001. **Gully initiation and road-to-stream linkage in a forested catchment, southeastern Australia.** Earth Surface Processes and Landforms. 26: 205-217.

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: 257-268.

Croke, J.; Mockler, S.; Hairsine, P.; Fogarty, P. 2006. **Relative contributions of runoff and sediment from sources within a road prism and implications for total sediment delivery.** Earth Surface Processes and Landforms. 31: 457-468.

Croke, J.C.; Hairsine, P.B. 2006. **Sediment delivery in managed forests: a review.** Environmental Review. 14: 59-87.

Curry, R.A.; Scruton, D.A.; Clarke, K.D. 2002. **The thermal regimes of brook trout incubation habitats and evidence of changes during forestry operations.** Canadian Journal of Forest Research. 32: 1200-1207.

Dabney, S.M.; McGregor, K.C.; Meyer, L.D.; Grissinger, E.H.; Foster, G.R. 1993. **Vegetative barriers for runoff and sediment control.** In: Mitchell, J.K., ed. Proceedings, integrated resource management and landscape modification for environmental protection. ASAE Publ. 93-13. St. Joseph, MI: American Society of Agricultural Engineers: 60-70.

Dabney, S.M.; Meyer, L.D.; Harmon, W.C.; Alonso, C.V.; Foster, G.R. 1995. **Depositional patterns of sediment trapped by grass hedges.** Transactions of the American Society of Agricultural Engineers. 38(6): 1719-1729.

Daigle, P. 2010. **A summary of the environmental impacts of roads, management responses, and research gaps: a literature review.** BC Journal of Ecosystems and Management. 10(3): 65-89.

Dalton, P.A.; Smith, R.J.; Truong, P.N.V. 1996. **Vetiver grass hedges for erosion control on a cropped flood plain: hedge hydraulics.** Agricultural Water Management. 31: 91-104.

Damian, F. 2003. **Cross-drain placement to reduce sediment delivery from forest roads to streams.** Seattle, WA: University of Washington. 52 p. M.S. thesis.

Danfors, B. 1990. **Compaction in the subsoil.** In: Proceedings, 1990 international conference on agricultural engineering. Dusseldorf, Germany: VDI-Gesellschaft Agrartechnik, Verein Deutcher Ingenieure: 27-28.

Danfors, B. 1994. **Changes in subsoil porosity caused by heavy vehicles.** Soil and Tillage Research. 29: 135-144.

Daniels, R.B.; Gilliam, J.W. 1996. **Sediment and chemical load reduction by grass and riparian filters.** Soil Science Society of America Journal. 60: 246-251.

Davidson, R.A.; Davidson, C.F.; Roa-Espinosa, A. 2009. **Linear anionic polyacrylamide as an effective post-fire soil treatment: understanding the chemistry and physical science.** Journal of Soil and Water Conservation. 64(4): 243-252.

Davies, P.E.; Nelson, M. 1994. **Relationships between riparian buffer widths and the effects of logging on stream habitat, invertebrate community composition and fish abundance.** Australian Journal of Marine and Freshwater Research. 45: 1289-1305.

Davis, A.P.; Shokouhian, M.; Sharma, H.; Minami, C.; Winogradoff, D. 2003. **Water quality improvement through bioretention: lead, copper, and zinc removal.** Water Environment Research. 75: 73-82.

Dawson, A.; Kolisoja, P. 2006. **Managing rutting in low volume roads. Roadex III Project: northern periphery.** Lulea, Sweden: Swedish Road Administration. 26 p.

De Baets, S.; Poesen, J.; Gyssels, G.; Knapen, A. 2006. **Effects of grass roots on the erodibility of topsoils during concentrated flow.** Geomorphology. 76: 54-67.

De Ploey, J.; Savat, J.; Moeyersons, J. 1976. **The differential impact of some soil loss factors on flow, runoff creep, and rainwash.** Earth Surface Processes. 1(2): 151-161.

DeGraff, J.V. 1990. **Landslide dams from the November 1988 storm event in southern Thailand.** Landslide News. 4: 12-15.

Deletic, A. 2001. **Modelling of water and sediment transport over grassed areas.** Journal of Hydrology. 248: 168-182.

Deletic, A. 2005. **Sediment transport in urban runoff over grassed areas.** Journal of Hydrology. 301: 108-122.

Deletic, A.; Fletcher, T.D. 2006. **Performance of grass filters used for stormwater treatment – a field and modelling study.** Journal of Hydrology. 317: 261-275.

Demars, K.R.; Long, R.P. 2001. **Performance specifications for wood waste materials as an erosion control mulch and as a filter berm.** Report No. NETCR 25. Prepared for: New England Transportation Consortium. Storrs, CT: University of Connecticut, Department of Civil and Environmental Engineering. 28 p.

Demars, K.R.; Long, R.P.; Ives, J.R. 2000. **Use of wood waste materials for erosion control.** Report No. NETCR 20. Prepared for: New England Transportation Consortium. Storrs, CT: University of Connecticut, Department of Civil and Environmental Engineering. 63 p.

Denkler, P.; Bowders, J.; Loehr, E. 2000. **Geotextiles and loess: long-term flow.** In: Proceedings, mid-continent transportation symposium. Ames, IA: Iowa State University: 249-253.

Desbonnet, A.; Pogue, P.; Lee, V.; Wolff, N. 1994. **Vegetated buffers in the coastal zone – a summary review and bibliography.** Coastal Resources Tech. Rep. 2064. Narragansett, RI: University of Rhode Island Graduate School of Oceanography. 72 p.

Descheemaeker, K.; Nyssen, J.; Poesen, J.; Raes, D.; Haile, M.; Muys, B.; Deckers, S. 2006. **Runoff on slopes with restoring vegetation: a case study from the Tigray highlands, Ethiopia.** Journal of Hydrology. 331: 219-241.

Descroix, L.; Viramontes, D.; Vauclin, M.; Gonzalez Barrios, J.L.; Esteves, M. 2001. **Influence of soil surface features and vegetation on runoff and erosion in the Western Sierra Madre (Durango, northwest Mexico).** Catena. 43: 115-135.

Dickinson, W.T.; Wall, G.J. 1977. **Theme 2: The relationship between source-area erosion and sediment yield.** Hydrological Sciences Bulletin. 22(4): 527-530.

Dillaha, T.A.; Beasley, D.B.; Huggins, L.F. 1982. **Using the ANSWERS model to estimate sediment yields on construction sites.** Journal of Soil and Water Conservation. 37: 117-120.

Dillaha, T.A.; Hayes, J.C. 1992. **Vegetative filter strips: II. Application of design procedures.** Paper No. 92-2103. St. Joseph, MI: American Society of Agricultural Engineers. 18 p.

Dillaha, T.A.; Reneau, R.B.; Mostaghimi, S.; Lee, D. 1989. **Vegetative filter strips for agricultural nonpoint source pollution control.** Transactions of the American Society of Agricultural Engineers. 32(2): 513-519.

Dillaha, T.A.; Reneau, R.B.; Mostaghimi, S.; Shanholtz, V.O.; Magette, W.L. 1987. **Evaluating nutrient and sediment losses from agricultural lands: vegetative filter strips.** Project No. X-00315-01-0. Annapolis, MD: U.S. Environmental Protection Agency, Region III, Chesapeake Bay Liaison Office. 93 p.

Dillaha, T.A.; Sherrard, J.H.; Lee, D. 1986. **Long-term effectiveness and maintenance of vegetative filter strips.** Bull. 153. Blacksburg, VA: Virginia Polytechnic Institute and State University, Virginia Water Resources Research Center. 31 p.

Dillaha, T.A.; Sherrard, J.H.; Lee, D.; Mostaghimi, S.; Shanholtz, V.O. 1988. **Evaluation of vegetative filter strips as a best management practice for feed lots.** Journal (Water Pollution Control Federation). 60(7): 1231-1238.

Diseker, E.G.; McGinnis, J.T. 1967. **Evaluation of climatic, slope, and site factors on erosion from unprotected roadbanks.** Transactions of the American Society of Agricultural Engineers. 10: 9-11, 14.

Diseker, E.G.; Sheridan, J.M. 1971. **Predicting sediment yield from roadbanks.** Transactions of the American Society of Agricultural Engineers. 14(1): 102-105.

Donaldson, S. 2009. **Low impact development in northern Nevada: vegetated swales and buffers.** Fact Sheet FS-09-26. Reno, NV: University of Nevada Cooperative Extension, Nonpoint Education for Municipal Officials. http://www.unce.unr.edu/publications/files/nr/2009/fs0926.pdf (accessed July 9, 2012).

Döring, T.F.; Brandt, M.; Heß, J.; Finckh, M.R.; Saucke, H. 2005. **Effects of straw mulch on soil nitrate dynamics, weeds, yield and soil erosion in organically grown potatoes.** Field Crops Research. 94: 238-249.

Dorman, M.E.; Hartigan, J.P.; Steg, R.F.; Quasebarth, T.F. 1996. **Retention, detention, and overland flow for pollutant removal from highway stormwater runoff. Vol. 1: Research report.** Report No. FHWA-RD-96-095. McLean, VA: U.S. Department of Transportation, Federal Highway Administration, Office of Research, Development and Technology. 179 p.

Dosskey, M.G.; Helmers, M.J.; Eisenhauer, D.E. 2008. **A design aid for determining width of filter strips.** Journal of Soil and Water Conservation. 63(4): 232-241.

Dosskey, M.G.; Helmers, M.J.; Eisenhauer, D.E. 2011. **A design aid for sizing filter strips using buffer area ratio.** Journal of Soil and Water Conservation. 66(1): 29-39.

Dosskey, M.G.; Helmers, M.J.; Eisenhauer, D.E.; Franti, T.G.; Hoagland, K.D. 2002. **Assessment of concentrated flow through riparian buffers.** Journal of Soil and Water Conservation. 57(6): 336-343.

Doudoroff, P.; Katz, M. 1953. **Critical review of literature on the toxicity of industrial wastes and their components to fish: II. The metals, as salts.** Sewage and Industrial Wastes. 25: 802-839.

Douglas, I. 1967. **Natural and man-made erosion in the humid tropics of Australia, Malaysia, and Singapore.** Publications of the International Association of Scientific Hydrology. 75: 17-30.

Douglas, R.A.; Woodward, W.D.H.; Woodside, A.R. 2000. **Road contact stresses and forces under tires with low inflation pressure.** Canadian Journal of Civil Engineering. 27: 1248-1258.

Dudeck, A.E.; Swanson, N.P.; Dedrick, A.R. 1967. **Mulches for grass establishment on steep construction slopes.** Highway Research Record. 206: 53-59.

Dudeck, A.E.; Swanson, N.P.; Mielke, L.N.; Dedrick, A.R. 1970. **Mulches for grass establishment on fill slopes.** Agronomy Journal. 62(6): 810-812.

Duncan, S.H.; Ward, J.W.; Anderson, R.J. 1987. **A method for assessing landslide potential as an aid in forest road replacement.** Northwest Science. 61(3): 152-159.

Dunn, G.H.; Dabney, S.M. 1996. **Modulus of elasticity and moment of inertia of grass hedge stems.** Transactions of the American Society of Agricultural Engineers. 39(3): 947-952.

Dutton, A.L. 2000. **Process-based simulations of near-surface hydrologic response for a forested upland catchment: the impact of a road.** Stanford, CA: Stanford University. 151 p. M.S. thesis.

Dyck, R.I.J.; Stukel, J.J. 1976. **Fugitive dust emissions from trucks on unpaved roads.** Environmental Science and Technology. 10(10): 1046-1048.

Dyrness, C.T. 1967. **Mass soil movements in the H.J. Andrews Experimental Forest.** Res. Pap. PNW-42. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 13 p.

Dyrness, C.T. 1970. **Stabilization of newly constructed road backslopes by mulch and grass-legume treatments.** Res. Note PNW-123. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 5 p.

Dyrness, C.T. 1975. **Grass-legume mixtures for erosion control along forest roads in western Oregon.** Journal of Soil and Water Conservation. 30(4): 169-173.

Eaglin, G.S.; Hubert, W.A. 1993. **Effects of logging and roads on substrate and trout in streams of the Medicine Bow National Forest, Wyoming.** North American Journal of Fisheries Management. 13: 844-846.

Eck, R.W.; Morgan, P.J. 1987. **Culverts versus dips in the Appalachian region: a performance-based decision-making guide.** In: Proceedings, fourth international conference on low-volume roads. Transportation Research Record 1106. Washington, DC: National Research Council, Transportation Research Board: 330-340.

Edmister, J.O.; Gray, J.W. 1948. **The toxicity threshold for three chlorides and three acids to the fry of whitefish and yellow pickerel.** Progressive Fish-Culturist. 10: 105-106.

Edwards, P.J. 2003. **Forestry best management practices: where we've been, where we're going.** In: Proceedings, 2003 Penn State forest resources issues conference. University Park, PA: Pennsylvania State University: 26-33.

Edwards, P.J.; Evans, G.L. 2004. **Giving greater consideration to cross-drainage discharge from forest roads.** In: Proceedings, riparian ecosystems and buffers: multi-scale structure, function, and management. AWRA summer specialty conference. Middleburg, VA: American Water Resources Association. 6 p.

Egan, A.F. 1999. **Forest roads: where soil and water don't mix.** Journal of Forestry. 97(8): 18-21.

Elliot, W.J.; Edwards, P.J.; Foltz, R.B. 2014. **Research related to roads in USDA experimental forests.** In: Hayes, D.C.; Stout, S.L.; Hoover, A.P.; Crawford, R.H., eds. USDA Forest Service experimental forests and ranges: research for the long term. New York: Springer: 367-385. Chapter 16.

Elliot, W.J.; Foltz, R.B.; Luce, C.H. 1999. **Modeling low-volume road erosion.** Transportation Research Record. 1652: 244-249.

Elliot, W.J.; Foltz, R.B.; Robichaud, P.R. 2009. **Recent findings related to measuring and modeling forest road erosion.** In: Anderssen, R.S.; Braddock, R.D.; Newham, L.T.H., eds. Proceedings, 18th world IMACS congress and MODSIM09 international congress on modelling and simulation. Melbourne, Australia: Modelling and Simulation Society of Australia and New Zealand; International Association for Mathematics and Computers in Simulation: 4078-4084.

Elliot, W.J.; Tysdal, L.M. 1999. **Understanding and reducing erosion from insloping roads.** Journal of Forestry. 97(8): 30-34.

El-Morsy, E.A.; Malik, M.; Letey, J. 1991. **Polymer effects on the hydraulic conductivity of saline and sodic soil conditions.** Soil Science. 151(6): 430-435.

Elwell, H.A.; Stocking, M.A. 1974. **Rainfall parameters and a cover model to predict runoff and soil loss from grazing trials in the Rhodesian Sandveld.** Proceedings of the Annual Congresses of the Grassland Society of Southern Africa. 9: 157-164.

Elwell, H.A.; Stocking, M.A. 1976. **Vegetal cover to estimate soil erosion hazard in Rhodesia.** Geoderma. 15: 61-70.

Engen, R.J.; Denison, J.G.; Goldschmid, O. 1976. **Chemical changes in lignin sulfonates during exposure to seawater.** Cellulose Chemistry and Technology. 10: 219-234.

Erman, D.C.; Newbold, J.D.; Roby, K.B. 1977. **Evaluation of streamside buffer strips for protecting aquatic organisms.** Contribution No. 165. Davis, CA: University of California, California Water Resources Center. 48 p.

Ernst, W.H.O. 1990. **Ecophysiology of plants in waterlogged and flooded environments.** Aquatic Botany. 38: 73-90.

Ettinger, W.S. 1987. **Impacts of a chemical dust suppressant/soil stabilizer on the physical and biological characteristics of a stream.** Journal of Soil and Water Conservation. 42(2): 111-114.

Ettlin, L.; Stewart, B. 1993. **Yard debris compost for erosion control.** BioCycle. 34(12): 46-47.

Etyemezian, V.; Kuhns, H.; Gillies, J.; Green, M.; Pitchford, M.; Watson, J. 2003. **Vehicle-based road dust emission measurement: I. Methods and calibration.** Atmospheric Environment. 37: 4559-4571.

Ewing, K. 1996. **Tolerance of four wetland plant species to flooding and sediment deposition.** Environmental and Experimental Botany. 36(2): 131-146.

Farias, R.J.C.; Palmeira, E.M.; Carvalho, J.C. 2006. **Performance of geotextile silt fences in large flume tests.** Geosynthetics International. 13(4): 133-144.

Faucette, L.B.; Cardoso-Gendreau, F.A.; Codling, E.; Sadeghi, A.M.; Pachepsky, Y.A.; Shelton, D.R. 2009a. **Storm water pollutant removal performance of compost filter socks.** Journal of Environmental Quality. 38: 1233-1239.

Faucette, L.B.; Governo, J.; Tyler, R.; Gigley, G.; Jordan, C.F.; Lockaby, B.G. 2009b. **Performance of compost filter socks and conventional sediment control barriers used for perimeter control on construction sites.** Journal of Soil and Water Conservation. 64(1): 81-88.

Faucette, L.B.; Jordan, C.F.; Risse, L.M.; Cabrera, M.; Coleman, D.C.; West, L.T. 2005. **Evaluation of stormwater from compost and conventional erosion control practices in construction activities.** Journal of Soil and Water Conservation. 60(6): 288-297.

Faucette, L.B.; Sefton, K.A.; Sadeghi, A.M.; Rowland, R.A. 2008. **Sediment and phosphorus removal from simulated storm runoff with compost filter socks and silt fence.** Journal of Soil and Water Conservation. 63(4): 257-264.

Faucette, L.B.; Shields, F.; Kurtz, K. 2006. **Removing storm water pollutants and determining relations between hydraulic flow-through rates, pollutant removal efficiency, and physical characteristics of compost filter media.** In: Fowler, D.L., comp., ed. Proceedings, second interagency conference on research in the watersheds. Otto, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station,Coweeta Hydrologic Laboratory: 107-115.

Faucette, L.B.; Tyler, R. 2006. **Organic BMPs used for stormwater management.** In: Proceedings, annual international erosion control conference. Denver, CO: International Erosion Control Association: 101-108.

Federation of Canadian Municipalities and National Research Council. 2005. **Dust control for unpaved roads.** National Guide to Sustainable Municipal Infrastructure (Infraguide). Ottawa, ON: National Research Council of Canada. 42 p.

Feick, G.; Horne, R.A.; Yeaple, D. 1972. **Release of mercury from contaminated freshwater sediments by the runoff of road deicing salt.** Science. 175(4026): 1142-1143.

Fennessy, M.S.; Cronk, J.K. 1997. **The effectiveness and restoration potential of riparian ecotones for the management of nonpoint source pollution, particularly nitrate.** Critical Reviews in Environmental Science and Technology. 27(4): 285-317.

Fiener, P.; Auerswald, K. 2006. **Seasonal variation of grassed waterway effectiveness in reducing runoff and sediment delivery from agricultural watersheds in temperate Europe.** Soil and Tillage Research. 87: 48-58.

