THESIS

AMENDMENT EFFECTS ON SOIL PHYSICAL PROPERTIES AND RESTORATION OF DECOMMISSIONED FOREST ROADS

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In partial fulfillment of the requirements

For the Degree of Master of Science

Colorado State University

Fort Collins, Colorado

Fall 2017

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ABSTRACT

AMENDMENT EFFECTS ON SOIL PHYSICAL PROPERTIES AND RESTORATION OF DECOMMISSIONED FOREST ROADS

Unsealed forest roads, including logging roads and unauthorized roads created by hunters, miners, and recreational users, generate significant harmful effects to local ecosystems and waterways. Rapid restoration of these roads is necessary to prevent erosion, downstream implications for water quality, and a variety of other deleterious ecosystem impacts. Soil amendments, including mulches, composts, and other materials, offer promise to improve soil health, restore soil structure, and support revegetation of these sites. I tested the viability of three locally-sourced soil amendments - wood straw mulch, Biosol fertilizer, and biochar - alone and in paired combinations to restore soil physical properties important for improved hydrologic function and plant growth. I found that amendment combinations of biochar + mulch and biochar + Biosol significantly reduced soil bulk density when compared to unamended controls. Other factors (aggregate stability, infiltration, sediment production) suggested potential for improvement relative to unamended control plots, but no significant differences between treatments were observed due to high variability within and between sites. Regression analyses revealed that soil physical properties, particularly wet aggregate stability, was significantly correlated with key soil erosion parameters such as infiltration and runoff, suggesting aggregate stability could provide a useful measure of soil restoration success.

DEDICATION

Nobody achieves anything alone.

I would like to dedicate this work to the following individuals for their contributions to my personal and professional growth: the late Roy Evans and Phil Hoeck, Dennis Benjamin, Ernesto Ortiz, Glenn Mason, Rebecca Loncosky, mom, dad, brother Joe, Steven Fonte, Charles Rhoades, Mark Paschke, Nora Flynn, Cassandra Schnarr, Sergi Domenech, Lee Friesen, Emmanuel Deleon, Lewis Messner, Julie Bushey, Matt Ramlow, M. Francesca Cotrufo, Greg Butters, Ann Hess, Joel Schneekloth, Phil Turk, Lee MacDonald, Bob Schindelbeck, the Colorado State University Soil, Water, and Plant Testing Laboratory, Wildlands Restoration Volunteers, the BANR project (Agriculture and Food Research Initiative Competitive Grant no. 2013-68005-21298 from USDA National Institute of Food and Agriculture), the US Forest Service, and every other supervisor, mentor, educator, and peer who helped me, in one capacity or another, to reach this milestone. I would be nothing without your generosity, patience and expertise.

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BC-BIOCHAR
BS-BIOSOL
C-CONTROL
GWC-GRAVIMETRIC WATER CONTENT
MU-MULCH
MWD-MEAN WEIGHT DIAMETER
RO-RUNOFF PROPORTION
SED-SEDIMENT PRODUCTION
TTR-TIME TO RUNOFF
USFS-UNITED STATES FOREST SERVICE

1. LITERATURE REVIEW

Roads, particularly unimproved roads, generate a plethora of challenges for land managers charged with simultaneously protecting and enabling access to public spaces. The network of unimproved roads in United States Forest Service (USFS) - administered lands created by logging operations, miners, hunters, and other recreational users is extensive and has a potentially massive area of ecological influence. Along with the vast network of authorized roads, roughly 96,500 km of unauthorized roads are estimated to exist on USFS-managed lands (USDA Forest Service, 2002). Since they are often created by logging activities or recreational users, unauthorized roads are usually unimproved dirt surfaces and hastily constructed with little or no consideration to their environmental impacts. Further, since these roads do not officially exist, maintenance is not normally performed on them. Unmaintained dirt or gravel roads often generate more ecological harm than similar maintained roads (Grayson et al., 1993) magnifying their damage over time. Indeed, unimproved roads can generate considerably more suspended material in surface runoff than even extremely disturbed agricultural soils (Motha et al., 2004). To mitigate the negative ecological effects of roads and to prevent unconstrained access to public lands, the USFS regularly decommissions unneeded roads (Gucinski et al., 2001). A wide range of road impacts on the environment have been documented (Forman & Alexander, 1998; Jones et al., 2000; Trombulak & Frissell, 2000; Coffin, 2007) and a synthesis of this literature, particularly issues related to unpaved forest roads like those studied here, is provided below.

1.1 Abiotic effects of roads and their immediate consequences on the biotic community

A large portion of road-associated damage stems from the physical disturbance of soils directly underneath and adjacent to the path of the road. Construction, maintenance, and use of roads compacts soil, negatively impacts soil structure, and results in soil, air, and water pollution.

These disturbances alter soil hydrologic regimes, contributing to erosion and impaired soil biological communities as well as other damage to local and regional ecosystems.

1.1.1 Soil structure

Soil forms from organic matter and mineral particle clusters of three size components: sand, silt, and clay-sized particles, the relative proportions of each is commonly used to describe a soil's *texture*, a property which has profound influence over water and nutrient cycling processes. These primary particles then combine to form aggregates of varying size that determine soil structure, with considerable implications for air and water movement and a range of soil biological processes. Foot and vehicle traffic causes most of the harm to bare road soils via soil compaction, the mechanical destruction of aggregates, and the loss of pore space (Iverson et al., 1981). Bulk density, one of the most-commonly used indicators of compaction, is defined as the mass of dry soil divided by its volume (including soil pores). Measurement of bulk density can be done via multiple approaches depending on the soil context, with varying degrees of complexity and accuracy (Blake & Hartge, 1986; Page-Dumroese et al., 1999). Since bulk density is used as an indicator of compaction, it can also be used to estimate soil porosity (Flint & Flint, 2002), which largely determines air and water movement and a range of associated soil functions. As bulk density increases, porosity generally decreases. A reduction in total pore space can reduce soil's water storage capacity and can restrict gas flow as well. The effects of compaction on soil biota vary; compaction can generate both positive and negative effects on soil and biota, though upper limit of roughly 1.7 g cm^{-3} was suggested to exist, with negative effects on soil structure (Horn & Fleige, 2009) and microbial communities (Beylich et al., 2010) mainly occurring once bulk density exceeds this value.