Fisher, L.M. 1939. **Pollution kills fish.** Scientific American. 160: 144-146.

Fisher, L.S.; Jarrett, A.R. 1984. **Sediment retention efficiency of synthetic filter fabrics.** Transactions of the American Society of Agricultural Engineers. 27(2): 429-436.

Flanagan, D.C.; Chaudhari, K.; Norton, L.D. 2002a. **Polyacrylamide soil amendment effects on runoff and sediment yield on steep slopes: Part I. Simulated rainfall conditions.** Transactions of the American Society of Agricultural Engineers. 45(5): 1327-1337.

Flanagan, D.C.; Chaudhari, K.; Norton, L.D. 2002b. **Polyacrylamide soil amendment effects on runoff and sediment yield on steep slopes: Part II. Natural rainfall conditions.** Transactions of the American Society of Agricultural Engineers. 45(5): 1-13.

Flanagan, D.C.; Norton, L.D.; Shainberg, I. 1992. **Water chemistry effects on infiltration, runoff, and erosion.** Paper No. 92-2053. St. Joseph, MI: American Society of Agricultural Engineers. 13 p.

Flanagan, D.C.; Norton, L.D.; Shainberg, I. 1997. **Effect of water chemistry and soil amendments on a silt loam soil: Part II. Soil erosion.** Transactions of the American Society of Agricultural Engineers. 40(6): 1555-1561.

Florida Industrial and Phosphate Research Institute. 2010. **Phosphate primer.** Bartow, FL: University of South Florida Polytechnic, Florida Industrial and Phosphate Research Institute. http://www1.fipr.state.fl.us/PhosphatePrimer (accessed August 19, 2013).

Foley, G.; Cropley, S.; Giummarra, G. 1996. **Road dust control techniques: evaluation of chemical dust suppressants' performance.** Special Report 54. Melbourne, Australia: ARRB Transport Research Ltd. 143 p.

Foltz, R.B. 1994. **Reducing tire pressure reduces sediment.** Tech Tips: Roads/Timber. San Dimas, CA: U.S. Department of Agriculture, Forest Service, Technology and Development Program. 1 p.

Foltz, R.B. 1995. **Sediment reduction from the use of lowered tire pressures.** SAE Technical Paper Series, 942244. In: Central tire inflation systems: managing the vehicle to surface. Warrendale, PA: Society of Automotive Engineers: 47-52.

Foltz, R.B. 1999. **Traffic and no-traffic on an aggregate surfaced road: sediment production differences.** In: Proceedings, seminar on environmentally sound forest roads and wood transport. Rome, Italy: Food and Agriculture Organization of the United Nations: 195-204.

Foltz, R.B.; Burroughs, E.R., Jr. 1990. **Sediment production from forest roads with wheel ruts.** In: Watershed planning and analysis in action symposium: proceedings, IR conference. Reston, VA: American Society of Civil Engineers: 266-275.

Foltz, R.B.; Burroughs, E.R., Jr. 1991. **A test of normal tire pressure and reduced tire pressure on forest roads: sedimentation effects.** In: Stokes, B.J.; Rawlins, C.L., eds. Proceedings, forestry and environment – engineering solutions. ASAE Publ. 09-91. St. Joseph, MI: American Society of Agricultural Engineers: 103-112.

Foltz, R.B.; Copeland, N.S. 2009. **Evaluating the efficacy of wood shreds for mitigating erosion.** Journal of Environmental Management. 90: 779-785.

Foltz, R.B.; Dooley, J.H. 2003. **Comparison of erosion reduction between wood strands and agricultural straw.** Transactions of the American Society of Agricultural Engineers. 46(5): 1389-1396.

Foltz, R.B.; Elliot, W.J. 1997. **Effect of lowered tire pressures on road erosion.** Paper No. 970638. Transportation Research Record. 1589: 19-25.

Foltz, R.B.; Truebe, M.A. 1995. **Effect of aggregate quality on sediment production from a forest road.** In: Low volume roads: proceedings, sixth conference conducted by the Transportation Research Board. Washington, DC: Transportation Research Board: 49-57.

Foltz, R.B.; Wagenbrenner, N.S. 2010. **An evaluation of three wood shred blends for post-fire erosion control using indoor simulated rain events.** Catena. 80: 86-94.

Forman, R.T.T.; Alexander, L.E. 1998. **Roads and their major ecological effects.** Annual Review of Ecology and Systematics. 29: 207-231.

Fox, D.; Bryan, R.B. 1992. **Influence of a polyacrylamide soil conditioner on runoff generation and soil erosion: field tests in Baringo District, Kenya.** Soil Technology. 5: 101-119.

Fox, D.E. 1972. **Ammonium lignosulfonates as dust palliatives and surface improvement agents for unpaved secondary roads.** Ames, IA: Iowa State University. 112 p. M.S. thesis.

Fox, D.M.; Bryan, R.B. 1999. **The relationship of soil loss by interrill erosion to slope gradient.** Catena. 38: 211-222.

Fredriksen, R.L. 1970. **Erosion and sedimentation following road construction and timber harvest on unstable soils in three small western Oregon watersheds.** Res. Pap. PNW-104. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 15 p.

Freebairn, D.M.; Wockner, G.H.; Silburn, D.M. 1986. **Effects of catchment management on runoff, water quality and yield potential from vertisols.** Agricultural Water Management. 12: 1-19.

Furniss, M.J.; Staab, B.P.; Hazelhurst, S.; Clifton, C.F.; Roby, K.B.; Ilhardt, B.L.; Larry, E.B.; Todd, A.H.; Reid, L.M.; Hines, S.J.; Bennett, K.A.; Luce, C.H.; Edwards, P.J. 2010. **Water, climate change, and forests: watershed stewardship for a changing climate.** Gen. Tech. Rep. PNW-812. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 75 p.

Gatto, L.W. 1998. **Freeze-thaw effects on vehicular ruts and natural rills: importance to soil-erosion and terrain modelling.** In: Summer, W.; Klaghofer, E.; Zhang, W., eds. Modelling soil erosion, sediment transport and closely related hydrological processes: proceedings of a symposium. IAHS Publ. No. 249. Wallingford, UK: International Association of Hydrological Sciences: 71-79.

Gebhart, D.L.; Hale, T.A.; Michaels-Busch, K. 1996. **Dust control material performance on unsurfaced roadways and tank trails.** USAEC/USACERL Tech. Rep. Champaign, IL: U.S. Army Construction Engineering Research Laboratories. 34 p.

Gharabaghi, B.; Rudra, R.P.; Goel, P.K. 2006. **Effectiveness of vegetative filter strips in removal of sediments from overland flow.** Water Quality Research Journal of Canada. 41(3): 275-282.

Ghate, S.R.; Burtle, G.J.; Vellidis, G.; Newton, G.L. 1997. **Effectiveness of grass strips to filter catfish (Ictalurus punctatus) pond effluent.** Aquacultural Engineering. 16: 149-159.

Gifford, G.F. 1985. **Cover allocation in rangeland watershed management (a review).** In: Jones, E.B.; Ward, T.J., eds. Proceedings, watershed management in the eighties. New York: American Society of Civil Engineers: 23-31.

Gillies, J.A.; Etyemezian, V.; Kuhns, H.; Nikolic, D.; Gillette, D.A. 2005. **Effect of vehicle characteristics on unpaved road dust emissions.** Atmospheric Environment. 39: 2341-2347.

Gillies, J.A.; Watson, J.G.; Rogers, C.F.; DuBois, D.; Chow, J.C.; Langston, R.; Sweet, J. 1999. **Long-term efficiencies of dust suppressants to reduce PM10 emissions from unpaved roads.** Journal of Air and Waste Management Association. 49: 3-16.

Giummarra, G.J.; Foley, G.; Cropley, S. 1997. **Dust control: Australasian experiences with various chemical additives.** Transportation Research Record. 1589: 50-53.

Goldberg, J.S.; Woelfel, C.; Chinn, A.; McDaniel, M.; Paston, C.; Thibert, N.; She, N. 1993. **Dayton Avenue swale biofiltration study.** Seattle, WA: Seattle Engineering Department, Drainage and Wastewater Utility. 36 p. + appendices.

Grace, J.M., III. 2002a. **Effectiveness of vegetation in erosion control from forest road sideslopes.** Transactions of the American Society of Agricultural Engineers. 45(3): 681-685.

Grace, J.M., III. 2002b. **Overview of best management practices related to forest roads: the southern states.** In: Proceedings, ASAE annual international meeting. Paper No. 025013. St. Joseph, MI: American Society of Agricultural Engineers. 14 p.

Grace, J.M., III. 2003. **Minimizing the impacts of the forest road system.** In: Proceedings, conference 34. Denver, CO: International Erosion Control Association: 301-310.

Grace, J.M., III. 2005. **Forest operations and water quality in the South.** Transactions of the American Society of Agricultural Engineers. 48(2): 871-880.

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

Grace, J.M., III; Wilhoit, J.; Rummer, R.; Stokes, B. 1996. **Surface erosion control techniques on newly constructed forest roads.** In: Proceedings, ASAE annual international meeting. Paper No. 965017. St. Joseph, MI: American Society of Agricultural Engineers. 14 p.

Granato, G.E.; Church, P.E.; Stone, V.J. 1995. **Mobilization of major and trace constituents of highway runoff in groundwater potentially caused by deicing chemical migration.** Transportation Research Record. 1483: 92-104.

Graynoth, E. 1979. **Effects of logging on stream environments and faunas in Nelson.** New Zealand Journal of Marine and Freshwater Research. 13(1): 79-109.

Green, V.S.; Stott, D.E. 2001. **Polyacrylamide: a review of the use, effectiveness, and cost of a soil erosion control amendment.** In: Stott, D.E.; Mohtar, R.H.; Steinhardt, G.C., eds. Sustaining the global farm: proceedings, tenth international soil conservation organization meeting. West Lafayette, IN: Purdue University: 384-389.

Green, V.S.; Stott, D.E.; Norton, L.D.; Graveel, J.G. 2000. **Polyacrylamide molecular weight and charge effects on infiltration under simulated rainfall.** Soil Science of America Journal. 64: 1786-1791.

Gregory, K.J.; Brookes, A. 1983. **Hydrogeomorphology downstream from bridges.** Applied Geography. 3: 145-159.

Griffin, J.M.; West, J.L. 1976. **Acute toxicity of ammonia-base neutral sulfite pulp mill waste liquor to rainbow trout.** Bulletin of Environmental Contamination and Toxicology. 15(5): 608-612.

Grismer, M.E.; Hogan, M.P. 2005. **Simulated rainfall evaluation of revegetation/mulch erosion control in the Lake Tahoe basin – 3: Soil treatment effects.** Land Degradation and Development. 16: 489-501.

Groen, A.H.; Woods, S.W. 2008. **Effectiveness of aerial seeding and straw mulch for reducing post-wildfire erosion, north-western Montana, USA.** International Journal of Wildland Fire. 17: 559-571.

Grushecky, S.T.; Spong, B.D.; McGill, D.W.; Edwards, J.W. 2009. **Reducing sediments from skid roads in West Virginia using fiber mats.** Northern Journal of Applied Forestry. 26(3): 118-121.

Gumiere, S.J.; Bissonnais, Y.L.; Raclot, D.; Cheviron, B. 2011. **Vegetated filter effects on sedimentological connectivity of agricultural catchments in erosion modelling: a review.** Earth Surface Processes and Landforms. 36: 3-19.

Gutierrez, J.; Hernandez, I.I. 1996. **Runoff and interrill erosion as affected by grass cover in a semi-arid rangeland of northern Mexico.** Journal of Arid Environments. 34: 287-295.

Gwynne, C.S. 1950. **Terraced highway side slopes in loess, southwestern Iowa.** Geological Society of America Bulletin. 61(12): 1347-1354.

Gyssels, G.; Poesen, J. 2003. **The importance of plant root characteristics in controlling concentrated flow erosion rates.** Earth Surface Processes and Landforms. 28: 371-384.

Gyssels, G.; Poesen, J.; Bochet, E.; Li, Y. 2005. **Impact of plant roots on the resistance of soils to erosion by water: a review.** Progress in Physical Geography. 29(2): 189-217.

Hafterson, H.D. 1973. **Dip design.** Engineering Field Notes. 5(10): 1-18. http://www.fs.fed.us/t-d/pubs/pdfpubs/pdf73713807/pdf73713807.pdf (accessed June 29, 2016).

Hagans, D.K.; Weaver, W.E. 1987. **Magnitude, cause and basin response to fluvial erosion, Redwood Creek basin, northern California.** In: Beschata, R.L.; Blinn, T.; Grant, G.E.; Swanson, F.J.; Ice, G.G., eds. Erosion and sedimentation in the Pacific Rim. Publ. No. 165. Wallingford, UK: International Association of Hydrologic Sciences Press: 419-428.

Hagans, D.K.; Weaver, W.E.; Madej, M.A. 1986. **Long term on-site and off-site effects of logging and erosion in the Redwood Creek basin, northern California.** In: Proceedings, American Geophysics Union meeting on cumulative effects. Tech. Bull. 490. New York: National Council of the Paper Industry for Air and Stream Improvement: 38-65.

Hairsine, P.B. 1997. **Controlling sediment and nutrient movement within catchments.** Industry Report 97/9. Melbourne, Australia: Cooperative Research Centre for Catchment Hydrology. 22 p.

Hairsine, P.B.; Rose, C.W. 1992. **Modeling water erosion due to overland flow using physical principles: 1. Sheet flow.** Water Resources Research. 28(1): 237-243.

Håkansson, I.; Petelkau, H. 1994. **Benefits of a limited axle load.** In: Soane, B.D.; Ouwerkerk, C.V., eds. Soil compaction in crop production. Amsterdam, the Netherlands: Elsevier Science B.V.: 479-499. Chapter 20.

Hammond, C.; Hall, D.; Miller, S.; Swetick, P. 1992. **Level I stability analysis (LISA) documentation for version 2.0.** Gen. Tech. Rep. INT-285. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 130 p.

Hamons, G. 2007. **Modeling sediment movement in forested watersheds using hill-slope attributes.** Morgantown, WV: West Virginia University. 123 p. M.S. thesis.

Hanes, R.E.; Zelazny, L.W.; Blaser, R.E. 1970. **Effects of deicing salts on water quality and biota: literature review and recommended research.** National Cooperative Highway Research Program Report 91. Washington, DC: National Research Council, Highway Research Board, National Academy of Sciences. 70 p.

Hanes, R.E.; Zelazny, L.W.; Verghese, K.G.; Bosshart, R.P.; Carson, E.W., Jr.; Blaser, R.E.; Wolf, D.D. 1976. **Effects of deicing salts on plant biota and soil: experimental phase.** National Cooperative Highway Research Program Report 170. Washington, DC: National Research Council, Transportation Research Board. 88 p.

Harbor, J. 1999. **Engineering geomorphology at the cutting edge of land disturbance: erosion and sediment control on construction sites.** Geomorphology. 31: 247-263.

Harding, M.V. 1988. **Erosion control effectiveness: comparative studies of alternative mulching techniques.** In: Proceedings, 67th annual meeting of the Transportation Research Board. Paper No. 870244. Washington, DC: Transportation Research Board. 11 p.

Harr, R.D.; Nichols, R.A. 1993. **Stabilizing forest roads to help restore fish habitats: a northwest Washington example.** Fisheries. 18(4): 18-22.

Harrison, B. 2011. **Erosion from a cross country gas pipeline in the central Appalachians.** Carbondale, IL: Southern Illinois University. 69 p. M.S. thesis.

Haupt, H.F. 1959a. **A method for controlling sediment from logging roads.** Misc. Publ. No. 22. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 22 p.

Haupt, H.F. 1959b. **Road and slope characteristics affecting sediment movement from logging roads.** Journal of Forestry. 57(5): 329-332.

Haupt, H.F.; Kidd, W.J., Jr. 1965. **Good logging practices reduce sedimentation in central Idaho.** Journal of Forestry. 63: 664-670.

Haussman, R.F.; Pruett, E.W. 1978. **Permanent logging roads for better woodlot management.** Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Area State and Private Forestry. 43 p.

Hayes, J.C.; Barfield, B.J.; Barnhisel, R.I. 1984. **Performance of grass filters under laboratory and field conditions.** Transactions of the American Society of Agricultural Engineers. 27(5): 1321-1331.

Hayes, J.C.; Hairston, J.E. 1983. **Modeling the long-term effectiveness of vegetative filters as on-site sediment controls.** Paper No. 83-2081. St. Joseph, MI: American Society of Agricultural Engineers. 27 p.

Hayes, S.A.; McLaughlin, R.A.; Osmond, D.L. 2005. **Polyacrylamide use for erosion and turbidity control on construction sites.** Journal of Soil and Water Conservation. 60(4): 193-199.

Heffner, K. 1997. **Water quality effects of three dust-abatement compounds.** Engineering Field Notes. 29(Jan.-April): 35-43. http://www.fs.fed.us/t-d/pubs/pdfpubs/pdf97713801/pdf97713801.pdf (accessed June 29, 2016).

Helalia, A.M.; Letey, J. 1989. **Effects of different polymers on seedling emergence, aggregate stability, and crust hardness.** Soil Science. 148(3): 199-203.

Heller, V.G. 1932. **Saline and alkaline drinking waters.** The Journal of Nutrition. 5(4): 421-429.

Heller, V.G. 1933. **The effect of saline and alkaline waters on domestic animals.** Oklahoma Agricultural Experiment Station Bull. No. 217. Stillwater, OK: Oklahoma Agricultural and Mechanical College, Agricultural Experiment Station. 23 p.

Heller, V.G.; Larwood, C.H. 1930. **Saline drinking water.** Science. 71(1834): 223-224.

Helms, J.A.; Hipkin, C. 1986. **Effects of soil compaction on tree volume in a California ponderosa pine plantation.** Western Journal of Applied Forestry. 1(4): 121-124.

Henderson, Z.L. 2001. **The influence of geology and timber harvesting on non-perennial stream channels on the Appalachian Plateau.** University Park, PA: Pennsylvania State University. 208 p. M.S. thesis.

Henry, K.S.; Walsh, M.R.; Morin, S.H. 1999. **Selection of silt fence filter to retain suspended toxic particles.** Geotextiles and Geomembranes. 17: 371-387.

Hewlett, J.D. 1971. **Comments on the catchment experiment to determine vegetal effects on water yield.** Water Resources Bulletin. 7(2): 376-381.

Hipps, N.A.; Hazelden, J.; Fairall, G.B.N. 1990. **Control of erosion in a mature orchard.** Soil Use and Management. 6: 32-35.

Hislop, L.E. 1996a. **Improving access and environmental sensitivity with portable surfaces on low volume roads.** 9624 1211-SDTDC. Washington, DC: U.S. Department of Agriculture, Forest Service, Technology and Development Program. 11 p.

Hislop, L.E. 1996b. **Portable surfaces for crossing unstable roadbeds.** In: Ritter, M.A.; Duwadi, S.R.; Lee, P.D.H., eds. Proceedings, national conference on wood transportation structures. Gen. Tech. Rep. FPL-GTR-94. Madison, WI: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory: 138-144.

Hoffman, D.J.; Eastin, W.C., Jr. 1981. **Effects of industrial effluents, heavy metals, and organic solvents on mallard embryo development.** Toxicology Letters. 9: 35-40.

Holtz, R.D.; Christopher, B.R.; Berg, R.R. 1998. **Geosynthetic design and construction guidelines.** Publ. No. FHWA HI-95-038. Washington, DC: U.S. Department of Transportation, Federal Highway Administration, National Highway Institute. 460 p.

Hoover, J.M. 1987. **Dust control on construction sites.** Report No. FHWA-AZ86-807. Phoenix, AZ: Arizona Department of Transportation. 61 p.

Hoover, J.M.; Bergeson, K.H.; Fox, D.E.; Denny, C.K.; Handy, R.L. 1973. **Surface improvement and dust palliation of unpaved secondary roads and streets.** Final report. ISU-ERI-AMES-72316. ERI Project 856-S. Ames, IA: Iowa State University, Engineering Research Institute. 228 p.

Hoover, J.M.; Pitt, J.M.; Lustig, M.T.; Fox, D.E. 1981. **Mission-oriented dust control and surface improvement processes for unpaved roads.** Final report. ISU-ERI-AMES-81225. ERI Project 1308. Ames, IA: Iowa State University, Engineering Research Institute. 216 p.

Hoover, J.M.; Squier, L.D.; Solomon, P.L.; Handy, R.L. 1975. **Evaluation of chemically stabilized secondary roads – Linn County, Iowa.** Progress Report 2. ISU-ERI-Ames-76202. ERI Project 1049S. Ames, IA: Iowa State University, Engineering Research Institute. 102 p.

Hornbeck, J.W.; Kochenderfer, J.N. 2001. **A century of lessons about water resources in northeastern forests.** In: A monumental event. Proceedings, Society of American Foresters 2000 national convention. Bethesda, MD: Society of American Foresters: 31-37.

Horner, R.R.; Guedry, J.; Kortenhof, M.H. 1990. **Improving the cost effectiveness of highway construction site erosion and pollution control.** Report No. WA-RD 200.1. Olympia, WA: Washington State Department of Transportation. 64 p.

Horner, R.R.; Skupien, J.J.; Livingston, E.H.; Shaver, H.E. 1994. **Urban runoff treatment practices.** In: Fundamentals of urban runoff management: technical and institutional issues. Washington, DC: Terrene Institute: 113-142. Chapter 8.

Hösl, R.; Strauss, P.; Glade, T. 2012. **Man-made linear flow paths at catchment scale: identification, factors, and consequences for the efficiency of vegetated filter strips.** Landscape and Urban Planning. 104: 245-252.

Hudson, N.W. 1957. **Erosion control research: progress report on experiments at Henderson Research Station 1953-1956.** Zimbabwe Agricultural Journal. 54: 297-323.

Hunter, J.V.; Sabatino, T.; Gomperts, R.; MacKenzie, M.J. 1979. **Contribution of urban runoff to hydrocarbon pollution.** Journal (Water Pollution Control Federation). 51(8): 2129-2138.

Hursh, C.R. 1939. **Roadbank stabilization at low cost: a progress report.** Tech. Note 38. Asheville, NC: U.S. Department of Agriculture, Forest Service, Appalachian Forest Experiment Station. 20 p.

Ice, G. [N.d.]. **Forty years of lessons learned about the impacts of forest practices on watershed hydrology and water quality.** Available at http://faculty.washington.edu/emer/eic/Forty%20 Years%20of%20Lessons.htm (accessed May 19, 2016).

Indiana Department of Natural Resources. 1992. **Indiana handbook for erosion controls in developing areas: guidelines for protecting water quality through the control of soil and erosion and sedimentation on construction sites. Practice 3.75, straw bale dam (straw bale filter).** Indianapolis, IN: Indiana Department of Natural Resources, Division of Soil Conservation: 3.75-1 through 3.75-2.

Ingold, T.S.; Thomson, J.C. 1986. **Results of current research of synthetic and natural fiber erosion control systems.** In: Proceedings, erosion control: protecting our future: conference XVII. Denver, CO: International Erosion Control Association: 63-75.

Iowa State University. 2009. **Iowa stormwater management manual: ver. 3.** Ames, IA: Iowa State University, Institute for Transportation. http://www.intrans.iastate/edu/publications/posts/?publications=stormwater-management-manual (accessed July 10, 2012).

Irwin, L.H.; Taylor, D.J.; Aneshansley, D.J. 1986. **Device to measure road dustiness on aggregate surfaced roads.** Cornell Local Roads Program Report 86-5. Ithaca, NY: Cornell University, Department of Agricultural Engineering. 101 p.

Island County Public Works. 2003. **Island County best management practices field manual for small construction sites.** Coupeville, WA: Island County Public Works. 51 p.

Jacobson, T. 1992. **Low traffic roads stabilised with bitumen binders.** Nordic Road and Transport Research. 1: 22-23.

Jańczuk, B.; Bialopiotrowicz, T.; Kliszcz, A.; Biliński, B.; Stawiński, J. 1991. **Influence of polyacrylamide on the surface free energy and wettability of a chernozem soil.** Geoderma. 50: 173-184.

Janzen, D.C. 1990. **An industry response to compaction concerns in agriculture.** In: Proceedings, international summer meeting of the ASAE. Paper No. 90-1074. St. Joseph, MI: American Society of Agricultural Engineers. 15 p.

Jennings, G.D.; Jarrett, A.R. 1985. **Laboratory evaluation of mulches in reducing erosion.** Transactions of the American Society of Agricultural Engineers. 28(5): 1466-1470.

Johansen, D.K.; Copstead, R.; Moll, J. 1997. **Relief culverts.** 9777 1812-SDTDC. San Dimas, CA: U.S. Department of Agriculture, Forest Service, San Dimas Technology and Development Center. 7 p.

Johnson, B.S.; Erickson, A.E.; Smucker, A.J.M. 1986. **Alleviation of compaction on a fine-textured soil.** Paper No. 86-1517. St. Joseph, MI: American Society of Agricultural Engineers. 10 p.

Jones, D. 1999. **Holistic approach to research into dust and dust control on unsealed roads.** Transportation Research Record. 1652(2): 3-9.

Jones, D. 2000a. **Road dust – just a nuisance or a significant road management issue?** In: Proceedings, South African transport conference. Pretoria, South Africa; Washington, DC: Transportation Research Board: 12 p.

Jones, J.A. 2000b. **Hydrologic processes and peak discharge response to forest removal, regrowth, and roads in 10 small experimental basins, western Cascades, Oregon.** Water Resources Research. 36(9): 2621-2642.

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

Jordán-López, A.; Martínez-Zavala, L.; Bellinfante, N. 2009. **Impact of different parts of unpaved forest roads on runoff and sediment yield in a Mediterranean area.** Science of the Total Environment. 407: 937-944.

Kadlec, R.H.; Knight, R.L. 1996. **Treatment wetlands.** Boca Raton, FL: Lewis Publishers. 928 p.

Kahklen, K.; Hartsog, W. 1999. **Results of road erosion studies on the Tongass National Forest.** Unpublished report on file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Juneau, AK. 47 p.

Kaighn, R.J., Jr.; Yu, S.L. 1996. **Testing of roadside vegetation for highway runoff pollutant removal.** Transportation Research Record. 1523: 116-123.

Kapfer, J.M.; Paloski, R.A. 2011. **On the threat to snakes of mesh deployed for erosion control and wildlife exclusion.** Herpetological Conservation and Biology. 6(1): 1-9.

Kaushal, S.S.; Groffman, P.M.; Likens, G.E.; Belt, K.T.; Stack, W.P.; Kelly, V.R.; Band, L.E.; Fisher, G.T. 2005. **Increased salinization of fresh water in the northeastern United States.** Proceedings of the National Academy of Sciences. 102(38): 13517-13520.

Kazman, Z.; Shainberg, I.; Gal, M. 1983. **Effect of low levels of exchangeable sodium and applied phosphogypsum on the infiltration rate of various soils.** Soil Science. 135(3): 184-192.

Keener, H.M.; Faucette, B.; Klingman, M.H. 2007. **Flow-through rates and evaluation of solids separation of compost filter socks versus silt fence in sediment control applications.** Journal of Environmental Quality. 36: 742-752.

Keliher, K.P.; Taylor, S.E.; Ritter, M.A. 1995. **Performance of portable bridges for skidder traffic.** In: Proceedings, 18th annual meeting of the Council on Forest Engineering. Corvallis, OR: Council on Forest Engineering: 37-46.

Keller, G.; Sherar, J. 2003. **Low-volume roads engineering: best management practices field guide.** Produced for the U.S. Agency for International Development in cooperation with USDA, Forest Service, International Programs and Conservation Management Institute, Virginia Polytechnic Institute and State University. 158 p. Available at http://ntl.bts.gov/lib/24000/24600/24650/Index_BMP_Field_Guide.htm (accessed May 31, 2016).

Kelsey, K.; Johnson, T.; Vavra, R. 2006. **Needed information: testing, analyses, and performance values for slope interruption and perimeter control BMPs.** In: Proceedings, International Erosion Control Association conference technical session. Denver, CO: International Erosion Control Association: 171-181.

Kennedy, R. 1997. **Road maintenance frequency vs. sediment production.** Engineering Field Notes. 29(Sept.-Dec.): 11-15. http://www.fs.fed.us/t-d/pubs/pdfpubs/pdf97713815/pdf97713815.pdf (accessed June 29, 2016).

Kercher, W.C., Jr.; Landon, J.C.; Massarelli, R. 1983. **Grassy swales prove cost-effective for water pollution control.** Public Works. April: 53-54.

Kestler, M.A.; Berg, R.L.; Nam, S.; Smith, C.E. 2007. **Reduced tire pressure: review of an alternate road usage technique for reducing springtime damage on low volume roads.** In: Proceedings, 11th international conference on cold regions engineering. Reston, VA: American Society of Civil Engineers: 382-393.

Kestler, M.A.; Nam, S. 1999. **Reducing damage to low volume asphalt-surfaced roads, and improving local economies: update on variable tire pressure project.** In: Zufel, J.E., ed. Cold regions engineering: putting research into practice – Proceedings, tenth international conference. Reston, VA: American Society of Civil Engineers: 461-471.

Ketcheson, G.L.; Megahan, W.F. 1996. **Sediment production and downslope sediment transport from forest roads in granitic watersheds.** Res. Pap. INT-RP-486. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 11 p.

Kidd, W.J., Jr. 1963. **Soil erosion control structures on skidtrails.** Res. Pap. INT-1. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 8 p.

Kill, D.L.; Foote, L.E. 1971. **Comparisons of long and short-fibered mulches.** Transactions of the American Society of Agricultural Engineers. 14(5): 942-944.

Kimball, C.E. 1997. **Evaluating groundwater pollution susceptibility of dust suppressants and roadbed stabilizers: case study of a petroleum-based product.** Paper No. 970815. Transportation Research Record. 1589: 64-69.

King, J.G. 1979. **Fill slope erosion from forest roads.** In: Proceedings, 34th annual meeting, Pacific Northwest Region. PNW 79-404. St. Joseph, MI: American Society of Agricultural Engineers. 11 p.

King, J.G. 1984. **Ongoing studies in Horse Creek on water quality and water yield.** Tech. Bull. 435. New York: National Council of the Paper Industry for Air and Stream Improvement: 28-35.

King, S.; McLaughlin, R.A. 2007. **Improved erosion control BMPs for road construction projects in the North Carolina mountains.** Sediments. 14(3): 1-2.

Kingsbury, P.A.; Hastie, W.J.; Harrington, A.J. 1991. **Regional landslip hazard assessment using a geographic information system.** In: Proceedings, sixth international symposium on landslides. Rotterdam, the Netherlands: Balkema Publishers: 995-999.

Kirchner, H.; Gall, J.A. 1991. **Liquid calcium chloride for dust control and base stabilization of unpaved road systems.** Transportation Research Record. 1291: 173-178.

Kirchner, H.W. 1988. **Road dust suppressants compared.** Public Works. 119(13): 27-28.

Knapp, P.A. 1992. **Soil loosening processes following the abandonment of two arid western Nevada townsites.** Great Basin Naturalist. 52(2): 149-154.

Koch, S.; Ksaibati, K. 2010. **Performance of recycled asphalt pavement in gravel roads.** Laramie, WY: University of Wyoming, Department of Civil and Architectural Engineering. 285 p.

Kochenderfer, J.N. 1970. **Erosion control on logging roads in the Appalachians.** Res. Pap. NE-158. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 28 p.

Kochenderfer, J.N. 1995. **Using open-top pipe culverts to control surface water on steep road grades.** Gen. Tech. Rep. NE-194. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 7 p.

Kochenderfer, J.N.; Edwards, P.J.; Wood, F. 1997. **Hydrologic impacts of logging an Appalachian watershed using West Virginia's best management practices.** Northern Journal of Applied Forestry. 14(4): 207-218.

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

Kochenderfer, J.N.; Hornbeck, J.W. 1999. **Contrasting timber harvesting operations illustrate the value of BMPs.** In: Stringer, J.W.; Loftis, D.L., eds. Proceedings, 12th central hardwood forest conference. Gen. Tech. Rep. SRS-24. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 128-136.

Kochenderfer, J.N.; Wendel, G.W.; Smith, H.C. 1984. **Cost of and soil loss on "minimum-standard" forest truck roads constructed in the central Appalachians.** Res. Pap. NE-544. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 8 p.

Kohnke, H.; Werkhoven, C.H. 1963. **Soil temperature and soil freezing as affected by an organic mulch.** Soil Science Society of America Journal. 27(1): 13-17.

Kouwen, N. 1990. **Silt fences to control sediment movement on construction sites.** MAT-90-03. Downsview, ON: Ontario Ministry of Transportation. 67 p.

Kouwen, N.; Li, R.; Simons, D.B. 1981. **Flow resistance in vegetated waterways.** Transactions of the American Society of Agricultural Engineers. 24(3): 684-690, 698.

Kouwen, N.; Unny, T.E. 1973. **Flexible roughness in open channels.** Journal of the Hydraulics Division. 99(5): 713-728.

Kozlowski, T.T. 1984. **Plant responses to flooding of soil.** BioScience. 34(3): 162-167.

Kramer, B.W. 2001. **Forest road contracting, construction, and maintenance for small forest woodland owners.** Research Contribution 35. Corvallis, OR: Oregon State University, College of Forestry, Forest Research Laboratory. 79 p.

Kramer, L.A.; Meyer, L.D. 1969. **Small amounts of surface mulch reduce soil erosion and runoff velocity.** Transactions of the American Society of Agricultural Engineers.12: 638-641, 645.

Krenitsky, E.C.; Carroll, M.J. 1994. **Use of erosion control materials to establish turf.** In: Sustaining environmental quality: the erosion control challenge – proceedings of conference XXV. Denver, CO: International Erosion Control Association: 79-90.

Kreutzweiser, D.P.; Capell, S.S. 2001. **Fine sediment deposition in streams after selective forest harvesting without riparian buffers.** Canadian Journal of Forest Research. 31: 2134-2142.

La Marche, J.L.; Lettenmaier, D.P. 2001. **Effects of forest roads on flood flows in the Deschutes River, Washington.** Earth Surface Processes and Landforms. 26: 115-134.

Laird, D.A. 1997. **Bonding between polyacrylamide and clay mineral surfaces.** Soil Science. 162(11): 826-832.

Landphair, H.C.; McFalls, J.A.; Peterson, B.E.; Li, M. 1997. **Alternatives to silt fence for temporary sediment control at highway construction sites: guidelines for TxDOT.** Report No. FHWA/TX-98/1737-S. Prepared for: Texas Department of Transportation, Research and Technology Transfer Office. College Station, TX: Texas A&M University System, Texas Transportation Institute. 104 p.

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

Lang, R.D. 1979. **The effect of ground cover on surface runoff from experimental plots.** Journal of Soil Conservation. 35: 108-114.

Langdon, B. 1980. **An evaluation of dust abatement materials used in Region 6.** Final report. Prepared for: U.S. Forest Service, Region 6. Corvallis, OR: Oregon State University, Transportation Research Institute. 109 p.

Langdon, B.; Williamson, R.K. 1983. **Dust-abatement materials: evaluation and selection.** Transportation Research Record. 898: 250-257.

Larsen, M.C.; Parks, J.E. 1997. **How wide is a road? The association of roads and mass-wasting in a forested montane environment.** Earth Surface Processes and Landforms. 22: 835-848.

Larsen, M.C.; Torres-Sánchez, A.J. 1992. **Landslides triggered by hurricane Hugo in eastern Puerto Rico, September 1989.** Caribbean Journal of Science. 28(3-4): 113-125.

Latimer, J.S.; Hoffman, E.J.; Hoffman, G.; Fasching, J.L.; Quinn, J.G. 1990. **Sources of petroleum hydrocarbons in urban runoff.** Water, Air, and Soil Pollution. 52: 1-21.

Lattanzi, A.R.; Meyer, L.D.; Baumgardner, M.F. 1974. **Influences of mulch rate and slope steepness on interrill erosion.** Soil Science Society of America Proceedings. 38(6): 946-950.

Ledingham, G.A.; Adams, G.A. 1942. **Biological decomposition of chemical lignin. II. Studies of the decomposition of calcium lignosulphonate by wood destroying and soil fungi.** Canadian Journal of Research. 20c(1): 13-27.

Lee, K.; Isenhart, T.M.; Schultz, R.C. 2003. **Sediment and nutrient removal in an established multi-species riparian buffer.** Journal of Soil and Water Conservation. 58: 1-8.

Lee, K.; Isenhart, T.M.; Schultz, R.C.; Mickelson, S.K. 1999. **Nutrient and sediment removal by switchgrass and cool-season grass filter strips in central Iowa, USA.** Agroforestry Systems. 44(2): 121-132.

Lee, K.; Isenhart, T.M.; Schultz, R.C.; Mickelson, S.K. 2000. **Multispecies riparian buffers trap sediment and nutrients during rainfall simulations.** Journal of Environmental Quality. 29: 1200-1205.

Lee, P.; Smyth, C.; Boutin, S. 2004. **Quantitative review of riparian buffer width guidelines from Canada and the United States.** Journal of Environmental Management. 70: 165-180.

Leib, B.G.; Redulla, C.A.; Stevens, R.G.; Matthews, G.R.; Strausz, D.A. 2005. **Erosion control practices integrated with polyacrylamide to reduce sediment loss in furrow irrigation.** Applied Engineering in Agriculture. 21(4): 595-603.

Lekha, K.R. 2004. **Field instrumentation and monitoring of soil erosion in coir geotextile stabilised slopes – a case study.** Geotextiles and Geomembranes. 22(5): 399-413.