Compaction damages soil structure by breaking soil aggregates and eliminating pore space necessary for natural aeration, heat, and water movement. Associated reductions in infiltration and hydraulic conductivity lead to increased erosion and a number of related ecological consequences (Assouline et al., 1997; Jones et al., 2000; Beylich et al., 2010). The immediate and short term effects of compaction are well-documented, though there is little research into the long-term response of soil to compaction. Thorud & Frissell (1976) artificially compacted forest soils and compared their bulk densities to undisturbed control plots over time. They reported natural decompaction of surface soil layers (~ 8 cm deep) in less than nine years, though compaction effects persisted below 15 cm in depth. Some studies reviewed by Håkansson & Reeder (1994) reported soil compaction up to fifty years after road use ceased. Intuitively, soil compaction depth and persistence were found to increase both with increased vehicle axle mass and increased number of passes on roads, with compaction measured as deep as 50 cm. Given that many unauthorized roads were originally logging roads, they likely experienced extreme compaction under the weight of loaded trucks and heavy logging machinery. The volume of traffic also influences the degree of compaction and its extent into soil, as even infrequent use of relatively light off-road vehicles can generate extensive harm to ecosystems by compacting soil, damaging habitat, inducing soil erosion, etc. (Brown, 1994; Davies et al., 2016). Iverson et al. (1981) reported that soil bulk density increases logarithmically with increasing number of vehicle passes, with the greatest density increases occurring during the first few passes. Their rainfall simulations showed increased runoff and sediment production, reduced resistance to overland flow resulting from reduced surface roughness, and channelized runoff, also resulting in increased flow velocity, in vehicle-disturbed plots. Natural biotic activity can alleviate compaction through the action of root growth, burrowing, and the addition of organic matter, but biological activity tends to be concentrated on and near the soil surface and reduces with depth. Thus, deep soil compaction can persist even when surface compaction is relieved. Establishment of plant cover is a principal goal of road restoration efforts (Switalski et al., 2004). Roots interact with other organisms, extract and cycle resources, and can help repair soil structure through the formation of macropores and contributing to aggregation. However, plant establishment and continued growth is greatly limited by the ability of roots to extend through soil pores (Passioura, 1991). Soil penetration resistance, defined as “force required to drive or push a device into the soil” (Lowery & Morrison Jr., 2002), increases linearly with bulk density (Ehlers et al., 1983). Thus, in compacted soils roots are not able to grow as deep and don’t contribute C inputs (e.g. sloughed roots, exudates) resulting in lower biological activity at depth in disturbed soils (Whalley et al., 1995). Plant roots may prefer to follow pre-existing burrows or root channels, but compaction hinders burrowing and also reduces the available pore volume for root growth. Seed germination and early survival of plants is strongly dependent on soil properties disturbed by roads, including compaction (Bewley, 1997), since limitations to resource flow in soil can prevent germination until those stresses are relieved. Davies et al. (2016) found measurable decreases in invertebrate species composition, with reduced richness, diversity, and abundance on off-road vehicle-disturbed sands attributed to compaction, rutting, and sand displacement. Trampling from foot traffic inflicts similar damage to soils as vehicle traffic and likewise increases with trampling intensity (Whitecotton et al, 2000). Trampling harms surface soils, with destruction of soil aggregates and reduced soil porosity, specifically microporosity, generating most of the damage to soil water and gas exchange (Alaoui & Helbling, 2006; Zhao et al., 2010).

1.1.2 Soil hydrology and mass wasting

Largely a consequence of soil compaction and the resultant reduced porosity, surface and subsurface water flows can become significantly altered by roads. Other factors also change soil hydrology, including the relative hillslope position of a road, the construction of roadside structures like ditches and berms, the generally linear nature of roads, and the presence of stream crossings. These road-induced disturbances to the natural soil hydraulic regime can have a profound effect on the hydraulic "connectivity" of landforms and geographic features within an ecosystem (Cooper, 2007); the consequences of disrupted hydrologic connectivity range in geographic extent from the patch to the catchment and beyond. While soil texture is a major determinant of compaction effects, soils also respond differently depending upon other physical and chemical characteristics, including pH, organic matter content, and clay content. Compacted soils usually have smaller and fewer macropores than uncompacted soils, with a consequent decrease in the ratio of macropores:micropores explaining much of the behavior (Richard et al., 2001). Macropores fill with water and drain more-rapidly than micropores due to stronger capillary forces in micropores according to Darcy's Law, causing compacted soils to have increased water retention and retarded flow characteristics despite having lower total pore space. The hydrological connection of roads to stream networks alters stream hydrology. Overland flow is more likely to become channelized on bare road surfaces when rills and gullies form, producing a similar effect to that of ditches and increasing the volume of sediment delivered to streams (Croke & Mockler, 2001). Combined, these channels effectively increase the extent of stream networks in drainage basins, amplifying the peak discharge of streams and reducing the duration of peak flooding events (Wemple et al., 1996); the physical features of the affected streams are consequently altered, partially by increased debris flow and sediment deposition and partially due to the erosive force of flowing water, thus generating a range of potential consequences for riparian communities (Jones et al., 2000).

Road and hill topography have a profound influence on the hydrological effects generated by roads (Wemple et al., 1996). For example, unsealed roads on steeper slopes are more likely to form gullies that channelize more water than on gentler slopes, and increasing channel length before reaching a drain generates a similar effect. Additionally, roads cut into the side of steep hills can intercept subsurface water flow, converting it to surface flow and increases in water and sediment loads downstream (Megahan, 1972). Soil surface roughness can be changed by the mechanical disturbance from traffic, raindrop impact, and overland flow with resulting changes in runoff and sediment load (Huang et al., 2002).

Soil aggregation contributes to a number of key soil functions that include protecting soil material from erosion, creating pore space through which water and nutrients can flow, and storing nutrients (Bronick & Lal, 2005; Six et al., 2000; Tisdall & Oades, 1982). Raindrops destroy aggregates, dislodge sediment particles that clog soil pores, compact soil, and when overland flow occurs, releases dislodged soil particles into the broader ecosystem. A multi-event rainfall simulation run by Luce (1997) on decommissioned forest roads compared pre- and post-ripping hydraulic conductivity of a ripped soil to that of a ripped and mulched soil. He concluded that surface sealing by sediment clogging of soil macropores did not occur on mulch-protected soils since mulch intercepted raindrops, preventing their kinetic energy from transferring to soil; plant cover and leaf litter protects soil from raindrop damage and erosion via similar mechanisms (Bochet et al., 2006; Geddes & Dunkerley, 1999). Raindrop impact is the dominant driver of soil transport in all but steeply sloped soils, but overland flow can also transport weakly-bound or detached soil particles on the soil surface to be deposited elsewhere (Bilby et al., 1989; Ferro, 1998). Sediment deposited on the soil surface results in crust formation that inhibits infiltration and increases surface runoff and the volume of water in streams (Moore & Singer, 1990; Morin & Van Winkel, 1996). A range of ecosystem effects of sediment transported into water bodies has been documented; effects vary in intensity depending on both concentration and exposure (Newcombe & Macdonald, 1991), with eutrophication of downstream water bodies likely having the greatest impact (Smith et al., 1998). Erosion of mountainous road soils is more severe on the lower one-third of a hill slope (Bloom, 1998; Madej, 2001) due to the surface and subsurface water burden from all soils higher on the slope within the drainage basin.