Lemly, A.D. 1982. **Erosion control at construction sites on red clay soils.** Environmental Management. 6(4): 343-352.

Lentz, R.D.; Shainberg, I.; Sojka, R.E.; Carter, D.L. 1992. **Preventing irrigation furrow erosion with small applications of polymers.** Soil Science Society of America Journal. 56(6): 1926-1932.

Lentz, R.D.; Sojka, R.E. 1994. **Field results using polyacrylamide to manage furrow erosion and infiltration.** Soil Science. 158(4): 274-282.

Lentz, R.D.; Sojka, R.E.; Carter, D.L. 1993. **Influence of polymer charge type and density on polyacrylamide ameliorated irrigated furrow erosion.** In: Proceedings, 24th annual International Erosion Control Association conference. Denver, CO: International Erosion Control Association: 161-168.

Lentz, R.D.; Stieber, T.D.; Sojka, R.E. 1995. **Applying polyacrylamide (PAM) to reduce erosion and increase infiltration under furrow irrigation.** In: Proceedings, Winter Commodity Schools. Moscow, ID: University of Idaho Cooperative Extension System: 79-92.

Lepore, B.J.; Thompson, A.M.; Petersen, A. 2009. **Impact of polyacrylamide delivery method with lime or gypsum for soil and nutrient stabilization.** Journal of Soil and Water Conservation. 64(3): 223-231.

Levy, G.J.; Agassi, M. 1995. **Polymer molecular weight and degree of drying effects on infiltration and erosion of three different soils.** Australian Journal of Soil Research. 33: 1007-1018.

Levy, G.J.; Ben-Hur, M.; Agassi, M. 1991. **The effect of polyacrylamide on runoff, erosion, and cotton yield from fields irrigated with moving sprinkler systems.** Irrigation Science. 12: 55-60.

Levy, G.J.; Levin, J.; Gal, M.; Ben-Hur, M.; Shainberg, I. 1992. **Polymers' effects on infiltration and soil erosion during consecutive simulated sprinkler irrigations.** Soil Science Society of America Journal. 56: 902-907.

Lewis, J.; Mori, S.R.; Keppeler, E.T.; Ziemer, R.R. 2001. **Impacts of logging on storm peak flows, flow volumes and suspended sediment loads in Caspar Creek, California.** In: Wigmosta, M.S.; Burges, S.J., eds. Land use and watersheds: human influence on hydrology and geomorphology in urban and forest areas. Water Science and Application, vol. 2. Washington, DC: American Geophysical Union: 85-125.

Lewis, L. 1995. **Olympic National Forest partnerships for slope repair and erosion control.** Restoration and Management Notes. 13(1): 37-39.

Leytem, A.B.; Bjorneberg, D.L. 2005. **Removing soluble phosphorus in irrigation return flows with alum additions.** Journal of Soil and Water Conservation. 60(4): 200-208.

Line, D.E. 1991. **Sediment trapping effectiveness of grass strips.** In: Proceedings, fifth federal interagency conference on sedimentation. Washington, DC: Federal Energy Regulatory Commission. 2: PS56-PS63.

Line, D.E.; White, N.M. 2001. **Efficiencies of temporary sediment traps on two North Carolina construction sites.** Transactions of the American Society of Agricultural Engineers. 44(5): 1207-1215.

Liu, X.; Zhang, X.; Zhang, M. 2008. **Major factors influencing the efficacy of vegetated buffers on sediment trapping: a review and analysis.** Journal of Environmental Quality. 37: 1667-1674.

Loch, R.J. 2000. **Effects of vegetation cover on runoff and erosion under simulated rain and overland flow on a rehabilitated site on the Meandu Mine, Tarong, Queensland.** Australian Journal of Soil Research. 38(2): 299-312.

Loch, R.J.; Espigares, T.; Costantini, A.; Garthe, R.; Bubb, K. 1999. **Vegetative filter strips to control sediment movement in forest plantations: validation of a simple model using field data.** Australian Journal of Soil Research. 37(5): 929-946.

Lohnes, R.A.; Coree, B.J. 2002. **Determination and evaluation of alternate methods for managing and controlling highway-related dust.** Final Report TR449. Prepared for: Iowa Department of Transportation, Iowa Highway Research Board. Ames, IA: Iowa State University, Department of Civil and Construction Engineering. 26 p.

Looney, T.E. 1981. **A comparison of the dam bridge with current methods of crossing streams with skidders.** Blacksburg, VA: Virginia Polytechnic Institute and State University. 58 p. M.S. thesis.

Loreto, D.; Batista, J.; Piechota, T.; Singh, V. 2002. **Impacts of dust suppressant application on the quality of urban runoff from disturbed lands.** Faculty Publications, Paper 37. Las Vegas, NV: University of Nevada, Department of Civil and Environmental Engineering. 11 p.

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

Luce, C.H.; Black, T.A. 2001. **Effects of traffic and ditch maintenance on forest road sediment production.** In: Proceedings, seventh federal interagency sedimentation conference. [N.p.]: Advisory Committee on Water Information, Subcommittee on Sedimentation: V67-V74.

Lunsford, G.B.; Mahoney, J.P. 2001. **Dust control on low volume roads: a review of techniques and chemicals used.** Technical Study Report No. FHWALT-01-002. Prepared for: U.S. Department of Transportation, Federal Highway Administration, Office of Professional Development. Seattle, WA: University of Washington. 59 p.

Lynch, J.A.; Corbett, E.S. 1990. **Evaluation of best management practices for controlling nonpoint pollution from silvicultural operations.** Water Resources Bulletin. 26(1): 41-52.

Lynch, J.A.; Corbett, E.S.; Mussallem, K. 1985. **Best management practices for controlling nonpoint-source pollution on forested watersheds.** Journal of Soil and Water Conservation. 40(1): 164-167.

Lyons, J.; Trimble, S.W.; Paine, L.K. 2000. **Grass versus trees: managing riparian areas to benefit streams of central North America.** Journal of the American Water Resources Association. 36(4): 919-930.

MacDonald, L.H.; Anderson, D.M.; Dietrich, W.E. 1997. **Paradise threatened: land use and erosion on St. John, US Virgin Islands.** Environmental Management. 21(6): 851-863.

MacDonald, L.H.; Robichaud, P.R. 2007. **JFSP final report: postfire erosion and the effectiveness of emergency rehabilitation treatments over time.** Project 03-2-3-22. Boise, ID: Joint Fire Science Program. 17 p.

MacDonald, L.H.; Sampson, R.W.; Anderson, D.M. 2001. **Runoff and road erosion at the plot and road segment scales, St. John, US Virgin Islands.** Earth Surface Processes and Landforms. 26: 251-272.

Magette, W.L.; Brinsfield, R.B.; Palmer, R.E.; Wood, J.D. 1989. **Nutrient and sediment removal by vegetated filter strips.** Transactions of the American Society of Agricultural Engineers. 32(2): 663-667.

Maharaj, R.J. 1993. **Landslide processes and landslide susceptibility analysis from an upland watershed: a case study from St. Andrew, Jamaica, West Indies.** Engineering Geology. 34: 53-79.

Malik, M.; Amrhein, C.; Letey, J. 1991. **Polyacrylamide to improve water flow and salt removal in a high shrink-swell soil.** Soil Science Society of America Journal. 55: 1664-1667.

Mannering, J.V.; Meyer, L.D. 1963. **The effects of various rates of surface mulch on infiltration and erosion.** Soil Science Society of America Proceedings. 27: 84-86.

Marshall, E.; Randhir, T. 2008. **Effect of climate change on watershed system: a regional analysis.** Climatic Change. 89: 263-280.

Martin, J.S. 1985. **Use of silt fences for control of sediment run-off.** In: Erosion control: a challenge in our time. Proceedings, International Erosion Control Association conference XVI. Pinole, CA: International Erosion Control Association: 79-84.

Mason, L. 1990. **Portable wetland area and stream crossings.** Publ. 9024-1203. San Dimas, CA: U.S. Department of Agriculture, Forest Service, Technology and Development Center. 110 p.

Mason, L.E.; Greenfield, P.H. 1995. **Portable crossings for weak soil areas and streams.** Transportation Research Record. 1504: 118-124.

Mazer, G.; Booth, D.; Ewing, K. 2001. **Limitations to vegetation establishment and growth in biofiltration swales.** Ecological Engineering. 17: 429-433.

McClelland, D.E.; Foltz, R.B.; Falter, C.M.; Wilson, W.D.; Cundy, T.; Schuster, R.L.; Saurbier, J.; Rabe, C.; Heinemann, R. 1997. **Relative effects on a low-volume road system of landslides resulting from episodic storms in northern Idaho.** Transportation Research Record. 1652: 235-243.

McHattie, R.L. 1994. **Elliott Highway dust control and stabilization with EMC2 and montmorillonite. Final evaluation of experimental features; Projects AK8701A and AK8701B.** Juneau, AK: Alaska Department of Transportation and Public Facilities. 6 p.

McKee, W.H., Jr.; Blaser, R.E.; Barkley, D.G. 1964. **Mulches for steep cut slopes.** Highway Research Record. 53: 35-42.

McLaughlin, R.A. 2003. **The potential for substantial improvements in sediment and turbidity control.** In: Saleh, A., ed. Proceedings, total maximum daily load (TMDL) environmental regulations-II. ASAE Publ. No. 701P1503. St. Joseph, MI: American Society of Agricultural Engineers: 262-272.

McLaughlin, R.A.; Hayes, S.A.; Clinton, D.L.; McCaleb, M.S.; Jennings, G.D. 2009a. **Water quality improvements using modified sediment control systems on construction sites.** Transactions of the American Society of Agricultural and Biological Engineers. 52(6): 1859-1867.

McLaughlin, R.A.; King, S.E.; Jennings, G.D. 2009b. **Improving construction site runoff quality with fiber check dams and polyacrylamide.** Journal of Soil and Water Conservation. 64(2): 144-154.

McLeod, H.E.; Reed, I.F.; Johnson, W.H.; Gill, W.R. 1966. **Draft, power efficiency, and soil-compaction characteristics of single, dual, and low-pressure tires.** Transactions of the American Society of Agricultural Engineers. 9(1): 41-44.

Megahan, W.F. 1974a. **Deep-rooted plants for erosion control on granitic road fills in the Idaho Batholith.** Res. Pap. INT-161. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 18 p.

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

Megahan, W.F.; Day, N.F.; Bliss, T.M. 1978. **Landslide occurrence in the western and central northern Rocky Mountain physiographic province in Idaho.** In: Youngberg, C.T., ed. Forest soils and land use: proceedings, fifth North American forest soils conference. Fort Collins, CO: Colorado State University: 116-139.

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

Megahan, W.F.; Kidd, W.J. 1972. **Effect of logging roads on sediment production rates in the Idaho Batholith.** Res. Pap. INT-123. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 14 p.

Megahan, W.F.; King, J.G. 2004. **Erosion, sedimentation, and cumulative effects in the northern Rocky Mountains.** In: Ice, G.G.; Stednick, J.D., eds. A century of forest and wildland watershed lessons. Bethesda, MD: Society of American Foresters: 201-222. Chapter 9.

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

Mehrotra, G.S.; Sarkar, S.; Dharmaraju, R. 1991. **Landslide hazard assessment in Rishikesh-Tehri area, Garhwal Himalaya, India.** In: Proceedings, sixth international symposium on landslides. Rotterdam, the Netherlands: Balkema Publishers: 1001-1007.

Merrill, M.A. 2005. **The effects of culverts and bridges on stream geomorphology.** Raleigh, NC: North Carolina State University. 153 p. M.S. thesis.

Meyer, L.D.; Dabney, S.M.; Harmon, W.C. 1995. **Sediment-trapping effectiveness of stiff-grass hedges.** Transactions of the American Society of Agricultural Engineers. 38(3): 809-815.

Meyer, L.D.; Johnson, C.B.; Foster, G.R. 1972. **Stone and woodchip mulches for erosion control on construction sites.** Journal of Soil and Water Conservation. 27(6): 264-269.

Meyer, L.D.; Mannering, J.V. 1963. **Crop residues as surface mulches for controlling erosion on sloping land under intensive cropping.** Transactions of the American Society of Agricultural Engineers. 6(4): 322-323, 327.

Meyer, L.D.; Wischmeier, W.H.; Foster, G.R. 1970. **Mulch rates required for erosion control on steep slopes.** Soil Science Society of America Proceedings. 34(6): 928-931.

Michaels, A.S.; Morelos, O. 1955. **Polyelectrolyte adsorption by kaolinite.** Industrial and Engineering Chemistry. 47(9): 1801-1809.

Milauskas, S.J. 1988. **Low-water stream crossing options for southern haul roads.** Southern Journal of Applied Forestry. 12: 11-15.

Miles, S.R.; Haskins, D.M.; Ranken, D.W. 1989. **Emergency burn rehabilitation: cost, risk, and effectiveness.** In: Berg, N.H., ed. Proceedings, symposium on fire and watershed management. Gen. Tech. Rep. PSW-109. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 97-102.

Miller, R.L., Jr.; Dewalle, D.R.; Brooks, R.P.; Finley, J.C. 1997. **Long-term impacts of forest road crossings of wetlands in Pennsylvania.** Northern Journal of Applied Forestry. 14(3): 109-116.

Miller, S.M.; Joaquin, A.C. 2011. **Geosynthetic stabilized earth using compost filter socks for sustainable vegetated walls and slopes.** In: Proceedings, 42nd conference and expo of the International Erosion Control Association. Denver, CO: International Erosion Control Association. 12 p.

Miller, W.P. 1987. **Infiltration and soil loss of three gypsum-amended ultisols under simulated rainfall.** Soil Science Society of America Journal. 51: 1314-1320.

Mills, K. 2006. **Environmental performance measures for forest roads.** In: Proceedings, 29th Council on Forest Engineering conference: working globally – sharing forest engineering challenges and technologies around the world. Morgantown, WV: Council on Forest Engineering: 291-300.

Mitchell, A.R. 1986. **Polyacrylamide application in irrigation water to increase infiltration.** Soil Science. 141(5): 353-358.

Mitchell, D.J.; Barton, A.P.; Fullen, M.A.; Hocking, T.J.; Zhi, W.B.; Yi, Z. 2003. **Field studies of the effects of jute geotextiles on runoff and erosion in Shropshire, UK.** Soil Use and Management. 19: 182-184.

Mitchell, W.C.; Trimble, G.R., Jr. 1959. **How much land is needed for the logging transport system?** Journal of Forestry. 57(1): 10-12.

Mockler, S.; Croke, J. 1999. **Prescriptive measures for the prevention of road to stream linkage.** In: Rutherford, I.; Bartley, R., eds. Proceedings, second Australian stream management conference. Canberra, Australia: Cooperative Research Centre for Catchment Hydrology: 2: 451-456.

Moll, J.; Copstead, R.; Johansen, D.K. 1997. **Traveled way surface shape.** 9777 1808-SDTDC. San Dimas, CA: U.S. Department of Agriculture, Forest Service, San Dimas Technology and Development Center. 10 p.

Monlux, S. 1993. **Dust abatement product comparisons in the Northern Region.** Engineering Field Notes. 26(May-June): 26-36. http://www.fs.fed.us/t-d/pubs/pdfpubs/pdf93713815/pdf93713815.pdf (accessed June 29, 2016).

Monlux, S. 2003. **Stabilizing unpaved roads with calcium chloride.** Paper No. LVR8-1110. Transportation Research Record. 1819: 52-56.

Monlux, S.; Mitchell, M.R. 2006. **Surface-aggregate stabilization with chloride materials.** 0677 1805-SDTDC. San Dimas, CA: U.S. Department of Agriculture, Forest Service, National Technology and Development Program. 23 p.

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

Montgomery, D.R.; Dietrich, W.E. 1994. **A physically based model for the topographic control on shallow landsliding.** Water Resources Research. 30(4): 1153-1171.

Moore, K.; Furniss, M.; Firor, S.; Love, M. 1999. **Fish passage through culverts: an annotated bibliography.** Eureka, CA: U.S. Department of Agriculture, Forest Service, Six Rivers National Forest Watershed Interactions Team. http://stream.fs.fed.us/fishxing/biblio.pdf (accessed December 4, 2014).

Moore, R.; Dibb, T.M.; Billing, D.W. 1991. **The distribution and causes of mass movement in Aurora Province, Philippines.** In: Proceedings, sixth international symposium on landslides. Rotterdam, the Netherlands: Balkema Publishers: 1023-1029.

Moore, R.D.; Wondzell, S.M. 2005. **Physical hydrology and the effects of forest harvesting in the Pacific Northwest: a review.** Journal of the American Water Resources Association. 41(4): 763-784.

Moreno-de las Heras, M.; Merino-Martín, L.; Nicolau, J.M. 2009. **Effect of vegetation cover on the hydrology of reclaimed mining soils under Mediterranean-Continental climate.** Catena. 77: 39-47.

Morgan, R.J.; Schaefer, V.R.; Sharma, R.S. 2005. **Determination and evaluation of alternative methods for managing and controlling highway-related dust: Phase II – demonstration project.** IHRB Project TR-506. Iowa Highway Research Board. Ames, IA: Iowa State University: Department of Civil, Construction, and Environmental Engineering. 107 p.

Morgan, R.P.C.; Rickson, R.J. 1995. **Slope stabilization and erosion control: a bioengineering approach.** London, England: Chapman and Hall. 274 p.

Moss, A.J. 1991. **Rain-impact soil crust I: Formation on a granite-derived soil.** Australian Journal of Soil Research. 29: 271-289.

Municipality of Metropolitan Seattle. 1992. **Biofiltration swale performance, recommendations, and design considerations.** Publication 657. Seattle, WA: Municipality of Metropolitan Seattle, Water Pollution Control Department.

Muñoz-Carpena, R.; Parsons, J.E. 2012. **VFSMOD-W: Vegetative Filter Strip Modeling System.** Gainesville, FL: University of Florida, Department of Agriculture and Biological Engineering. http://abe.ufl.edu/carpena/vfsmod/ (accessed December 2, 2014).

Murray, H.H. 2000. **Traditional and new applications for kaolin, smectite, and palygorskite: a general overview.** Applied Clay Science. 17: 207-221.

Nadler, A.; Malik, M.; Letey, J. 1992. **Desorption of polyacrylamide and polysaccharide polymers from soil materials.** Soil Technology. 5: 91-95.

Natural Resources Conservation Service. 2003. **Grassed waterway and vegetated filter.** Conservation Practice Job Sheet 412. Washington, DC: U.S. Department of Agriculture, Natural Resources Conservation Service. Available at http://www.mn.nrcs.usda.gov/technica/ecs/agron/crp/plan%20forms/cp8a%20plan%204-03.pdf (accessed July 9, 2012).

Neibling, W.H.; Alberts, E.E. 1979. **Composition and yield of soil particles transported through sod strips.** Paper No. 79–2065. St. Joseph, MI: American Society of Agricultural Engineers. 13 p.

Ngabe, B.; Bidleman, T.F.; Scott, G.I. 2000. **Polycyclic aromatic hydrocarbons in storm runoff from urban and coastal South Carolina.** The Science of the Total Environment. 255: 1-9.

Nicholson, K.W.; Branson, J.R.; Giess, P.; Cannell, R.J. 1989. **The effects of vehicle activity on particle resuspension.** Journal of Aerosol Science. 20(8): 1425-1428.

Norman, J.R. 2006. **Effects of road–stream crossings on stream geomorphology and the movement of small bodied fishes in the Etowah River Basin, USA.** Athens, GA: University of Georgia. 154 p. M.S. thesis.

Norman, M.; Johansson, C. 2006. **Studies of some measures to reduce road dust emissions from paved roads in Scandinavia.** Atmospheric Environment. 40: 6154-6164.

Normann, J.M.; Houghtalen, R.J.; Johnston, W.J. 1985. **Hydraulic design of highway culverts.** Report No. FHWA-IP-85-15. McLean, VA: U.S. Department of Transportation, Federal Highway Administration, Office of Research, Development and Technology. 272 p.

Norris, V. 1993. **The use of buffer zones to protect water quality: a review.** Water Resources Management. 7(4): 257-272.

Norton, L.D. 1992. **Chemical amendments for erosion control.** Paper No. 92-2562. St. Joseph, MI: American Society of Agricultural Engineers. 9 p.

Norton, L.D.; Shainberg, I.; King, K.W. 1993. **Utilization of gypsiferous amendments to reduce surface sealing in some humid soils of the eastern USA.** Catena Supplement. 24: 77-92.

Novotny, V.; Olem, H. 1994. **Water quality: prevention, identification, and management of diffuse pollution.** New York: Van Nostrand Reinhold. 1054 p.

Oakland, P.H. 1983. **An evaluation of urban storm water pollutant removal through grassed swale treatment.** In: De Vore, R.W., ed. Proceedings, 1983 international symposium on urban hydrology, hydraulics, and sediment control. New York: OES Publications: 183-185.

Odemerho, F.O. 1986. **Variation in erosion-slope relationship on cut-slopes along a tropical highway.** Singapore Journal of Tropical Geography. 7(2): 98-107.

Ohlander, C.A. 1976. **Defining the sediment trapping characteristics of a vegetative buffer. Special case: road erosion.** In: Proceedings, third federal interagency sedimentation conference. Paper PB-245 100. Washington, DC: Sedimentation Committee, Water Resources Council: 77-82.

Ollesch, G.; Vacca, A. 2002. **Influence of time on measurement results of erosion plot studies.** Soil and Tillage Research. 67: 23-39.

O'Loughlin, C. 1975. **The effects of forest removal on the slope stability and hydrology of steep catchments.** In: Proceedings, Water and Soil Division Course No. 2. Rotorua, New Zealand: National Water and Soil Conservation Organisation. 8 p.

Orr, H.K. 1970. **Runoff and erosion control by seeded and native vegetation on a forest burn: Black Hills, South Dakota.** Res. Pap. RM-60. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 12 p.

Orts, W.J.; Roa-Espinosa, A.; Sojka, R.E.; Glenn, G.M.; Imam, S.H.; Erlacher, K.; Pedersen, J.S. 2007. **Use of synthetic polymers and biopolymers for soil stabilization in agricultural, construction, and military applications.** Journal of Materials in Civil Engineering. 19: 58-66.

Ossege, A.L. 1993. **Why percent open area?** 4 p. http://carthagemills.com/pdf/Why_Percent-Open-Area%20in%20Geotextiles.pdf (accessed June 29, 2016).

Owino, J.; Gretzmacher, R. 2002. **Performance of narrow strips of vetiver grass (*Vetiveria zizanioides*) and napier grass (*Pennisetum purpureum*) as barriers against runoff and soil loss on a clay loam soil in Kenya.** In: Deininger, A., comp. ed. Challenges to organic farming and sustainable land use in the tropics and subtropics. Deutscher Tropentag 2002. Kassel, Germany: Kassel University Press. 8 p.

Packer, P.E. 1951. **An approach to watershed protection criteria.** Journal of Forestry. 49(9): 639-644.

Packer, P.E. 1967. **Criteria for designing and locating roads to control sediment.** Forest Science. 13: 2-18.

Packer, P.E.; Christensen, G.F. 1964. **Guides for controlling sediment from secondary logging roads.** Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station; Missoula, MT: U.S. Department of Agriculture, Forest Service, Northern Region. 42 p.

Pan, C.; Ma, L.; Shangguan, Z. 2010. **Effectiveness of grass strips in trapping suspended sediments from runoff.** Earth Surface Processes and Landforms. 35: 1006-1013.

Pandila, M.M. 1973. **Micro-organisms associated with microbiological degradation of lignosulphonates: a review of literature.** Pulp and Paper Magazine of Canada. 74: 80-84.

Pankau, R.C. 2010. **Concentrated flow paths in riparian buffer zones of southern Illinois.** Carbondale, IL: Southern Illinois University. 102 p. M.S. thesis.

Pankau, R.C.; Schoonover, J.E.; Williard, K.W.J.; Edwards, P.J. 2012. **Concentrated flow paths in riparian buffer zones of southern Illinois.** Agroforestry Systems. 84: 191-205.

Pannkuk, C.D.; Robichaud, P.R. 2003. **Effectiveness of needle cast at reducing erosion after forest fires.** Water Resources Research. 39(12): 1333-1341.

Parizek, R.R. 1971. **Impact of highways on the hydrogeologic environment.** In: Coates, D.R., ed. Environmental geomorphology. Binghamton, NY: New York University: 151-199. Chapter 9.

Parsons, J.E.; Daniels, R.B.; Gilliam, J.W.; Dillaha, T.A. 1994. **Reduction in sediment and chemical load agricultural field runoff by vegetative filter strips.** UNC-WRRI-94-286. Washington, DC: U.S. Department of Agriculture and U.S. Environmental Protection Agency; Chapel Hill, NC: University of North Carolina, Water Resources Research Institute. 95 p.

Paterson, J.J.; Jones, J.H.; Olsen, F.J.; McCoy, G.C. 1980. **Dairy liquid waste distribution in an overland flow vegetative-soil filter system.** Transactions of the American Society of Agricultural Engineers. 23(4): 973-977, 984.

Pearce, R.A.; Trlica, M.J.; Leininger, W.C.; Mergen, D.E.; Frasier, G. 1998. **Sediment movement through riparian vegetation under simulated rainfall and overland flow.** Journal of Range Management. 51(3): 301-308.

Peirce, A.W. 1966. **Studies on salt tolerance of sheep: VI. The tolerance of wethers in pens for drinking waters of the types obtained from underground sources in Australia.** Australian Journal of Agricultural Research. 17: 209-218.

Penn State Center for Dirt and Gravel Road Studies. 2014. **Driving surface aggregate.** Informational Bulletin. University Park, PA: Pennsylvania State University. 4 p.

Peterjohn, W.T.; Correll, D.L. 1984. **Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest.** Ecology. 65(5): 1466-1475.

Peters, S.; Peters, G. 2009. **Wildland CPR issues new report on Forest Service road management.** Missoula, MT: Wildlands CPR. 2 p. Available at http://www.wildlandscpr.org/road-riporter/wildlands-cpr-issues-new-report-forest-service-road-management (accessed September 12, 2012).

Petersen, A.L.; Thompson, A.M.; Baxter, C.A.; Norman, J.M.; Roa-Espinosa, A. 2007. **A new polyacrylamide (PAM) formulation for reducing erosion and phosphorus loss in rainfed agriculture.** Transactions of the American Society of Agricultural and Biological Engineers. 50(6): 2091-2101.

Peterson, J.R.; Flanagan, D.C.; Tishmack, J.K. 2002. **PAM application method and electrolyte source effects on plot-scale runoff and erosion.** Transactions of the American Society of Agricultural Engineers. 45(6): 1859-1867.

Pethick, J.; Leggett, D.; Husain, L. 1990. **Boundary layers under salt marsh vegetation developed in tidal currents.** In: Thornes, J.B., ed. Vegetation and erosion: processes and environments. Hoboken, NJ: John Wiley & Sons Ltd.: 113-124. Chapter 9.

Phillips, J.D. 1989. **An evaluation of the factors determining the effectiveness of water quality buffer zones.** Journal of Hydrology. 107: 133-145.

Piechota, T.; Van Ee, J.; Batista, J.; Stave, K.; James, D., eds. 2002. **Potential environmental impacts of dust suppressants: "avoiding another Times Beach."** EPA/600/R-04/031. Washington, DC: U.S. Environmental Protection Agency; Las Vegas, NV: University of Nevada. 97 p.

Piehl, B.T.; Beschta, R.L.; Pyles, M.R. 1988. **Ditch-relief culverts and low-volume forest roads in the Oregon Coast Range.** Northwest Science. 62(3): 91-98.

Pinnick, R.G.; Fernandez, G.; Hinds, B.D.; Bruce, C.W.; Schaefer, R.W.; Pendelton, J.D. 1985. **Dust generated by vehicular traffic on unpaved roadways: size and infrared extinction characteristics.** Aerosol Science and Technology. 4(1): 99-121.

Pitt, R. 1979. **Demonstration of non point pollution abatement through improved street cleaning practices.** Report No. EPA 600/2-79-161. Cincinnati, OH: U.S. Environmental Protection Agency, Office of Research and Development. 288 p.

Pitt, R.E.; Amy, G. 1973. **Toxic materials analysis of street surface contaminants.** Environmental Protection Series Report No. EPA-R2-72-283. Washington, DC: U.S. Environmental Protection Agency, Office of Research and Development. 145 p.

Pitts, J. 1992. **Slope stability in Singapore.** In: Physical adjustments in a changing landscape, the Singapore story. Queenstown, Singapore: Singapore University Press: 259-300. Chapter 9.

Plamondon, A.P. 1982. **Augmentation de la concentration des sédiments en suspension suite à l'exploitation forestière et durée de l'effet [Increase in suspended sediments after logging, and duration of the effect].** Canadian Journal of Forest Research. 12: 883-892.

Poesen, J.; De Luna, E.; Franca, A.; Nachtergaele, J.; Govers, G. 1999. **Concentrated flow erosion rates as affected by rock fragment cover and initial soil moisture content.** Catena. 36: 315-329.

Poesen, J.; Ingelmo-Sanchez, F. 1992. **Runoff and sediment yield from topsoils with different porosity as affected by rock fragment cover and position.** Catena. 19: 451-474.

Poesen, J.W.; Torri, D.; Bunte, K. 1994. **Effects of rock fragments on soil erosion by water at different spatial scales: a review.** Catena. 23: 141-166.

Polyakov, V.; Fares, A.; Ryder, M.H. 2005. **Precision riparian buffers for the control of nonpoint source pollutant loading into surface water: a review.** Environmental Review. 13: 129-144.

Poole, N.J.; Wildish, D.J.; Kristmanson, D.D. 1978. **The effects of the pulp and paper industry on the aquatic environment.** CRC Critical Reviews in Environmental Control. 8: 153-195.

Prosser, I.P.; Dietrich, W.E. 1995. **Field experiments on erosion by overland flow and their implication for a digital terrain model of channel initiation.** Water Resources Research. 31(11): 2867-2876.

Prosser, I.P.; Dietrich, W.E.; Stevenson, J. 1995. **Flow resistance and sediment transport by concentrated overland flow in a grassland valley.** Geomorphology. 13: 71-86.

Putthacharoen, S.; Howeler, R.H.; Jantawat, S.; Vichukit, V. 1998. **Nutrient uptake and soil erosion losses in cassava and six other crops in a psamment in eastern Thailand.** Field Crops Research. 57: 113-126.

Qiu, Z.; Prato, T. 1998. **Economic evaluation of riparian buffers in an agricultural watershed.** Journal of the American Water Resources Association. 34(4): 877-890.

Quinton, J.N.; Edwards, G.M.; Morgan, R.P.C. 1997. **The influence of vegetation species and plant properties on runoff and soil erosion: results from a rainfall simulation study in south east Spain.** Soil Use and Management. 13: 143-148.

Raabe, E.W. 1968. **Biochemical oxygen demand and degradation of lignin in natural waters.** Journal (Water Pollution Control Federation). 40(5): R145-R150.

Rackley, J.; Chung, W. 2008. **Incorporating forest road erosion into forest resource transportation planning: a case study in the Mica Creek watershed in northern Idaho.** Transactions of the American Society of Agricultural and Biological Engineers. 51(1): 115-127.

Ramsey, A.A. 1924. **Waters suitable for live stock: analyses and experiences in New South Wales.** Agricultural Gazette of New South Wales. 35: 339-342.

Rashin, E.B.; Clishe, C.J.; Loch, A.T.; Bell, J.M. 2006. **Effectiveness of timber harvest practices for controlling sediment related water quality impacts.** Journal of the American Water Resources Association. 42(5): 1307-1327.

Raut Desai, A.B. 2004. **Use of compost filter berms for sediment trapping: primary focus on water quality and structural stability.** College Station, TX: Texas A&M University. 79 p. M.S. thesis.

Reed, D.; Wolanek, M.D. 1994. **Fisheries and water quality monitoring summary for woodwaste disposal sites near St. John Harbor, Zarembo Island.** Unpublished report on file with: U.S. Department of Agriculture, Forest Service, Tongass National Forest, Petersburg, AK. 15 p.

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

Reinhart, K.G.; Eschner, A.R.; Trimble, G.R., Jr. 1963. **Effect on streamflow of four forest practices in the mountains of West Virginia.** Res. Pap. NE-1. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 79 p.

Resource Conservation District of Monterey County. 2005. **Vegetated waterways.** Salinas, CA. Available at http://www.rcdmonterey.org/Growers_Ranchers_Landowners/Planning_Design/vegetated_waterway.html (accessed July 9, 2012).

Reuss, J.O.; Johnson, D.W. 1986. **Acid deposition and the acidification of soils and waters.** Ecological Studies 59. New York: Springer-Verlag. 119 p.

Revitt, D.M.; Ellis, J.B. 1980. **Rain water leachates of heavy metals in road surface sediments.** Water Research. 14(10): 1403-1407.

Rey, F. 2003. **Influence of vegetation distribution on sediment yield in forested marly gullies.** Catena. 50: 549-562.

Rice, R.M.; Lewis, J. 1991. **Estimating erosion risks associated with logging and forest roads in northwestern California.** Water Resources Bulletin 27(5): 809-818.

Rice, R.M.; Tilley, F.B.; Datzman, P.A. 1979. **A watershed's response to logging and roads: South Fork of Caspar Creek, 1967-1976.** Res. Pap. PSW-146. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 12 p.

Richardson, J.S. 2004. **Meeting the conflicting objectives of stream conservation and land use through riparian management: another balancing act.** In: Scrimgeour, G.J.; Eisler, G.; McCulloch, B.; Silins, U.; Monita, M., eds. Proceedings, forest land – fish conference II: ecosystem stewardship through collaboration. Ottawa, ON: Natural Resources Canada, Canadian Forest Service: 1-6.

Rickson, R.J. 2000. **The use of geotextiles for vegetation management.** Aspects of Applied Biology. 58: 107-114.

Rickson, R.J. 2006. **Controlling sediment at source: an evaluation of erosion control geotextiles.** Earth Surface Processes and Landforms. 31: 550-560.

Rimoldi, P.; Zhao, A. 1996. **A three dimensional geosynthetic product for erosion control.** In: Erosion control technology: bringing it home – proceedings of conference XXVII. Denver, CO: International Erosion Control Association: 337-343.

Risse, M.; Faucette, B. 2001. **Compost utilization for erosion control.** Cooperative Extension Services Bull. 1200. Athens, GA: University of Georgia, Cooperative Extension Service. 7 p.

Rivenbark, B.L.; Jackson, C.R. 2004. **Concentrated flow breakthroughs moving through silvicultural streamside management zones: southeastern Piedmont, USA.** Journal of the American Water Resources Association. 40(4): 1043-1052.

Roads and Transportation Association of Canada [RTAC]. 1987. **Guidelines for cost effective use and application of dust palliatives.** Ottawa, ON: Roads and Transportation Association of Canada. 87 p.

Roa-Espinosa, A.; Bubenzer, G.D.; Miyashita, E.S. 1999. **Sediment and runoff control on construction sites using four application methods of polyacrylamide mix.** In: Proceedings, ASAE annual international meeting. Paper No. 99-2013. St. Joseph, MI: American Society of Agricultural Engineers. 6 p.

Roald, S.O. 1977a. **Acute toxicity of lignosulphonates on rainbow trout (*Salmo gairdneri*).** Bulletin of Environmental Contamination and Toxicology. 17(6): 702-706.

Roald, S.O. 1977b. **Effects of sublethal concentrations of lignosulphonates on growth, intestinal flora and some digestive enzymes of rainbow trout (*Salmo gairdneri*)**. Aquaculture. 12: 327-335.

Robichaud, P.R. 2005. **Measurement of post-fire hillslope erosion to evaluate and model rehabilitation treatment effectiveness and recovery.** International Journal of Wildland Fire. 14: 475-485.

Robichaud, P.R. 2010. **Post-fire stabilization and rehabilitation.** In: Cerdà, A.; Robichaud, P.R., eds. Fire effects on soils and restoration strategies. Boca Raton, FL: CRC Press: 299-320. Chapter 10.

Robichaud, P.R.; Ashmun, L.E.; Sims, B.D. 2010. **Post-fire treatment effectiveness for hillslope stabilization.** Gen. Tech. Rep. RMRS-GTR-240. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 62 p.

Robichaud, P.R.; Beyers, J.L.; Neary, D.G. 2000. **Evaluating the effectiveness of postfire rehabilitation treatments.** Gen. Tech. Rep. RMRS-GTR-63. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 85 p.

Robichaud, P.R.; Brown, R.E. 2002. **Silt fences: an economical technique for measuring hillslope soil erosion.** Gen. Tech. Rep. RMRS-GTR-94. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 24 p.

Robichaud, P.R.; Jordan, P.; Lewis, S.A.; Ashmun, L.E.; Covert, S.A.; Brown, R.E. 2013. **Evaluating the effectiveness of wood shred and agricultural straw mulches as a treatment to reduce post-wildfire hillslope erosion in southern British Columbia, Canada.** Geomorphology. 197: 21-33.

Robinson, C.A.; Ghaffarzadeh, M.; Cruse, R.M. 1996. **Vegetative filter strip effects on sediment concentration in cropland runoff.** Journal of Soil and Water Conservation. 50(3): 227-230.

Rodgers, M.; Kielty, A.; Healy, M.G. 2014. **Performance of granitic, shale, and limestone forest road aggregates subjected to repeated loading.** Journal of Transportation Engineering. 140(4): 04014002. 8 p.

Rogge, W.F.; Hildemann, L.M.; Mazurek, M.A.; Cass, G.R.; Simoneit, B.R.T. 1993. **Sources of fine organic aerosol. 3. Road dust, tire debris, and organometallic brake lining dust: roads as sources and sinks.** Environmental Science and Technology. 27(9): 1892-1904.