Following road establishment the highest rates of erosion occur during the first few years after disturbance and with rare high-intensity rainstorms (Beschta, 1978; Megahan, 1974; Megahan et al., 2001). The relatively linear nature of roads channelizes runoff and increases its velocity, magnifying erosion potential. Soils with reduced or no plant cover suffer from increased erosion vulnerability due to raindrop impact and surface runoff (Vaezi et al., 2017), though even highly disturbed soils may develop erosion-resistance over time (Megahan, 1974; Luce & Black, 2001) when detached soil particles form surface-sealing crusts (Moore, 1981). Wet soils are especially susceptible to disturbance and erosion, and a study of logging roads showed that suspended logging activity during rain corresponded with a reduction of suspended material in surface runoff (Grayson et al., 1993). Given the depth of soil compaction can extend at least 50 cm underground, reduced hydraulic conductivity of subsurface soils can create a dam of denser soil that resists lateral subsurface water flow. Subsurface "pooling" can lead to severe consequences such as landslides. For example, McClelland et al. (1999), studying on sediment production after large rain events in the Clearwater National Forest of Idaho, found over half of landslides were related to roads. Landslides on mountainous forest roads were also studied by Wemple et. al., (2001). They found that roads provided a net increase in basin-wide sediment production, mostly from mass wasting debris.

1.1.3 Chemical pollution

Though not as impactful as compaction issues, road soils can also face severe issues associated with reduced resource flow (water, nutrients, oxygen, etc.), changes to soil chemistry, and generation of sediment pollution through erosional processes (reviews by Forman & Alexander, 1998; Trombulak & Frissell, 2000; Coffin, 2007). Pollutants such as heavy metals, organic compounds, and nitrogen-oxide compounds from fuel leaks and gaseous emissions contaminate soils, water bodies, and the atmosphere with many deleterious impacts for the ecosystems in which roads exist. Heavy metals accumulate in soil and biota, and both these and other chemical pollutants can enter the local watershed and have impacts downstream (Coffin, 2007; Trombulak & Frissell, 2000).

Vehicle-related contaminants are commonly detected near roads and can be dispersed hundreds of meters into ecosystems. Chemical contamination decreases exponentially with distance from roads (reviewed by Trombulak & Frissell, 2000), but can still reach great distances if transported by water (reviewed by Coffin, 2007). Biota sensitive to road contaminants or other effects suffer, potentially hindering restoration efforts. Since seeding with native species is a common restoration method, pollution preventing the growth of plants (Bignal et al., 2007) introduced during decommissioning activities can impede restoration efforts.

1.2 Ecological (biotic) effects of road disturbance

In the previous section, I discussed how roads disturb the physical environment, and how biota respond to these disturbances. Vegetation removal and degraded soil structure of roads alters habitats and communities and can have a series of ecological consequences; this extends well beyond the immediate vicinity of a road.

Like abiotic effects, most biotic impacts of roads are negative, though some positive effects can occur. Construction equipment and processes destroy and fragment habitat and cause mortality or injury to organisms unable to escape. The harm to species due to habitat loss and fragmentation is well documented (eg. Bender et al., 1998; Carlson, 2000; Cushman, 2006; Prugh et al., 2008), as habitat loss is the leading cause of species extinction and is more likely to induce extinction than habitat fragmentation (Fahrig, 1997).

Road construction removes vegetation on and often near the path of the road. This, combined with the abiotic effects of roads, causes roads to generate edge effects in terrestrial ecosystems (Murcia, 1995) and extends ecological harm to aquatic ecosystems. Edge effects such as changes to species richness (Pocock & Lawrence, 2005), alterations to animal social behavior (Fuentes-Montemayor et al., 2009), and increased light penetration (Delgado & Fernández-Palacios, 2007) can extend as far as 900 m from road edge and possibly further, but most effects occur within 5 m (Avon et al., 2010). A review by Greenberg et al. (1997) showed that exotic species often benefit from road disturbances while a range of native plant and animal species are harmed. Genetic diversity of species usually suffers when affected by roads (Holderegger & Di Giulio, 2010) through barrier effects including inducing road-avoidance behavior, roadkill mortality, trampling (Kissling et al., 2009), and increased hunting pressure (Janeau et al., 1999).

Soil material lost from roads by erosion and mass wasting directly harms life in aquatic ecosystems (Chutter, 1969; Wood & Armitage, 1997). Suspended sediment increases the turbidity of waters resulting in reduced light penetration to aquatic primary producers and altering water temperature. Sediment deposited on the floors of lakes and rivers can also smother aquatic biota through means such as impaired feeding and destroyed or modified habitat (Newcombe & Macdonald, 1991).. Given that vegetation is often the primary agent responsible for preventing these issues, its removal during road construction and the resultant degradation of soils therefore generates widespread environmental degradation.

1.3 Restoring roads

Given the ecological consequences of roads, unneeded roads should be prevented and removed as quickly as possible to reduce their environmental impact; road decommissioning is therefore an important component of public lands management. The USFS decommissions roads using many different methods depending upon restoration goals , including blocking entrances, scattering debris on roadbeds, revegetation, and recontouring (Coghlan & Sowa, 1998). Restoration methods typically include decompaction by ripping as first step for restoring decommissioned roads due to the range of immediately beneficial effects and relatively simple implementation. Ripping soils is a simple and effective technique, but particularly disturbed soils may simply “break” into large clods along fracture planes rather than become finely tilled if unsuitable equipment is used (McNabb, 1994). While effective at alleviating compaction, ripping alone may not be sufficient in all scenarios given individual restoration objectives. Madej (2001) applied common decommissioning methods (ripping, draining, seeding & mulching; and three levels of recontouring intensity) to forest roads and found that, while effective at reducing erosion compared to un-treated roads, treated roads nevertheless still experience significant erosion and produced large quantities of sediment. Similarly to ripping, recontouring the road surface to match the surrounding topography reduces soil bulk density (Lloyd et al., 2013) with numerous other benefits to be expected as well.