Römkens, M.J.M.; Wang, J.Y. 1987. **Soil roughness changes from rainfall.** Transactions of the American Society of Agricultural Engineers. 30(1): 101-107.

Rosgen, D. 1996. **Applied river morphology.** Pagosa Springs, CO: Wildland Hydrology. 350 p.

Rothwell, R.L. 1978. **Watershed management guidelines for logging and road construction in Alberta.** Information Report NOR-X-208. Edmonton, AB: Fisheries and Environment Canada, Canadian Forestry Service, Northern Forest Research Centre. 43 p.

Rothwell, R.L. 1983. **Erosion and sediment control at road–stream crossings.** Forestry Chronicle. 23: 62-66.

Ruiz, L. 2005. **Guidelines for road maintenance levels.** 0577 1205-SDTDC. Washington, DC: U.S. Department of Agriculture, Forest Service, Technology and Development Program. 54 p.

Rummer, B. 1999. **Water quality effects of forest roads in bottomland hardwood stands.** Paper 99-5051. St. Joseph, MI: American Society of Agricultural Engineers. 9 p.

Rummer, B.; Stokes, B.; Lockaby, G. 1997. **Sedimentation associated with forest road surfacing in a bottomland hardwood ecosystem.** Forest Ecology and Management. 90: 195-200.

Rustom, R.N.H. 1993. **Experimental study of soil erosion control systems: behavior and effectiveness.** Philadelphia, PA: Drexel University. 206 p. Ph.D. dissertation.

Sadeghi, A.M.; Sefton, K.A.; Faucette, L.B. 2006. **Sediment and nutrient removal from storm runoff with compost filter socks and silt fence.** In: Proceedings, annual international meeting of the ASABE. St. Joseph, MI: American Society of Agricultural and Biological Engineers. 15 p.

Sample, L.J.; Steichen, J.; Kelley, J.R., Jr. 1998. **Water quality impacts from low water fords on military training lands.** Journal of the American Water Resources Association. 34(4): 939-949.

Sanders, T.G.; Addo, J.Q. 1993. **Effectiveness and environmental impact of road dust suppressants.** Fort Collins, CO: Colorado State University, Department of Civil Engineering. 33 p.

Sanders, T.G.; Addo, J.Q.; Ariniello, A.; Heiden, W.F. 1997. **Relative effectiveness of road dust suppressants.** Journal of Transportation Engineering. 123(5): 393-397.

Sanger, D.M.; Holland, A.F.; Scott, G.I. 1999. **Tidal creek and salt marsh sediments in South Carolina coastal estuaries: II. Distribution of organic contaminants.** Archives of Environmental Contamination and Toxicology. 37: 458-471.

Sartor, J.D.; Boyd, G.B. 1972. **Water pollution aspects of street surface contaminants.** Environmental Protection Series Report No. EPA-R2-72-081. Washington, DC: U.S. Environmental Protection Agency, Office of Research and Monitoring. 257 p.

Scatena, F.N.; Larsen, M.C. 1991. **Physical aspects of hurricane Hugo in Puerto Rico.** Biotropica. 23(4a): 317-323.

Schafer, S. 2008. **Size up your soil.** 2 p. http://www.agweb.com/article/Size_Up_Your_Soil_203112 (accessed November 19, 2012).

Scheetz, B.E.; Bloser, S.M. 2008. **Environmentally sensitive maintenance practices for unpaved roads: sediment reduction study.** Final report. University Park, PA: Pennsylvania State University, Center for Dirt and Gravel Road Studies. 58 p.

Schmitt, T.J.; Dosskey, M.G.; Hoagland, K.D. 1999. **Filter strip performance and processes for different vegetation, widths, and contaminants.** Journal of Environmental Quality. 28: 1479-1489.

Schnackenberg, E.S.; MacDonald, L.H. 1998. **Detecting cumulative effects on headwater streams in the Routt National Forest, Colorado.** Journal of the American Water Resources Association. 34(5): 1163-1177.

Scholen, D.E. 1992. **Non-standard stabilizers.** Report No. FHWA-FLP-92-011. Washington, DC: U.S. Department of Transportation, Federal Highway Administration. 113 p.

Schoonover, J.E.; Williard, K.W.J.; Zaczek, J.J.; Mangun, J.C.; Carver, A.D. 2005. **Nutrient attenuation in agricultural surface runoff by riparian buffer zones in southern Illinois, USA.** Agroforestry Systems. 64: 169-180.

Schueler, T. 1994. **Performance of grassed swales along east coast highways.** Tech. Note No. 32. Watershed Protection Techniques. 1(3): 122-123.

Schueler, T.R.; Kumble, P.A.; Heraty, M.A. 1992. **A current assessment of urban best management practices: techniques for reducing non-point source pollution in the coastal zone.** Prepared for: U.S. Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds. Washington, DC: Metropolitan Washington Council of Governments, Department of Environmental Programs, Anacostia Restoration Team. 127 p.

Schueler, T.R.; Lugbill, J. 1990. **Performance of current sediment control measures at Maryland construction sites.** Final report. Washington, DC: Metropolitan Washington Council of Governments, Department of Environmental Programs. 89 p.

Schultz, D. 1998. **Current status of vegetation management in roadside ditches and stormwater management facilities.** General report. Seattle, WA: University of Washington, Center for Urban Water Resources Management. 14 p. Available at https://digital.lib.washington.edu/researchworks/handle/1773/16322 (accessed September 22, 2016).

Schultz, E. 1993. **Size-fractionated measurement of coarse black carbon particles in deposition samples.** Atmospheric Environment. Part A. General Topics. 27(8): 1241-1249.

Schwendeman, T. 1981. **Dust control study – Gallatin National Forest. Part II: Dust palliative evaluation.** Bozeman, MT: U.S. Department of Agriculture, Forest Service, Gallatin National Forest. 31 p.

Sebaaly, P.E. 1992. **Pavement damage as related to tires, pressures, axle loads, and configurations.** In: Henry, J.J.; Wambold, J.C., eds. Vehicle, tire, pavement interface, ASTM STP 1164. Philadelphia, PA: American Society for Testing and Materials: 54-68.

Sessions, J.; Balcom, J.C.; Boston, K. 1987. **Road location and construction practices: effects on landslide frequency and size in the Oregon Coast Range.** Western Journal of Applied Forestry. 2(4): 119-124.

Seybold, C.A. 1994. **Polyacrylamide review: soil conditioning and environmental fate.** Communications in Soil Science and Plant Analysis. 25(11&12): 2171-2185.

Shainberg, I.; Levy, G.J. 1994. **Organic polymers and soil sealing in cultivated soils.** Soil Science. 158(4): 267-273.

Shainberg, I.; Levy, G.J.; Rengasamy, P.; Frenkel, H. 1992. **Aggregate stability and seal formation as affected by drops' impact energy and soil amendments.** Soil Science. 154(2): 113-119.

Shainberg, I.; Warrington, D.N.; Rengasamy, P. 1990. **Water quality and PAM interactions in reducing surface sealing.** Soil Science. 149(5): 301-307.

Shalaby, A.; Reggin, A. 2002. **Surface rutting of thin pavements and gravel roads under standard and reduced tire inflation pressures.** Canadian Journal of Civil Engineering. 29(5): 679-691.

Shariff, M. 2000. **The U.S. Army experience: cold tolerance and seed viability characteristics of vetiver.** In: Proceedings of the second international vetiver conference. San Antonio, TX: The Vetiver Network International: 412-415.

Sharma, P.P. 1996. **Interill erosion.** In: Agassi, M., ed. Soil erosion, conservation, and rehabilitation. Boca Raton, FL: CRC Press: 125-152. Chapter 7.

Shaviv, A.; Ravina, I.; Zaslavsky, D. 1987. **Field evaluation of methods of incorporating soil conditioners.** Soil and Tillage Research. 9: 151-160.

Shaviv, A.; Ravina, I.; Zaslavsky, D. 1988. **Flocculation of clay suspensions by an anionic soil conditioner.** Applied Clay Science. 3: 193-203.

Sheldon, J.C.; Bradshaw, A.D. 1977. **The development of a hydraulic seeding technique for unstable sand slopes. I. Effects of fertilizers, mulches, and stabilizers.** Journal of Applied Ecology. 14: 905-918.

Shiel, R.S.; Adey, M.A.; Lodder, M. 1988. **The effect of successive wet/dry cycles on aggregate size distribution in a clay texture soil.** Journal of Soil Science. 39: 71-80.

Shipitalo, M.J.; Bonta, J.V.; Dayton, E.A.; Owens, L.B. 2010. **Impact of grassed waterways and compost filter socks on the quality of surface runoff from corn fields.** Journal of Environmental Quality. 39: 1009-1018.

Shipitalo, M.J.; Bonta, J.V.; Owens, L.B. 2012. **Sorbent-amended compost filter socks in grassed waterways reduce nutrient losses in surface runoff from corn fields.** Journal of Soil and Water Conservation. 67(5): 433-441.

Shoop, S.; Haehnel, R.; Janoo, V.; Harjes, D.; Liston, R. 2006. **Seasonal deterioration of unsurfaced roads.** Journal of Geotechnical and Geoenvironmental Engineering. 132(7): 852-860.

Sidle, R.C.; Pearce, A.J.; O'Loughlin, C.L. 1985. **Hillslope stability and land use.** Washington, DC: American Geophysical Union. 140 p.

Singer, R.D.; Stevens, J.B.; Gleason, J.R.; Baker, D.A.; Baker, T.M.; McEmber, A.V. 1982. **An environmental evaluation of dust suppressants: calcium chloride and ligninsulfonates.** Final report. Prepared for: U.S. Department of the Interior, Bureau of Mines. Minneapolis, MN: University of Minnesota, School of Public Health. 33 p.

Skaugset, A.; Allen, M.M. 1998. **Forest road sediment and drainage monitoring project report for private and state lands in western Oregon.** Prepared for: Oregon Department of Forestry; Corvallis, OR: Oregon State University, Forest Engineering Department. 20 p.

Skaugset, A.E.; Marbet, C.E.; Gilbert, E.H.; Ellingson, K.S. 2007. **The hydrology of forest roads in the central Oregon Coast Range.** In: McFarland, A.; Saleh, A., eds. Watershed management to meet water quality standards and TMDLs (total maximum daily load). Paper 701P0207. St. Joseph, MI: American Society of Agricultural Engineers: 598-604.

Skorseth, K.; Selim, A.A. 2000. **Gravel roads: maintenance and design manual – Section 1: Routine maintenance and rehabilitation.** Pierre, SD: U.S. Department of Transportation, Federal Highway Administration and South Dakota Local Transportation Assistance Program. 32 p.

Slesser, C. 1943. **Movement of calcium chloride and sodium chloride in soil.** Highway Research Board Proceedings. 23: 460-468.

Smets, T.; Poesen, J. 2009. **Effects of initial soil condition on the effectiveness of biological geotextiles in reducing interrill runoff and erosion [abstract].** In: Geophysical research abstracts: 6th European Geosciences Union general assembly. Gottingen, Germany: Copernicus Publications: 11.

Smets, T.; Poesen, J.; Bochet, E. 2008. **Impact of plot length on the effectiveness of different soil-surface covers in reducing runoff and soil loss by water.** Progress in Physical Geography. 32(6): 654-677.

Smith, D.M. 1993. **Effects of variable tire pressure on road surfacings. Vol. II: Analyses of test results.** Tech. Rep. GL-93-20. Vicksburg, MS: U.S. Army Corps of Engineers, Waterways Experiment Station. 67 p.

Smith, H.J.C.; Levy, G.J.; Shainberg, I. 1990. **Water-droplet energy and soil amendments: effect on infiltration and erosion.** Soil Science Society of America Journal. 54: 1084-1087.

Smith, J.L. 2007. **The use of rolled erosion control products (RECPs) for minimizing soil erosion.** Syracuse, NY: Syracuse University. 272 p. Ph.D. dissertation.

Smith, O.M. 1920. **The removal of clay and silica from water.** Journal of American Water Works Association. 7(3): 302-314.

Snelder, D.J.; Bryan, R.B. 1995. **The use of rainfall simulation tests to assess the influence of vegetation density on soil loss on degraded rangelands in the Baringo District, Kenya.** Catena. 25: 105-116.

Society of American Foresters. 2008. **Dictionary of forestry.** Bethesda, MD. http://dictionaryofforestry.org/ (accessed June 29, 2016).

Sojka, B.; Lentz, R. 1996. **Polyacrylamide for furrow-irrigation erosion control.** Irrigation Journal. 46(1): 8-11.

Sojka, R.E.; Bjorneberg, D.L.; Entry, J.A.; Lentz, R.D.; Orts, W.J. 2007. **Polyacrylamide in agriculture and environmental land management.** Advances in Agronomy. 92: 75-162.

Sojka, R.E.; Lentz, R.D. 1997. **Reducing furrow irrigation erosion with polyacrylamide (PAM).** Journal of Production Agriculture. 10(1): 47-51.

Soupir, M.L.; Mostaghimi, S.; Masters, A.; Flahive, K.A.; Vaughan, D.H.; Mendez, A.; McClellan, P.W. 2004. **Effectiveness of polyacrylamide (PAM) in improving runoff water quality from construction sites.** Journal of the American Water Resources Association. 40: 53-66.

Spaan, W.P.; Sikking, A.F.S.; Hoogmoed, W.B. 2005. **Vegetation barrier and tillage effects on runoff and sediment in an alley crop system on a Luvisol in Burkina Faso.** Soil and Tillage Research. 83: 194-203.

Stafford, C.; Leathers, M.; Briggs, R. 1996. **Forestry-related nonpoint source pollution in Maine: a literature review.** Misc. Rep. 399. Orono, ME: Maine Agricultural and Forest Experiment Station. 20 p.

Stagge, J.H.; Davis, A.P.; Jamil, E.; Kim, H. 2012. **Performance of grass swales for improving water quality from highway runoff.** Water Research. 46: 6731-6742.

Stapanian, M.A.; Shea, D.W. 1986. **Lignsulfonates: effects on plant growth and survival and migration through the soil profile.** International Journal of Environmental Studies. 27: 45-56.

Stedman, J.T. 2008. **To-stream sediment delivery and associated particle size distributions in unmanaged and managed forested watersheds.** Carbondale, IL: Southern Illinois University. 117 p. M.S. thesis.

Stephens, S.H. 1991. **Final report on the safety assessment of polyacrylamide.** Journal of the American College of Toxicology. 10(1):193-203.

Stevens, E.; Yeri, S.; Barfield, B.; Gasem, K.; Hayes, J. 2007. **Modeling the effectiveness of the FAEST silt fence technology.** In: Proceedings, 2007 world environmental and water resources congress. Reston, VA: American Society of Civil Engineers. 10 p.

Steward, J.E. 1994. **Tire pressure control through central tire inflation – the "lightest foot" on the road.** In: Proceedings, international road federation conference. Ottawa, ON: Transportation Association of Canada: C29-C52.

Stone, D.M.; Elioff, J.D. 1998. **Soil properties and aspen development five years after compaction and forest floor removal.** Canadian Journal of Soil Science. 78: 51-58.

Storey, B.B.; Raut Desai, A.B.; Li, M.; Landphair, H.C.; Kramer, T. 2006. **Water quality characteristics and performance of compost filter berms.** Report No. FHWA/TX-06/0-4572-1. Prepared for: Texas Department of Transportation, Research and Technology Implementation Office. College Station, TX: Texas A&M University System, Texas Transportation Institute. 119 p.

Stuart, G.W.; Edwards, P.J. 2006. **Concepts about forests and water.** Northern Journal of Applied Forestry. 23(1): 11-19.

Succarieh, M. 1992. **Final report: control of dust emissions from unpaved roads.** Report No. INE/TRC/QRP-92.05. Prepared for: Alaska Cooperative Transportation and Public Facilities Research Program. Fairbanks, AK: University of Alaska Fairbanks, School of Engineering, Institute of Northern Engineering, Transportation Research Center. 54 p.

Sucoff, E. 1975. **Effect of deicing salts on woody vegetation along Minnesota roads.** Tech. Bull. 303, Forestry Series 20. St. Paul, MN: University of Minnesota, College of Forestry; Minnesota Highway Department, Minnesota Local Road Research Board. 49 p.

Sugden, B.D.; Woods, S.W. 2007. **Sediment production from forest roads in western Montana.** Journal of the American Water Resources Association. 43(1): 193-206.

Sullivan, K. 1985. **Long-term patterns of water quality in a managed watershed in Oregon: 1. Suspended sediment.** Water Resources Bulletin. 21(6): 977-987.

Sutherland, R.A. 1998a. **A critical assessment of the research conducted at the hydraulics and erosion control laboratory—a focus on rolled erosion control systems applied to hillslopes.** Geotextiles and Geomembranes. 16: 87-118.

Sutherland, R.A. 1998b. **Rolled erosion control systems for hillslope surface protection: a critical review, synthesis and analysis of available data. I. Background and formative years.** Land Degradation and Development. 9: 465-486.

Sutherland, R.A. 1998c. **Rolled erosion control systems for hillslope surface protection: a critical review, synthesis and analysis of available data. II. The post-1990 period.** Land Degradation and Development. 9: 487-511.

Sutherland, R.A.; Ziegler, A.D. 1996. **Geotextile effectiveness in reducing interrill runoff and sediment flux.** In: Erosion control technology: bringing it home – proceedings of conference XXVII. Denver, CO: International Erosion Control Association: 393-405.

Sutherland, R.A.; Ziegler, A.D. 1997. **A field examination of rolled erosion control systems and their influence on runoff and sediment transport.** In: Working in harmony: proceedings of conference XXVIII. Denver, CO: International Erosion Control Association: 301-315.

Sutherland, R.A.; Ziegler, A.D. 2007. **Effectiveness of coir-based rolled erosion control systems in reducing sediment transport from hillslopes.** Applied Geography. 27: 150-164.

Sutherland, R.A.; Ziegler, A.D.; Tran, L.T. 1997. **Rolled erosion control systems and their effect on sediment redistribution by rainsplash – a laboratory investigation.** In: Working in harmony: proceedings of conference XXVIII. Denver, CO: International Erosion Control Association: 427-443.

Swanson, N.P.; Dedrick, A.R.; Dudeck, A.E. 1967. **Protecting steep construction slopes against water erosion.** Highway Research Record. 206: 46-52.

Swanston, D.N. 1974. **The forest ecosystem of southeast Alaska: 5. Soil mass movement.** Gen. Tech. Rep. PNW-17. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 22 p.

Swanston, D.N.; Swanson, F.J. 1976. **Timber harvesting, mass erosion, and steepland forest geomorphology in the Pacific Northwest.** In: Coates, D.R., ed. Geomorphology and Engineering. Stroudsburg, PA: Dowden, Hutchinson & Ross Inc.: 199-221. Chapter 10.

Swift, L.W., Jr. 1984a. **Gravel and grass surfacing reduces soil loss from mountain roads.** Forest Science. 30(3): 657-670.