Road restoration is a complex venture and while abundant literature exists on the immediate ecological effects of roads and their restoration, there is much less emphasis on the long-term responses to road restoration. Application of seed and amendments, including mulches and fertilizers, may increase the likelihood of successful restoration but knowledge of the effectiveness of such additions is lacking and purchasing and transporting large volumes of amendments can be costly. Revegetating roads is often a short-term restoration goal, since plants stabilize eroding soils and can physically impede access to vehicles or foot traffic, but can also be a long-term restoration strategy. Germination and seedling establishment is a crucial step towards restoring a disturbed ecosystem's functionality and soil properties play a large role in this process. Tolerance to poor soil quality varies by species; exotics, weeds and other early successional species are short-lived, but pave the way for the long-term productivity of an ecosystem. As early successional species colonize a site they create physical and biological conditions favorable to later successional species by improving nutrient and water flow through soil (Jones et al., 1994). Established vegetation can provide protection to vulnerable developing seedlings via a nurse-protégé interaction (Cody, 1993). These interactions are most common in arid and semi-arid ecosystems (Flores & Jurado, 2003), but could also be leveraged in other environments (Ren et al., 2008). Quick establishment of these interactions could improve restoration success. For example, Krauss et al. (2010) predicted and experimentally observed delayed extinction of plant species in altered habitats after disturbance, and as previously discussed, soil physical problems like compaction and erosion can persist for great lengths of time, supporting an argument that ideal road remediation techniques should produce benefits as quickly as possible.

Amending soils with additional material like fertilizers and mulches, whether by surface application, mixed into soil, or a combination of the two are another common restoration technique. Mulches or other surface residue can help reduce soil erosion, improve soil moisture, and enhance structure. Land managers can apply soil amendments like mulch, compost, and fertilizer to protect, provide nutrients, and support beneficial soil biota, ideally during the first years after restoration when soils are most vulnerable to erosion and mature ecological communities are not yet established. Mulch primarily serves in a protective capacity: mulch protects soil from erosion, moderates soil temperature and moisture, and facilitates plant germination and establishment (Chalker-Scott, 2007). Mulch keeps otherwise-bare soil moist, and physically protects seedlings during their vulnerable early growth stages (Bronick & Lal, 2005), though these responses are not guaranteed. Ripping and seeding treatment combinations on road restoration plots increased plant cover and density in seeded plots when compared to control (Elseroad et al., 2003), but mulching and topsoil additions had no effect. Forest road sediment production decreases with increased mulch cover percentage (Burroughs & King, 1989), important since recently ripped soils would have little surface residue to protect them from erosion. Luce (1997) ran a rainfall simulation on recently-ripped roads studying crusting and soil compaction due to precipitation. Treatments with mulch showed improved hydraulic conductivity rates and lower bulk densities when compared to ripped-only and control soils, and suggested that mulch protected soil from raindrop impact and splash erosion by reducing water velocity, resulting in less surface sealing and infiltration resistance. Reduced water velocity of raindrops and surface runoff by mulch also reduces surface erosion, since water's ability to carry suspended material increases with velocity.

Fertilizers have been used since antiquity to improve soil nutrient availability with the goal of facilitating plant growth, particularly agricultural crops. A review by Switalski et al. (2004) of road restoration practices showed that fertilizers can be successfully employed to encourage revegetation of decommissioned forest roads, as road soils are often nutrient-poor. However, the effects of fertilizer additions can be short-lived and may encourage weed growth (Maynard & Hill, 1992; Paschke et al., 2000), so should ideally be used only in nutrient deficient soils. Like fertilizers, composts have also been successfully used in road restoration projects. Composts can be used to improve soil nutrition, but they provide additional value by contributing to soil organic matter, inoculating soils with soil microbes, reducing bulk density, improving erosion resistance, increasing plant available water, and modulating soil chemistry (Curtis & Claassen, 2009; Henry & Bergeron, 2005). Compost performance varies with soil texture and the parent material of compost (Duong et al., 2012). For example, when testing the response of soils of varying textures to compost additions, Duong et al. found the sandiest soils (sand > 80%) in their pot study responded with the greatest increase in soil respiration, and compost derived from agricultural waste increased soil aggregation more than compost derived from garden waste. Haynes et al., (2013) examined the impacts of mushroom compost, among other soil amendments, on infiltration and vegetation establishment in North Carolina field sites with regular vehicle traffic and found that compost-fertilized soils did not have improved infiltration rates, but that improvements in vegetation cover and decompaction by tillage enhanced restoration outcomes.

Biochar, pyrolyzed carbon derived from incomplete combustion of biomass (Lehmann & Joseph, 2009), is of increasing interest to a variety of stakeholders due to its potential to increase soil carbon sequestration (Du et al., 2016), enhance soil structure and water-holding capacity, reduce penetration resistance (Mukherjee & Lal, 2013), modify soil chemical properties like pH (Kim et al., 2013), and promote biological activity (Lehmann et al., 2011), all properties important to plant growth. Recent interest in biochar has revealed great utility as a restoration amendment due to its wide array of effects on abiotic and biotic processes in soil (Lehmann & Joseph, 2009; Lehmann et al., 2011). While an extensive body of literature exists on biochar, its extremely variable nature means investigations frequently present conflicting results. Detailed analyses of biochar properties (eg. Allaire et al., 2015; Teβin, 2016) reveal considerable differences in biochar structure potentially leading to conflicting results in the literature. For example, Hardie et al. (2014) studied the effect of biochar's (made from an *Acacia* sp.) on soil water and physical properties and found no change in aggregate stability in soils of an apple orchard amended with biochar, while Fungo et al. (2017) found increased aggregation only in biochar soils when other amendments were applied as well. Elzobair et al. (2016) found biochar had no impact on the soil microbial community, while Chen et al., (2017) reported increases in microbial activity in biochar-amended soils.

2. AMENDMENT EFFECTS ON SOIL PHYSICAL PROPERTIES AND RESTORATION OF DECOMMISSIONED FOREST ROADS

2.1 Introduction

Roads, particularly unimproved roads, generate a plethora of challenges for land managers charged with simultaneously protecting, and enabling access to remote places. In areas administered by the United States Forest Service (USFS), the network of unimproved roads, created by logging and mining operations and for a variety of recreational uses, is extensive and has a potentially massive area of ecological influence. Along with the vast network of authorized roads, roughly 96,500 km of unauthorized roads are estimated to exist on USFS-managed lands (USDA, 2000). These roads typically consist of unimproved, unsealed surfaces. A wide range of environmental effects of roads has been documented including disrupted soil and watershed hydrology (Cooper, 2007; Jones et al., 2000), increased soil erosion and landslides (McClelland et al., 1999; Wemple et al., 2001), and an array of other negative ecological effects including habitat loss, behavior modification, and mortality (Forman & Alexander, 1998; Trombulak & Frissell, 2000). Foot and/or vehicle traffic combined with precipitation causes most of the harm to bare road soils by the mechanical destruction of soil structure, compaction, and erosion (Håkansson & Reeder, 1994; Iverson et al., 1981; Sosa-Pérez & MacDonald, 2017a; Whitecotton et al., 2000).