Swift, L.W., Jr. 1984b. **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: Blackmon, B.G., ed. Proceedings, forestry and water quality: a mid-south symposium. Monticello, AR: University of Arkansas: 141-151.

Swift, L.W., Jr. 1986. **Filter strip widths for forest roads in the southern Appalachians.** Southern Journal of Applied Forestry. 10(1): 27-34.

Swift, L.W., Jr. 1988. **Forest access roads: design, maintenance, and soil loss.** In: Swank, W.T.; Crossley, D.A., eds. Forest hydrology and ecology at Coweeta. Ecological Studies Vol. 66. New York: Springer-Verlag: 313-324. Chapter 23.

SynTech Products. 2011. **Understanding your dust problem and choosing the right solution.** Toledo, OH. Available at http://www.syntechproducts.com/syntech4.ivnu (accessed February 23, 2011).

Tague, C.; Band, L. 2001. **Simulating the impact of road construction and forest harvesting on hydrologic response.** Earth Surface Processes and Landforms. 26: 131-151.

Takken, I.; Croke, J.; Lane, P. 2008. **A methodology to assess the delivery of road runoff in forestry environments.** Hydrological Processes. 22: 254-264.

Taylor, B.R.; Goudey, J.S.; Carmichael, N.B. 1996a. **Toxicity of aspen wood leachate to aquatic life: laboratory studies.** Environmental Toxicology and Chemistry. 15(2): 150-159.

Taylor, D.J. 1988. **National central tire inflation program – Olympic National Forest field operational tests.** Project Report 8871 1201. San Dimas, CA: U.S. Department of Agriculture, Forest Service, Technology and Development Center. 38 p.

Taylor, D.J.; Irwin, L.H.; Aneshansley, D.J. 1987. **A device to measure road dustiness on aggregate-surfaced roads.** Report EM 7170-8. Washington, DC: U.S. Department of Agriculture, Forest Service. 72 p.

Taylor, S.E.; Ritter, M.A.; Keliher, K.P.; Thompson, J.D. 1996b. **Portable glulam timber bridge systems.** In: Proceedings, international wood engineering conference. Vol. 2. Baton Rouge, LA: Louisiana State University: 368-375.

Tchir, J.P.; Hvenegaard, P.J.; Scrimgeour, G.J. 2004. **Stream crossing inventories in the Swan and Notikewin River basins of northwest Alberta: resolution at the watershed scale.** In: Scrimgeour, G.J.; Eisler, G.; McCulloch, B.; Silins, U.; Monita, M., eds. Proceedings, forest land – fish conference II: ecosystem stewardship through collaboration. Ottawa, ON: Natural Resources Canada, Canadian Forest Service: 53-62.

Temple, D.M. 1991. **Changes in vegetal flow resistance during long-duration flows.** Transactions of the American Society of Agricultural Engineers. 34(4): 1769-1774.

Tengbeh, G.T. 1993. **The effect of grass roots on shear strength variations with moisture content.** Soil Technology. 6: 287-295.

Terry, R.E.; Nelson, S.D. 1986. **Effects of polyacrylamide and irrigation method on soil physical properties.** Soil Science. 141(5): 317-320.

Theisen, M.S. 1992. **The role of geosynthetics in erosion and sediment control: an overview.** Geotextiles and Geomembranes. 11: 535-550.

Thompson, A.M.; Wilson, B.N.; Nguyen, H.V. 2001. **Effectiveness of erosion control blankets on reducing shear stress acting on soil particles.** In: Ascough, J.C., II; Flanagan, D.C., eds. Soil erosion research for the 21st century: proceedings of the international symposium. St. Joseph, MI: American Society of Agricultural Engineers: 28-31.

Thompson, C.H.; Kyker-Snowman, T.D. 1989. **Evaluation of non-point source pollution problems from crossing streams with logging equipment and off-road vehicles in Massachusetts: 1987-1988.** Amherst, MA: University of Massachusetts, Department of Forestry and Wildlife Management. 78 p.

Thompson, J.D.; Taylor, S.E.; Gazin, J.E.; Rummer, R.B.; Albright, R.A. 1996. **Water quality impacts from low-water stream crossings.** In: Proceedings, annual international meeting of the ASAE. Paper No. 96-5015. St. Joseph, MI: American Society of Agricultural Engineers. 15 p.

Thompson, J.D.; Taylor, S.E.; Yoo, K.H.; Brinker, R.W.; Tufts, R.A. 1994. **Water quality impacts from different forest road stream crossings.** In: Proceedings, international ASAE winter meeting. Paper No. 94-7510. St. Joseph, MI: American Society of Agricultural Engineers. 12 p.

Thompson, J.D.; Taylor, S.E.; Yoo, K.H.; Brinker, R.W.; Tufts, R.A. 1995. **Water quality impacts from different forest road stream crossings.** In: Proceedings, 18th annual meeting of the Council on Forest Engineering. Corvallis, OR: Council on Forest Engineering: 68-76.

Thompson, M.; Sessions, J.; Boston, K.; Skaugset, A.; Tomberlin, D. 2010. **Forest road erosion control using multiobjective optimization.** Journal of the American Water Resources Association. 46(4): 712-723.

Tibke, G. 1988. **Basic principles of wind erosion control.** Agriculture, Ecosystems & Environment. 22/23: 103-122.

Timberharvesting.com. 2011. **2011 Logging business survey: big squeeze.** Montgomery, AL: Timber Harvesting. 17 p. http://www.timberharvesting.com/2011-logging-business-survey-big-squeeze/ (accessed July 6, 2013).

Tisdall, J.M.; Cockroft, B.; Uren, N.C. 1978. **The stability of soil aggregates as affected by organic materials, microbial activity and physical disruption.** Australian Journal of Soil Research. 16: 9-17.

Todd, G.D.; Chessin, R.L.; Colman, J. 1999. **Toxicological profile for total petroleum hydrocarbons (TPH).** Atlanta, GA: U.S. Department of Health and Human Services, Agency for Toxic Substances and Disease Registry. 38 p.

Tollner, E.W.; Barfield, B.J.; Haan, C.T.; Kao, T.Y. 1976. **Suspended sediment filtration capacity of simulated vegetation.** Transactions of the American Society of Agricultural Engineers. 19: 678-682.

Toman, E.M. 2004. **Forest road hydrology: the influence of forest roads on stream flow at stream crossings.** Corvallis, OR: Oregon State University. 78 p. M.S. thesis.

Toman, E.M.; Skaugset, A.E. 2011. **Reducing sediment production from forest roads during wet-weather hauling.** Transportation Research Record. 2203: 13-19.

Tornatore, T.A. 1995. **Short-term impacts of forest road and skid trail stream crossings on suspended solids and turbidity.** State College, PA: Pennsylvania State University. 160 p. M.S. thesis.

Tornatore, T.A.; DeWalle, D.R.; Sharpe, W.E. 1994. **Short stream impacts associated with forest road and skid trail crossings.** Gen. Tech. Rep. NE-219. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 162-166.

Traoré, O.; Groleau-Renaud, V.; Plantureux, S.; Tubeileh, A.; Bœuf-Tremblay, V. 2000. **Effect of root mucilage and modelled root exudates on soil structure.** European Journal of Soil Science. 51: 575-581.

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

Trimble, G.R., Jr.; Weitzman, S. 1953. **Soil erosion on logging roads.** Soil Science Society Proceedings. 17(2): 152-154.

Trout, T.J.; Ajwa, H. 2001. **Polyacrylamide effects on infiltration in San Joaquin Valley sandy loam soils.** In: Proceedings, ASAE annual international meeting. Paper No. 01-2259. St. Joseph, MI: Society of Agricultural and Biological Engineers. 8 p.

Trout, T.J.; Sojka, R.E.; Lentz, R.D. 1995. **Polyacrylamide effect on furrow erosion and infiltration.** Transactions of the American Society of Agricultural Engineers. 38(3): 761-765.

Truong, P. 2000. **Vetiver grass system: potential applications for soil and water conservation in northern California.** In: Proceedings, stiff grass technology seminar. Woodland, CA: Yolo County Flood Control and Water Conservation District: 562-571.

Urroz, G.E.; Israelsen, C.E. 1995. **Effectiveness of selected erosion control materials under simulated rain and sunlight.** Mountain-Plains Consortium Report No. 95-44. Logan, UT: Utah State University, College of Engineering, Utah Water Research Laboratory. 30 p.

U.S. Department of Transportation. 1995. **Best management practices for erosion and sediment control.** Report No. FHWA-FLP-94-005. Sterling, VA: U.S. Department of Transportation, Federal Highway Administration. 187 p.

U.S. Environmental Protection Agency [U.S. EPA]. 1990. **EPA Superfund explanation of significant differences.** EPA/ESD/R07-90/509. Kansas City, MO: U.S. Environmental Protection Agency, Region 7. 9 p.

U.S. Environmental Protection Agency [U.S. EPA]. 1991. **Standards for managing specific hazardous wastes and specific types of facilities (Part 266).** EPA/530-SW-91-0621. Washington, DC: U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response. 392 p.

U.S. Environmental Protection Agency [U.S. EPA]. 1993. **Guidance specifying management measures for sources of nonpoint source pollution in coastal waters.** EPA 840-B-92-002. Washington, DC: U.S. Environmental Protection Agency, Office of Water. 822 p.

U.S. Environmental Protection Agency [U.S. EPA]. 1995. **Compilation of air pollutant emission factors. Vol. 1: Stationary point and area sources.** Publ. No. AP-42. 5th ed. Research Triangle Park, NC: U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards. Available at https://www.epa.gov/air-emissions-factors-and-quantification/ap-42-compilation-air-emission-factors (accessed September 28, 2016).

U.S. Environmental Protection Agency [U.S. EPA]. 1996. **National air pollutant emission trends, 1900-1995.** EPA-454/R-96-007. Research Triangle Park, NC: U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards. 74 p.

U.S. Environmental Protection Agency [U.S. EPA]. 1999. **Vegetated swales.** Storm Water Technology Fact Sheet No. 832-F-99-006. Washington, DC: U.S. Environmental Protection Agency, Office of Water. 7 p.

U.S. Environmental Protection Agency [U.S. EPA]. 2002. **Development document for proposed effluent guidelines and standards for the construction and development category.** EPA-821-R-02-007. Washington, DC: U.S. Environmental Protection Agency, Office of Water. 245 p.

U.S. Environmental Protection Agency [U.S. EPA]. 2010a. **Compost filter berm fact sheet.** Washington, DC. Available at http://cfpub.epa.gov/npdes/stormwater/menuofbmps/index.cfm?action=factsheet_results&view=specific&bmp=119 (accessed October 18, 2012).

U.S. Environmental Protection Agency [U.S. EPA]. 2010b. **Compost filter socks fact sheet.** Washington, DC. Available at http://cfpub.epa.gov/npdes/stormwater/menuofbmps/index.cfm?action=browse&Rbutton=detail&bmp=120&minmeasure=4 (accessed October 18, 2012).

U.S. Environmental Protection Agency [U.S. EPA]. 2012. **National menu of stormwater best management practices.** Washington, DC: U.S. Environmental Protection Agency, National Pollutant Discharge Elimination System (NPDES). Available at http://cfpub.epa.gov/npdes/stormwater/menuofbmps/index.cfm (accessed November 19, 2012).

U.S. Forest Service. 1981. **Interim report in research, Horse Creek Administration Research Project.** Boise, ID: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station: 126-129.

U.S. Forest Service. 1993. **Central tire inflation: What's in it for me?** FS-415. San Dimas, CA: U.S. Department of Agriculture, Forest Service, San Dimas Technology and Development Center. 3 p.

U.S. Forest Service. 2009. **Forest Service handbook (FSH) 7709.59- Road system operations and maintenance handbook.** Chapter 60 – Road maintenance. Available at http://www.fs.fed.us/cgi-bin/Directives/get_dirs/fsh?7709.59 (accessed June 30, 2016).

U.S. Forest Service. [N.d.]. **Forest Service Water Erosion Prediction Project.** U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Air, Water, & Aquatic Environments Program. http://www.fs.fed.us/rm/boise/AWAE/projects/water_erosion_prediction_project.shtml (accessed May 19, 2016).

U.S. National Archives and Records Administration. 2015. **Code of federal regulations.** Title 36, chapter II, part 212, subpart A, section 212.5. Road system management. Available at https://www.law.cornell.edu/cfr/text/36/212.5 (accessed June 30, 2016).

U.S. Roads. 1998. **Dust: Don't eat it! Control it!** Reprinted with permission in Road Management and Engineering Journal. 4 p. Available at http://www.usroads.com/journals/rmej/9806/rm980603.htm (accessed March 9, 2011).

Vacher, C.A.; Loch, R.J.; Raine, S.R. 2003. **Effect of polyacrylamide additions on infiltration and erosion of disturbed lands.** Australian Journal of Soil Research. 41: 1509-1520.

Van Dijk, P.M.; Kwaad, F.J.P.M.; Klapwijk, M. 1996. **Retention of water and sediment by grass strips.** Hydrological Processes. 10(8): 1069-1080.

van Groenewoud, H. 1977. **Interim recommendation for the use of buffer strips for the protection of small streams in the Maritimes.** Information Report M-X-74. Fredericton, NB: Canadian Forestry Service, Maritimes Forest Research Centre. 18 p.

van Meerveld, H.J.; Baird, E.J.; Floyd, W.C. 2014. **Controls on sediment production from an unpaved resource road in a Pacific maritime watershed.** Water Resources Research. 50: 4803-4820.

Vanacker, V.; von Blanckenburg, F.; Govers, G.; Molina, A.; Poesen, J.; Deckers, J.; Kubik, P. 2007. **Restoring dense vegetation can slow mountain erosion to near natural benchmark levels.** Geology. 35(4): 303-306.

Vandaele K.; Poesen, J. 1995. **Spatial and temporal patterns of soil erosion rates in an agricultural catchment, central Belgium.** Catena. 25: 213-226.

Veith, T.L.; Wolfe, M.L.; Heatwole, C.D. 2003. **Optimization procedure for cost effective BMP placement at a watershed scale.** Journal of the American Water Resources Association. 39(6): 1331-1343.

Virginia Department of Conservation and Recreation. 1999. **Virginia stormwater management handbook, vol. 1.** Richmond, VA: Virginia Department of Conservation and Recreation, Division of Soil and Water Conservation. 619 p.

Wade, C.R. 2010. **Evaluation of best management practices for bladed skid trail erosion control and determination of erosion model accuracy and applicability.** Blacksburg, VA: Virginia Polytechnic Institute and State University. 109 p. M.S. thesis.

Wagenbrenner, J.W.; MacDonald, L.H.; Rough, D. 2006. **Effectiveness of three post-fire rehabilitation treatments in the Colorado Front Range.** Hydrological Processes. 20: 2989-3006.

Wahbeh, A.M. 1990. **Dust control for secondary limestone roads using bentonite.** Ames, IA: Iowa State University. 90 p. M.S. thesis.

Wald, A.R. 1975. **The impact of truck traffic and road maintenance on suspended-sediment yield from a 14' standard forest road.** Seattle, WA: University of Washington. 38 p. M.S. thesis.

Walden, C.C. 1976. **The toxicity of pulp and paper mill effluents and corresponding measurement procedures.** Water Research. 10: 639-664.

Wallace, A.; Wallace, G.A.; Abouzamzam, A.M. 1986. **Effects of excess levels of a polymer as a soil conditioner on yields and mineral nutrition of plants.** Soil Science. 141(5): 377-380.

Wallace, G.A.; Wallace, A. 1986. **Control of soil erosion by polymeric soil conditioners.** Soil Science. 141(5): 363-367.

Walley, H.D.; King, R.B.; Ray, J.M.; Robinson, J. 2005. **What should be done about erosion mesh netting and its destruction of herpetofauna?** Journal of Kansas Herpetology. 16: 26-28.

Walling, D.E. 1983. **The sediment delivery problem.** Journal of Hydrology. 65: 209-237.

Walsh, F.M.; Barrett, M.E.; Malina, J.F., Jr.; Charbeneau, R.J. 1998. **Use of vegetative controls for treatment of highway runoff.** Tech. Rep. No. FHWA/TX-7-2954-2. Austin, TX: University of Texas at Austin, Center for Transportation Research. 130 p.

Wang, T.; Spyridakis, D.E.; Mar, B.W.; Horner, R.R. 1981. **Transport deposition and control of heavy metals in highway runoff.** Report No. WA-RD-39.10. Olympia, WA: Washington State Department of Transportation. 43 p.

Wargo, R.S.; Weisman, R.N. 2006. **A comparison of single-cell and multicell culverts for stream crossings.** Journal of the American Water Resources Association. 42(4): 989-995.

Warrington, D.; Shainberg, I.; Agassi, M.; Morin, J. 1989. **Slope and phosphogypsum's effects on runoff and erosion.** Soil Science Society of America Journal. 53: 1201-1205.

Washington Forest Practices Board. 2000. **Washington Forest Practices Board manual.** Olympia, WA. 36 p.

Wasniewski, L.W. 1994. **Hillslope sediment routing below new forest roads in central Idaho.** Corvallis, OR: Oregon State University. 105 p. M.S. thesis.

Watkins, S.H. 1970. **Bacterial degradation of lignosulfonates and related model compounds.** Journal (Water Pollution Control Federation). 42(2), Part 2: R47-R56.

Watson, J.; Frazier, C.; Chow, J.; Farber, R.J.; Countess, R.J. 1989. **Survey of fugitive dust control methods for PM10.** In: Proceedings, 82nd annual meeting and exhibition. Pittsburgh, PA: Air and Waste Management Association. 20 p.

Watson, J.G. 1996. **Final study plan for: Effectiveness demonstration of fugitive dust control methods for public unpaved roads and unpaved shoulders on paved roads.** DRI Doc. No. 685-5200.1F2. Prepared for: San Joaquin Valley Unified Air Pollution Control District. Reno, NV: Desert Research Institute, Energy and Environmental Engineering Center.

Watson, J.G.; Chow, J.C.; Pace, T.G. 2000. **Fugitive dust emissions.** In: Davis, W.T., ed. Air pollution engineering manual. New York: John Wiley & Sons, Inc.: 117-135. Chapter 4.

Watt, J.; Marcus, R. 1974. **Effect of lignosulfonate on the colon of guinea-pigs.** Proceedings of the Nutritional Society. 33: 65A-66A.

Watt, J.; Marcus, S.N. 1976. **Effect of various salts of lignosulfonate on the colon of guinea-pigs.** Proceedings of the Nutritional Society. 35: 76A.

Weaver, W.; Hagans, D. 2004. **Road upgrading, decommissioning and maintenance – estimating costs on small and large scales.** In: Allen, S.T.; Thomson, C.; Carlson, R., eds. Proceedings, salmon habitat restoration cost workshop. Portland, OR: Pacific States Marine Fisheries Commission: 80-103.

Weeks, L.E.; Colter, W.G. 1952. **Effects of synthetic soil conditioners on erosion control.** Soil Science. 73: 473-484.