To mitigate the negative ecological effects of roads and to prevent unconstrained access to public lands the USFS regularly decommissions unneeded roads (Gucinski et al., 2001). Decommissioning contrasts with abandonment in that one or more remediation techniques are used in addition to closure to speed the recovery of road soils and encourage recruitment of biota in a decommissioning project. When common decommissioning methods were applied (i.e., ripping, mulching), significant reductions in soil erosion and increased revegetation of decommissioned roads in the Red Feather Lakes region of Northern Colorado were found in under a year (Sosa-Pérez & MacDonald, 2017b). Thus, road decommissioning is a method of environmental restoration, while closure alone makes no active attempt to restore disturbed soils. Roads present many restoration challenges due to their numerous abiotic and biotic effects, and these challenges can be exacerbated by continued disturbance and the passage of time. Based on these challenges, there exists an impetus to quickly and efficiently deny access to, and restore ecological function of, unauthorized forest roads.

Restoration methods typically include decompaction by ripping the most compacted soil layers followed by seeding. Soil amendments such as mulches, composts, fertilizers, and other organic matter additions have been shown to reduce erosion, and expedite revegetation and the remediation of disturbed soils (Chalker-Scott, 2007; Elseroad et al., 2003; Luce, 1997; Madej, 2001); quick remediation of unsealed roads is desirable due to the extensive spatial and temporal harm potential of unremediated roads on local and regional ecosystems (Beschta, 1978; Delgado et al., 2007; Pocock & Lawrence, 2005). Further, vegetation can physically impede continued road usage and mask the road's presence, increasing the importance of revegetating roads as quickly as possible. Soil amendments such as mulch, fertilizer, and compost are often used in road decommissioning projects because they can hasten ecosystem recovery by enhancing soil nutrient content, improving soil structure, mitigating erosion and compaction, and facilitating plant recruitment and growth. Biochar, pyrolyzed biomass, has gained considerable recent interest as a soil amendment (Lehmann & Joseph, 2009) due to its potential to contribute to numerous desirable soil functions including carbon sequestration, nutrient and water storage and alterations to pH and soil biological communities, among other effects (Lehmann et al., 2011;

Lehmann & Joseph, 2009). Past research suggests biochar could be an effective soil amendment for use in soil restoration projects (Mukherjee et al., 2014; Rhoades et al., 2017; Tess et al., 2014; Thomas & Gale, 2015) particularly where abundant feedstock is locally available for biochar production. However, few have explored biochar's impact in a road restoration setting, with most studies of biochar having focused on non-road soils in forests and agricultural fields, or other contexts.

Based on the clear need for restoration and the promise offered by a number of soil amendments to improve diverse restoration indicators, and given that ample pine-beetle killed biomass was available nearby (Klutsch et al., 2009), this study sought to explore the potential of locally available soil amendments on soil structure and water movement, key determinants of erosion and vegetation establishment. More specifically, this study investigated decommissioned forest roads created by logging activities on sandy mountainous soils. I hypothesized that soil amendments, alone and in combination, would enhance soil structure (aggregation, compaction/bulk density) and associated water dynamics (infiltration, sediment runoff, water storage) in decommissioned forest roads undergoing restoration relative to unamended controls.

2.2 Methods

2.2.1 Study area description

This research was conducted in the Canyon Lakes Ranger District of the Roosevelt National Forest near Red Feather Lakes, Colorado. With an elevation of approximately 2,750 m, this region has annual precipitation between 381 and 508 mm with approximately 14% falling as rain and 86% typically falling as snow ("Red Feather Lakes CO Climate Summary," n.d.) and 75 – 100 frost free days per year (Moreland, 1980). Average monthly temperatures range from 0.8° C in January to 23.1° C in July.

All studied soils formed from weathered granite and support native vegetation including grasses dominated by blue grama, forbs, shrubs, cactus, and lodgepole pine (Moreland, 1980). Soil pH was largely neutral across sites (average 6.9), while soil textures included sandy loams, loamy sands, and sandy clay loams (Table 1). While much of the terrain is mountainous with steep slopes (3-60%), the soils examined here all occurred on decommissioned Forest Service roads with relatively gentle slopes (≤10%).

2.2.2 Study design

In Fall 2014, four sites were considered for testing of different soil amendment combinations. Roads were decommissioned approximately one week prior to treatment application and blocked off with physical obstructions (large boulders, tree trunks, and barbed wire fencing) at either end to prevent traffic. Prior to treatment establishment, the entire test road at each study site was decompacted to 30 cm depth with a three-tine ripper. Seven plots ~3 x 5 m were established along the length of each road, controlling for slope, with a 1 m buffer between each plot. Each plot received one of seven randomly assigned treatments: control (no amendments), biochar (BC), biochar + Biosol (BC + BS), biochar + mulch (BC + MU), Biosol (BS), Biosol + mulch (BS + MU), and mulch (MU), with all treatments applied to each of the four replicate road sites (blocks). In plots with biochar, biochar was raked into soil manually, while in plots with Biosol, this material was sprayed in liquid form evenly across the appropriate plots. All plots then received a hand-broadcast of a locally-sourced native seed mixture. Finally, mulch was applied on appropriate plots and was left on the soil surface. Biochar was applied at a rate of 25 Mg ha^{-1}, Biosol at 225 kg ha^{-1}, and wood straw mulch at 500 bales ha^{-1}. Biochar was produced from local pine beetle-killed trees pyrolyzed at 550° C with chips generally less than 2.5 mm (H:C 0.70, C:N 255.3, ash content 1.2%, pH 8.49, and 0.32% N). Biosol Forte, a commercially-available organic fertilizer manufactured from mushroom compost, contained 7% N, 2% P (P_2O_5), and 3% K (K_2O)(Sandoz Gmbh, 2016). Wood straw mulch was also produced from pine beetle-killed trees by shredding wood to < 2 cm. All field measurements occurred in June 2016. Paired sub-samples for all parameters measured were conducted by plot to address expected heterogeneity.