Weeth, H.J.; Haverland, L.H. 1961. **Tolerance of growing cattle for drinking water containing sodium chloride.** Journal of Animal Science. 20: 518-521.

Weeth, H.J.; Haverland, L.H.; Cassard, D.W. 1960. **Consumption of sodium chloride water by heifers.** Journal of Animal Science. 19: 845-851.

Weggel, J.R.; Rustom, R. 1992. **Soil erosion by rainfall and runoff – state of the art.** Geotextiles and Geomembranes. 11: 551-572.

Weiss, P.T.; Gulliver, J.S.; Erickson, A.J. 2010. **The performance of grassed swales as infiltration and pollution prevention practices: a literature review.** 50 p. http://stormwater.dl.umn.edu/sites/g/files/pua941/f/media/weiss-gulliver-erickson_2010_-_performance_of_grassed_swales_as_infiltration_and_pollutation_prevention_practices.pdf (accessed June 30, 2016).

Weitzman, S. 1952. **Mountain log roads – design and construction.** Facts from Fernow Forest. Parsons, WV: West Virginia Conservation. 16: 16-21.

Welborn, C.T.; Veenhuis, J.E. 1987. **Effects of runoff controls on the quantity and quality of urban runoff at two locations in Austin, Texas.** Water-Resources Investigation Report No. 87-4004. Austin, TX: U.S. Geological Survey. 101 p.

Welch, R.A., Jr.; Taylor, S.E.; Yoo, K.H.; Thompson, J.D.; Rummer, R.B.; Albright, R.A. 1998. **Life-cycle water quality impacts from low-water stream crossings.** Paper No. 98-7027. St. Joseph, MI: American Society of Agricultural Engineers. 24 p.

Weller, D.E.; Jordan, T.E.; Correll, D.L. 1998. **Heuristic models for material discharge from landscapes with riparian buffers.** Ecological Applications. 8(4): 1156-1169.

Wellman, E.A.; Barraclough, S. 1972. **Establishment of acceptable dusting criteria for aggregate surface roads.** Administrative Study 7110. Klamath Falls, OR: U.S. Department of Agriculture, Forest Service, Winema National Forest.

Wellman, J.C.; Combs, D.L.; Cook, S.B. 2000. **Long-term impacts of bridge and culvert construction or replacement on fish communities and sediment characteristics of streams.** Journal of Freshwater Ecology. 15(3): 317-328.

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

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

Wemple, B.C.; Swanson, F.J.; Jones, J.A. 2001. **Forest roads and geomorphic process interactions, Cascade Range, Oregon.** Earth Surface Processes and Landforms. 26: 191-204.

Wenger, S. 1999. **A review of the scientific literature on riparian buffer width, extent and vegetation.** Athens, GA: University of Georgia, Institute of Ecology, Office of Public Service and Outreach. 59 p.

Whipple, W., Jr.; Hunter, J.V. 1979. **Petroleum hydrocarbons in urban runoff.** Water Resources Bulletin. 15(4): 1096-1105.

Wiebe, A.H.; Burr, J.G.; Faubion, H.E. 1934. **The problem of stream pollution in Texas with special reference to salt water from oil fields.** Transactions of the American Fisheries Society. 64: 81-86.

Wiest, R.L. 1998. **A landowner's guide to building forest access roads.** Tech. Publ. NA-06-98. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Area State and Private Forestry. 45 p.

Wilber, W.G.; Hunter, J.V. 1979. **Distribution of metals in street sweepings, stormwater solids, and urban aquatic sediments.** Journal (Water Pollution Control Federation). 51: 2810-2822.

Williams, D.S.; Shukla, M.K.; Ross, J. 2008. **Particulate matter emission by a vehicle running on unpaved road.** Atmospheric Environment. 42: 3899-3905.

Wischmeier, W.H.; Mannering, J.V. 1965. **Effect of organic matter content of the soil on infiltration.** Journal of Soil and Water Conservation. 20(4): 150-152.

Wischmeier, W.H.; Smith, D.D. 1958. **Rainfall energy and its relationship to soil loss.** Transactions of the American Geophysical Union. 39(2): 285-291.

Wisconsin Department of Natural Resources. 2006. **Silt fence (1056).** Madison, WI: Wisconsin Department of Natural Resources. 4 p. http://dnr.wi.gov/topic/stormwater/documents/silt_fence_1056_v2.pdf (accessed June 9, 2016).

Wisconsin Transportation Information Center [WTIC]. 1997. **Dust control on unpaved roads.** Wisconsin Transportation Bull. No. 13. Madison, WI: University of Wisconsin. 6 p.

Wishowski, J.M.; Mamo, M.; Bubenzer, G.D. 1998. **Trap efficiencies of filter fabric fence.** In: Proceedings, annual meeting of the ASAE. Paper No. 98-2158. St. Joseph, MI: American Society of Agricultural Engineers. 14 p.

Witmer, P.L.; Stewart, P.M.; Metcalf, C.K. 2009. **Development and use of a sedimentation risk index for unpaved road–stream crossings in the Choctawhatchee watershed.** Journal of the American Water Resources Association. 45(3): 734-747.

Witt, E.L.; Barton, C.D.; Stringer, J.W.; Bowker, D.W.; Kolka, R.K. 2011. **Evaluating best management practices for ephemeral channel protection following forest harvest in the Cumberland Plateau – preliminary findings.** In: Fei, S.; Lhotka, J.M.; Stringer, J.W.; Gottschalk, K.W.; Miller, G.W., eds. Proceedings, 17th central hardwood forest conference. Gen. Tech. Rep. NRS-GTR-P-78. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station: 365-374.

Wohlgemuth, P.M.; Robichaud, P.R. 2007. **The effects of selected post-fire emergency rehabilitation techniques on small watershed sediment yields in southern California.** In: Furniss, M.; Clifton, C.; Ronnenberg, K., eds. Proceedings, Forest Service national earth sciences conference: advancing the fundamental sciences. Gen. Tech. Rep. PNW-GTR-689. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 36-42.

Wolanek, M.D. 1995. **Wood fiber road construction influences on stream water quality in southeast Alaska.** Unpublished report (draft), on file with: U.S. Department of Agriculture, Forest Service, Tongass National Forest. 26 p.

Wolfe, M.D.; Williams, J.W. 1986. **Rates of landsliding as impacted by timber management activities in northwestern California.** Bulletin of the Association of Engineering Geologists. 23(1): 53-60.

Wollum, A.G., II. 1962. **Grass seeding as a control for roadbank erosion.** Res. Note PNW-218. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 5 p.

Wong, S.L.; McCuen, R.H. 1982. **The design of vegetative buffer strips for runoff and sediment control.** Tech. Paper. College Park, MD: University of Maryland. 25 p.

Woo, M.; Fang, G.; diCenzo, P.D. 1997. **The role of vegetation in the retardation of rill erosion.** Catena. 29: 145-159.

Wooldridge, D.D. 1979. **Suspended sediment from truck traffic on forest roads, Meadow and Coal Creeks.** Seattle, WA: University of Washington, College of Forest Resources. 33 p.

Wright, K.N. 2010. **Evaluation of check dams for sediment control on disturbed land surfaces.** Urbana, IL: University of Illinois at Urbana-Champaign. 114 p. M.S. thesis.

Wu, J.S.; Allan, C.J.; Saunders, W.L.; Evett, J.B. 1998. **Characterization and pollutant loading estimation for highway runoff.** Journal of Environmental Engineering. 124: 584-592.

Wyant, D.C. 1980. **Evaluation of filter fabrics for use as silt fences.** VHTRC 80-R49. Charlottesville, VA: Virginia Highway & Transportation Research Council. 37 p. + appendices.

Xiao, B.; Wang, Q.; Wu, J.; Huang, C.; Yu, D. 2010. **Protective function of narrow grass hedges on soil and water loss on sloping croplands in northern China.** Agriculture, Ecosystems & Environment. 139: 653-664.

Yanders, A.F.; Orazio, C.E.; Puri, R.K.; Kapila, S. 1989. **On translocation of 2,3,7,8-tetrachlorodibenzo-p-dioxin: time dependent analysis at the Times Beach experimental site.** Chemosphere. 19(1-6): 429-432.

Yanosek, K.A.; Foltz, R.B.; Dooley, J.H. 2006. **Performance assessment of wood strand erosion control materials among varying slopes, soil textures, and cover amounts.** Journal of Soil and Water Conservation. 61(2): 45-51.

Yap, P. 1989. **Truck tire types and road contact pressures.** In: Proceedings, second international symposium on heavy vehicle weights and dimensions. Ottawa, ON: Transportation Association of Canada. 17 p.

Yardley, D.H.; Lacabanne, W.D.; Nelson, C.R. 1980. **Airborne dust from taconite tailings.** Washington, DC: U.S. Department of the Interior, Bureau of Mines. 51 p.

Yee, C.S. 1976. **Small rock buttresses – useful for some forest road slump failures.** Journal of Forestry. 74(10): 688-689.

Yeri, S.; Barfield, B.; Stevens, E.; Gasem, K.; Jeyarathan, A.; Hayes, J. 2005. **Effective storm water and sediment control during pipe line construction using new filter fence concept.** In: Proceedings, 12th annual international petroleum environmental conference. Tulsa, OK: Integrated Petroleum Environmental Consortium (IPEC). 15 p.

Yonge, D. 2000. **Contaminant detention in highway grass filter strips.** Res. Rep. No. WA-RD 474.1. Prepared for: Washington State Department of Transportation, Research Office. Pullman, WA: Washington State University, Department of Civil and Environmental Engineering. 66 p.

Young, R.A.; Wiersma, J.L. 1973. **The role of rainfall impact in soil detachment and transport.** Water Resources Research. 9(6): 1629-1636.

Yousef, Y.A.; Wanielista, M.P.; Harper, H.H. 1985. **Removal of highway contaminants by roadside swales.** Transportation Research Record. 1017: 62-68.

Yu, S.L.; Barnes, S.L.; Gerde, V.W. 1993. **Testing of best management practices for controlling highway runoff.** Report No. VTRC 93-R16. Charlottesville, VA: Virginia Transportation Research Council. 60 p.

Yu, S.L.; Kuo, J.; Fassman, E.A.; Pan, H. 2001. **Field test of grassed-swale performance in removing runoff pollution.** Journal of Water Resources Planning and Management. 127(3): 168-171.

Zanders, J.M. 2005. **Road sediment: characterization and implications for the performance of vegetated strips for treating road run-off.** Science of the Total Environment. 339: 41-47.

Zellmer, S.D.; Edgar, D.E.; Isaacson, H.R. 1991. **Erosion control on a steeply sloped pipeline right-of-way in southwestern Pennsylvania.** Gas Research Institute Final Report 005832. Des Plaines, IL: Gas Research Institute. 19 p.

Zhang, X.C.; Miller, W.P. 1996a. **Physical and chemical crusting processes affecting runoff and erosion in furrows.** Soil Science Society of America Journal. 60: 860-865.

Zhang, X.C.; Miller, W.P. 1996b. **Polyacrylamide effect on infiltration and erosion in furrows.** Soil Science Society of America Journal. 60: 866-872.

Ziegler, A.D.; Sutherland, R.A. 1998. **Reduction in interrill sediment transport by rolled erosion control systems.** Soil and Tillage Research. 45: 265-278.

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

Ziegler, A.D.; Sutherland, R.A.; Tran, L.T. 1997. **Influence of rolled erosion control systems on temporal rainsplash response – a laboratory rainfall simulation experiment.** Land Degradation and Development. 8: 139-157.

Zitko, V.; Carson, W.V.; Carson, W.G. 1975. **Thallium: occurrence in the environment and toxicity to fish.** Bulletin of Environmental Contamination and Toxicology. 13(1): 23-30.

APPENDIX

English, Metric, and Gradient Conversions Applicable to Text, Tables, and Figures

Converting metric units to English units

If you know	Multiply by	To convert to
cm	0.394	inch
g	0.0022	lb
g L^{-1}	0.0624	lb ft^{-3}
g L^{-1}	0.00835	lb gal^{-1}
g m^{-2}	0.000205	lb ft^{-2}
g mm^{-1}	0.056	lb $inch^{-1}$
g m^{-2} h^{-1} (mm precip)$^{-1}$	3.613×10^{-5}	lb $inch^{-2}$ h^{-1} (inch precip)$^{-1}$
ha	2.471	ac
kg	2.205	lb
kg ha^{-1}	0.892	lb ac^{-1}
kg 100 m^{-2}	0.205	lb 100 ft^{-2}
kg km^{-1}	3.548	lb mi^{-1}
kg m^{-2}	0.205	lb ft^{-2}
kg m^{-3}	0.0624	lb ft^{-3}
kg min^{-1} m^{-1}	0.672	lb min^{-1} ft^{-1}
kg mm^{-1}	55.99	lb $inch^{-1}$
km	0.621	mi
km^2	0.386	mi^2
kPa	0.145	psi (lb $inch^{-2}$)
L m^{-2}	0.0245	gal ft^{-2}
L s^{-1}	0.0353	ft^3 s^{-1}
m	3.281	ft
m^2	10.76	ft^2
m^3	35.31	ft^3
Mg	1.102	ton
ml	0.0338	oz
mm	0.0394	inch
m^3 ha^{-1}	14.29	ft^3 ac^{-1}
Mg ha^{-1} yr^{-1}	0.446	ton ac^{-1} yr^{-1}
µg L^{-1}	1	ppb
µm	3.937×10^{-5}	inch
mg kg^{-1}	1	ppm
mg L^{-1}	1,000	ppb
mg L^{-1}	1	ppm
N·m ha^{-1}	0.298	ft-lb ac^{-1}
tonne	1.102	ton
tonne ha^{-1}	0.446	ton ac^{-1}
tonne km^{-1}	1.774	ton mi^{-1}
tonne km^{-2} day^{-1}	2.855	ton mi^{-2} day^{-1}

Converting English units to metric units

If you know	Multiply by	To convert to
ac	0.405	ha
ft	0.305	m
ft^2	0.0929	m^2
ft^3	0.0283	m^3
ft-lb ac^{-1}	3.350	N·m ha^{-1}
ft^3 ac^{-1}	0.07	m^3 ha^{-1}
ft^3 s^{-1}	28.32	L s^{-1}
gal ft^{-2}	40.75	L m^{-2}
inch	2.54	cm
inch	25.4	mm
inch	25,400	µm
lb	453.6	g
lb	0.454	kg
lb ac^{-1}	1.121	kg ha^{-1}
lb ft^{-2}	4,882.43	g m^{-2}
lb ft^{-2}	4.882	kg m^{-2}
lb 100 ft^{-2}	4.882	kg 100 m^{-2}
lb ft^{-3}	16.019	kg m^{-3}
lb ft^{-3}	16.019	g L^{-1}
lb gal^{-1}	119.83	g L^{-1}
lb $inch^{-1}$	17.86	g mm^{-1}
lb $inch^{-1}$	0.0179	kg mm^{-1}
lb $inch^{-2}$ h^{-1} (inch precip)$^{-1}$	27,680.37	g m^{-2} h^{-1} (mm precip)$^{-1}$
lb mi^{-1}	0.282	kg km^{-1}
lb min^{-1} ft^{-1}	1.488	kg min^{-1} m^{-1}
mi	1.609	km
mi^2	2.59	km^2
oz	29.57	ml
ppb	0.001	mg L^{-1}
ppb	1	µg L^{-1}
ppm	1	mg L^{-1}
ppm	1	mg kg^{-1}
psi (lb $inch^{-2}$)	6.895	kPa
ton	0.907	tonne
ton	0.907	Mg
ton ac^{-1}	2.242	tonne ha^{-1}
ton ac^{-1}	2.242	Mg ha^{-1}
ton mi^{-1}	0.564	tonne km^{-1}
ton mi^{-2} day^{-1}	0.350	tonne km^{-2} day^{-1}

APPENDIX

English, Metric, and Gradient Conversions Applicable to Text, Tables, and Figures

Factors for gradient conversions

If you know	Multiply by	To convert to
degrees[a] (°)	tan (degrees) × 100	percent slope (%)
percent slope (%)	$\tan^{-1}$ (percent slope/100)	degrees (°)

[a] The conversion from degrees to percent slope holds only for gradients that are <90°.

Edwards, Pamela J.; Wood, Frederica; Quinlivan, Robin L. 2016. **Effectiveness of best management practices that have application to forest roads: a literature synthesis.** Gen. Tech. Rep. NRS-163. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 171 p.

Literature describing the effectiveness of best management practices (BMPs) applicable to forest roads is reviewed and synthesized. Effectiveness is considered from the perspective of protecting water quality and water resources. Both paved and unpaved forest roads are considered, but BMPs that involve substantial engineering are not considered. Some of the BMPs included are commonly used on roads; others are used less often. The synthesis focuses on quantitative BMP effectiveness and descriptions of processes or characteristics that influenced the effectiveness. Qualitative results and observations not supported by data are excluded. Most of the effectiveness results describe sediment losses and sediment delivery, but there is also some coverage of chemicals used as BMPs, such as dust palliatives and soil conditioners. Chapters and subheadings are based on how or where protection is provided, or type of BMP. The final chapter provides information on research needs and potential direction of BMP implementation in the future. Although there remains a great need to quantify BMP effectiveness more rigorously across more physiographic, topographic, climate, and soil conditions, the data provided in this synthesis give road and watershed managers and landowners a starting place for evaluating and selecting BMPs.

KEY WORDS: erosion, sediment delivery, road location, drainage control, soil protection, road characteristics, research needs

In accordance with Federal civil rights law and U.S. Department of Agriculture (USDA) civil rights regulations and policies, the USDA, its Agencies, offices, and employees, and institutions participating in or administering USDA programs are prohibited from discriminating based on race, color, national origin, religion, sex, gender identity (including gender expression), sexual orientation, disability, age, marital status, family/parental status, income derived from a public assistance program, political beliefs, or reprisal or retaliation for prior civil rights activity, in any program or activity conducted or funded by USDA (not all bases apply to all programs). Remedies and complaint filing deadlines vary by program or incident.

Persons with disabilities who require alternative means of communication for program information (e.g., Braille, large print, audiotape, American Sign Language, etc.) should contact the responsible Agency or USDA's TARGET Center at (202) 720-2600 (voice and TTY) or contact USDA through the Federal Relay Service at (800) 877-8339. Additionally, program information may be made available in languages other than English.

To file a program discrimination complaint, complete the USDA Program Discrimination Complaint Form, AD-3027, found online at http://www. ascr.usda.gov/complaint_filing_cust.html and at any USDA office or write a letter addressed to USDA and provide in the letter all of the information requested in the form. To request a copy of the complaint form, call (866) 632-9992. Submit your completed form or letter to USDA by: (1) mail: U.S. Department of Agriculture, Office of the Assistant Secretary for Civil Rights, 1400 Independence Avenue, SW, Washington, D.C. 20250-9410; (2) fax: (202) 690-7442; or (3) email: program.intake@usda.gov.

USDA is an equal opportunity provider, employer, and lender.

Northern Research Station

www.nrs.fs.fed.us