2.2.3 Field sampling

Sampling occurred in late June 2016, but sampling issues in one of the sites resulted in incomplete data collection and eventual exclusion from analysis. Therefore, just three sites were analyzed. In each plot, potential infiltration, runoff, and associated sediment production were evaluated using a Cornell Sprinkle Infiltrometer (Ogden et al., 1997) calibrated to deliver 0.5 cm water min^{-1} according to methods outlined by (van Es & Schindelbeck, 2006). Two metal rings (22 cm dia.) were inserted into the soil within each plot to a depth of 7.5 cm along the centerline of the road and a small hole was excavated to the side of each ring for collecting runoff. The infiltrometers were then placed on top of each ring and water was delivered via a series of small drop emitters (7.5 cm above the soil surface) over a 45-min. period. Runoff was collected from a port on the downhill side of the metal ring which was level with the soil surface and fitted with a hose to drain the water into a container at the side of the ring. Runoff and infiltration readings were recorded every minute for the first five min., then once every three min. thereafter. The data collected were used to calculate time to initial runoff (maximum of 2700s equals no runoff observed), total runoff volume, total infiltration volume, and infiltration rate (mL min^{-1}). Additionally, sediment in runoff was estimated by collecting an aliquot of a known volume of the water-sediment solution from each infiltration test. After returning to the laboratory, this solution was oven dried at 105° C, the remaining sediment weighed, and the quantity of sediment in each aliquot was used along with total runoff volume to calculate total soil loss per infiltration test.

Adjacent to each infiltration test a small pit was dug to estimate bulk density using an excavation method (Blake & Hartge, 1986). A 20 x 20 x 10 cm pit was lined with a thin cloth sheet and filled with millet seed. This seed was then extracted and its volume measured to determine the pit volume. Rocks were removed from the soil and weighed along with the field moist soil. Soil samples (0-10 cm) were then collected from the edge of this pit (and adjacent to each infiltration test) for evaluation of field gravimetric water content, wet aggregate stability, texture, and available water holding capacity measurements in the laboratory. All samples were placed in sealed plastic bags within protective plastic sleeves (to avoid compaction), and kept cool until returned to the laboratory.

2.2.4 Laboratory analyses

The field-moist soils were gently passed thought through an 8 mm sieve by breaking soil clods along natural planes of weakness to homogenize soils and remove large organic debris (roots, twigs, etc.) and rocks. This soil was then air-dried for subsequent analyses. Additionally, a sub-sample of the field-moist soil was weighed and dried at 105° C to determine gravimetric water content. I analyzed water-stable aggregate fractions following a wet-sieving method adapted from Elliott (1986). A 40 g representative sub-sample of the air-dried soil was placed on a 2 mm sieve and slaked in deionized water for 5 min. Samples were then hand-sieved in water by gently moving the sieve in and out of the water in a vertical motion 50 times over a 2 min. period. This process used a series of progressively smaller sieves: 2 mm, 250 µm, and 53 µm to create four size fractions: large macroaggregates (diameter > 2 mm), small macroaggregates (250 µm - 2 mm), microaggregates (53 - 250 µm), and the silt & clay fraction (< 53 µm). Material remaining on the top of each sieve was transferred to pre-weighed aluminum pans and dried at 60° C to determine the mass of each fraction. Rocks > 2mm were separated and weighed apart. Aggregate mean weight diameter (MWD) of each sample was calculated according to van Bavel (1950) using the following formula:

$$MWD = \sum_{i=1}^{n} dw$$

Where *n* = number of size fractions, *d* = maximum diameter of each size fraction, and *w* = size fraction's proportion of total sample mass.

In addition to aggregate stability, available water storage capacity was measured as the difference between water content at saturation vs. at 10 kPa using a pressure plate according to (Scanlon et al., 2002). Soil texture was determined by hydrometer method (Gee & Bauder, 1986).

2.2.5 Statistical analyses

Treatment comparisons were conducted on average values for pairs of measurements taken in each plot, except in a few cases of missing data points where only a single value was available due to excessively rocky terrain. All data were checked for assumptions of ANOVA and transformed using natural log, inverse, or Box-Cox transformations as necessary. All analyses were conducted using R version 3.4.1 (R Development Core Team, 2017). The seven treatments across the three sites were compared using ANOVA with a randomized complete block approach using the *lme4* package (Bates et al., 2015) with each site considered a block and treated as a random effect. Tukey-adjusted pairwise comparisons of all treatment combinations were examined using the *lsmeans* package (Lenth, 2016). I also examined correlations between the measured variables across all treatments and blocks; Pearson's correlation coefficients and significance levels were calculated using the *Hmisc* package (Harrell Jr., 2017).

2.3 Results

2.3.1 Soil structural properties

Clear treatment differences between treatments were observed for bulk density, an indicator of soil compaction. The BC-BS and BC-MU treatments were found to be 29.49% and 26.82% lower, respectively, than soil in the control plots ($p = 0.014$; Fig. 1). While there were notable differences between sites, no other significant treatment differences were observed (Table 2). While not significant, I note that all treatments suggested increased mean aggregate stability relative to the control (Table 2).

2.3.2 Soil hydraulic properties

High variability was observed both between and within sites for several measures related to soil water movement and storage and no significant treatment effects were found. While not significantly different, amended plots tended to have a longer time to the initiation of runoff (TTR) relative to the control, as well as higher mean proportion of simulated rainfall occurring as runoff (Table 2). No treatment differences were observed in available water storage capacity or gravimetric water content of the field-moist soil.

2.3.3 Relationships between measured variables

Correlations between key variables are summarized in Table 3. Aggregate stability (MWD) was negatively correlated with the proportion of added water occurring as runoff ($p < 0.005$, $r2 = 0.511$; Fig. 2) and positively correlated with infiltration rate ($p < 0.005$, $r2 = 0.530$; Fig. 3). MWD was also found to be negatively correlated with production of sediment in the runoff ($p = 0.007$, $r2 = 0.344$, Fig. 5) and showed a marginally significant positive correlation with TTR ($p =$

0.054, $r^2 = 0.191$, Table 3). Increasing time to runoff corresponds with a reduction of surface runoff ($p < 0.005$, $\alpha = 0.05$, Fig. 4).

2.4 Discussion

2.4.1 Treatment impacts on soil physical properties

Despite high variability both within and across the experimental sites, several notable trends were observed in examining the effect of different amendments on soil physical properties. I found that both BC-BS and BC-MU resulted in significantly lower bulk density than the control plots (Fig 1). This finding is broadly in agreement with others who have reported that biochar additions can reduce soil compaction (Andrenelli et al., 2016; Hardie et al., 2014; Peake et al., 2014). Since all of the amendments tested are considerably less dense than soil, the reductions in bulk density may simply be the result of diluting the soil particles with a lighter material. Interestingly, I only observed a significant effect in plots that received biochar together with mulch or Biosol amendments. This may simply be due to an additive effect associated with greater overall amounts of material being incorporated into the soil in the BC-BS and BC-MU treatments. However, other mechanisms may also play an important role in driving the observed differences in bulk density. For example, biochar was manually mixed into plots. The mechanical disturbance of the shoveling action could have resulted in even more decompaction, but I do not see this effect in the other BC-containing treatment.

I note that higher bulk density in the control fits well with the general trend of lower aggregate stability in the control. While not significant here, higher MWD would be expected to decrease bulk density due to the creation of larger macropores between larger vs. smaller aggregates (Bronick & Lal, 2005; Sun & Lu, 2014). Mulch and Biosol alone and in combination showed no significant bulk density reduction, implying that additions of these organic materials alone are insufficient to affect soil structure, and that biochar is likely playing an important role in the observed bulk density effects despite the different natures of mulch and Biosol. A number of studies have suggested biochar, whether alone or in combination with other amendments, can improve aggregation (Du et al., 2016; Fungo et al., 2017; Sun & Lu, 2014); however, the effects are not always consistent and the mechanisms not entirely clear. For example, Wang et al, (2017) found biochar to increase aggregation in a silt loam soil of California and attributed this to microbial utilization of biochar as a substrate. Ajayi et al. (2016) reported increased aggregate strength in biochar-amended sandy soils and determined biochar increased soil surface area, and thus increased the number of binding sites between particles. The lack of a biochar effect on aggregation in this study may be due to many factors, including high variability within and across sites, as well as the low aggregate-forming clay content of these soils (Table 1).

In addition to the direct effects of amendments on soil properties, the vegetation seeded into the plots during experiment establishment may also be playing an indirect role in facilitating changes to soil physical properties. Working in these same plots, (Ramlow et al., 2017) found higher grass cover in BC-BS and BC-MU treatments relative to other treatments. Increased rooting activity can help reduce soil compaction by loosening of the soil, the creation of macropores (Angers & Caron, 1998; Lesturgez et al., 2004) and contributions to aggregation via organic matter inputs or particle enmeshment (Angers & Caron, 1998; Fonte et al, 2012; Oades, 1993). A number of studies have suggested that biochar can improve plant growth (Lehmann et al., 2011, and others), and this may occur through a variety of mechanisms, including changing nutrient and water dynamics, and biological activity in soils (Basso et al., 2013; Rutigliano et al., 2014; Xiao et al., 2016). Given that I did not see significant effects on bulk density in plots that received biochar alone, this suggests that other amendments may have contributed to improved soil and plant growth processes. Mulch, for example, can help retain soil moisture and protect the soil surface from raindrop impact, thus helping to maintain soil structure, protect juvenile plants, and reducing soil erosion (Luce, 1997; Madej, 2001). Biosol was also surface applied (albeit at a much lower rate) and may have had similar effects as the mulch. At the same time, Biosol may be more important for contributing nutrients necessary for plant growth, and this in combination with the positive effects of biochar appeared to have supported improved plant growth and reduced bulk density. This idea broadly agrees with the findings of Mete et. al. (2015), who reported a synergistic effect of biochar and NPK fertilizers on soybean growth and yield. While not measured in this study, it seems that plants are likely to have played an important role in the observed changes to soil physical properties.

2.4.2 Implications for water dynamics and erosion

Treatment comparisons of water movement and storage parameters indicated only a marginally significant effect of amendments on sediment produced in runoff, such that the control yielded the highest amount of eroded sediments (Table 2).

While treatment differences were not always clear, I observed many significant correlations between soil structural properties and water dynamics. For example, I found a strong negative correlation between aggregate stability (MWD) and the proportion of simulated rainfall occurring as runoff (Fig. 2) and a similarly strong positive correlation between aggregate stability and infiltration rate (Fig 3). I also found MWD to be negatively correlated with sediment production in the runoff water (Fig. 5), confirming that increased aggregate stability is a signal of improved soil health (and hydrologic function). The relationships I observed between MWD and runoff proportion and infiltration rate were largely predictable, since water stable aggregates, by definition, resist slaking and rupture under saturated conditions (Elliott, 1986).

Broken aggregates can clog soil macropores and reduce overall porosity, thus inhibiting infiltration rates. Others have suggested aggregate stability to be an important determinant of infiltration and erosion potential (Barthès & Roose, 2002). Based on the strong correlations between aggregation and water dynamics observed here and in other studies, I feel that MWD could be a valuable indicator for assessing the success of restoration measures on soil erosion and related processes of surface water movement.

I suspect that high variability associated with the heterogeneous nature of this mountain landscape may have masked differences between treatments. Despite these challenges, the numerous significant correlations and general trends discussed above for amended treatments vs. the control suggest that the soil amendments can have important impacts on soil structure, and this in turn has critical implications for soil water dynamics and erosion. It is also worth noting that vegetation > 2 cm in height was removed under the infiltrometers to avoid interference with the drip emitters. Another possible explanation for a general lack of clear difference between treatments may be related to the relatively coarse texture of the soils at these field sites, with sand content over 65% in nearly all plots. Soil texture influences its response to raindrop impact and surface flow with respect to erosion. The clay and, to a lesser extent, silt fractions are important for soil aggregation, which is clearly important for soil physical and hydrologic properties. Rainfall simulations by Ekwue & Harrilal (2010) found that, of the various measured properties (soil texture, compaction, slope, and peat content), texture had the greatest effect on erosion. They also reported that sandy soils (> 60% sand) produced more erosion than clayey soils (< 45% sand), suggesting that the lack of clay particles decreased soil cohesion. Similarly, soil response to amendment additions depends upon texture. Aggregation responds to biochar amendments differently depending on soil texture, with less influence in coarse-textured soils.

Liu et al. (2012) reported increased aggregation of loamy soils, but not in sandy soils in an eleven-month incubation study. Wang et al. (2017) reported similar findings in a study of biochar's effect on soil organic matter; biochar increased aggregate stability and physically protected soil organic matter in aggregates, but this effect was stronger in fine-textured soils relative to a sandier soil.

2.4.3 Applications in road restoration

Restoration of unsealed roads should occur as quickly as possible after closure to minimize their harm to local and regional ecosystems, especially since their potential temporal and spatial ecological effects can be extensive without remediation (Beschta, 1978; Krauss et al., 2010; Pocock & Lawrence, 2005). Decompaction as a first step is usually beneficial since it generates immediate effects including enhancing soil water uptake, reducing runoff, and improving the ability of flora and fauna to colonize the site. Reducing soil bulk density should result in prompt improvements in soil water dynamics and promote a range of soil functions. Iverson et al. (1981) noted that vehicle traffic increased bulk density by collapsing the largest pores in road soils, with a consequent reduction in infiltration, increased runoff, and increased erosion observed in rainfall simulations on arid loamy sand soils. Reducing bulk density, especially by methods that simultaneously increase aggregation, could improve soil and ecosystem health by improving access to water for plants and other soil biota, reducing runoff burden on waterways, and reducing sediment loads harmful to aquatic life. Biochar, when combined with additional soil amendments, was shown here to reduce soil bulk density, suggesting these amendment combinations may be appropriate treatments to decompact unsealed forest roads.

The findings presented here suggest that water-stable aggregation (MWD) is a useful indicator of soil health and structural integrity due to its role in many physical and biological processes critical to overall ecosystem function. While not showing clear differences between treatments in this study, MWD was positively correlated with infiltration rate, and both soil erosion and runoff correlated negatively with MWD, thus exemplifying its role in regulating soil water movement and erosion dynamics. Given that MWD is relatively easier to measure than water movement and erosion, MWD could provide a useful indicator of restoration impacts on soil hydrologic properties. This also suggests that restoration treatments that improve aggregation should be considered when selecting soil restoration methods. The findings from this work suggest biochar influences soil structural properties (bulk density), but that its effects are not always consistent and depend upon soil properties such as texture.

Feedstock materials and pyrolysis parameters influence biochar chemical and physical properties, and therefore, its performance in soil and its effect on the biotic community (Ameloot et al, 2013; Elzobair et al., 2016; Jiang et al., 2015; Kelly et al., 2015). Application rate effects the performance of soil amendments like mulch, biochar, and composts, so in scenarios requiring large quantities of amendments such as an extensive road decommissioning project, there exists the possibility that prohibitively large quantities may be necessary to achieve the desired restoration outcome. Practically any organic material can be pyrolyzed to produce biochar and similarly mulch can be produced from a wide range of materials. These results show that soil amendments produced from locally-available pine biomass can be potentially utilized to reduce purchasing, manufacturing, and transportation costs, especially in scenarios requiring large volumes of material. Soil amendments may be prohibitively expensive or difficult to acquire in large-enough quantities for extensive road decommissioning projects, but my results show that abundant, locally-available forest detritus is an attractive feedstock to produce both biochar and mulch.

Given the results of this study and similar findings from the literature I observe that biochar is a potentially valuable soil amendment in restoration scenarios, but the wildly-varying physio-chemical properties of biochar reported in the literature means that careful consideration must be given to its properties and the potential interactions with soil and soil biota. Biochar has the potential to ameliorate soil structural and chemical deficiencies and can enhance the effects of other amendments, but it can also increase the toxicity of soils depending upon their conditions, or have no effect due to soil physical properties like extremely sandy texture. Land managers should consider using biochar only after identifying restoration goals (e.g. reduce bulk density and establish grass cover) and understanding a biochar's effect on properties relevant to those goals. If ample feedstock is locally-available and its properties when pyrolyzed are favorable, it can be a convenient amendment for use in road restoration projects.

The heterogeneous nature of soil likewise results in spatial variability, which must be accounted for when planning a road restoration project. Despite high variability in these sites, my correlation analysis revealed that improving soil properties like water-stable aggregate MWD and compaction are likely important even in variable soil conditions and should be considered key parameters to improve in road restoration scenarios. Efficient road restoration plans should thus include pre-treatment soil analyses to determine soil physical properties of restoration interest, then select soil amendments known to improve those properties.

This research suggests biochar may have had an effect on road soil properties other than bulk density, but variability and the short time frame of the study likely masked effects. Thus, a future experiment better-accounting for soil heterogeneity could reveal additional effects of biochar in these environments. Future research opportunities also exist regarding the long-term impact of biochar in road restoration scenarios, since I found scant long-term research in the literature.

2.4.4 Conclusions

These findings suggest that key soil amendment combinations reduced bulk density of sandy soils when compared to unamended controls; biochar, when combined with either mulch or Biosol, reduced bulk density of decommissioned forest roads soils. The mechanism(s) by which this occurred are not clear, but are likely due to synergistic effects between biochar and organic matter additions that encourage plant growth and soil aggregation. The correlations that I observed align with the results of similar studies in both road and non-road contexts, suggesting some of the properties I studied (aggregation in particular) are relevant to successful road restoration outcomes in sandy mountain road soils. Therefore, biochar + organic matter soil amendments may be leveraged to promote enhanced positive outcomes in restoration projects when combined with conventional methods like ripping and seeding. While its utility is promising, the variability of biochar due to parent material, pyrolysis techniques, and application rates needs to be taken into consideration. Restoration success depends upon restoration goals, inherent soil properties, particularly texture, as well as the nature and availability of soil amendments. The abundance of locally-available feedstock suggests that biochar is an attractive and potentially viable option for road restoration in this setting; however, complementary amendments to provide nutrients and/or soil cover should be considered as well.

2.5 Tables and figures

Table 1: General site information for road restoration study sites in the Roosevelt National Forest, Colorado. Texture information determined from samples collected on-site in 2016. pH data collected on-site after treatment application in 2014 (unpublished data). General on-site observations recorded over the course of the study were reviewed and used for the “Recreational Disturbance” table. Soil Survey Mapping Unit information from USDA Web Soil Survey, and Soil Survey of Larimer County Area, Colorado (1980).

Site	S1: Sevenmile Creek	S2: Downed Trees	S3: Loop
Location	40° 42’ 54.16” N, 105° 36’ 0.97” W	40° 43’ 33.74” N, 105° 36’ 2.74” W	40° 44’ 58.45” N, 105° 36’ 53.33” W
Aspect	southerly	southeasterly	southwesterly
Elevation (m)	2,331	2,641	2,667
sand / silt / clay (%)	77 / 11 / 12	72 / 19 / 9	75 / 15 / 10
Average pH	8.0	6.8	6.2
Soil Survey Mapping Unit	Breece coarse sandy loam, 3 to 9 % slopes	Wetmore-Boyle-Rock outcrop complex, 5 to 60 % slopes	Breece coarse sandy loam, 3 to 9 % slopes

Table 2: Means of selected soil properties (0-10 cm) measured in June 2016 from soil amendment treatments applied across three soil restoration sites located within the Roosevelt National Forest, Colorado. Treatments include control (C), biochar (BC), Biosol (BS), wood straw mulch (MU), biochar + Biosol (BC + BS), biochar + mulch (BC + MU), and Biosol + mulch (BS + MU). Standard errors are presented in parentheses. ANOVA p-values presented at end.

Treatment	TTR (s)[1]	RO (%)[2]	SED (g)[3]	% Sand	% Clay	Available Water	GWC[4]	MWD[5] (µm)
C	165 (41)	63.9 (9.6)	1.35 (0.15)	74 (3)	10 (2)	0.38 (0.01)	0.03 (0.01)	812 (104)
BC	582 (168)	40.3 (10.5)	0.55 (0.10)	79 (4)	7 (1)	0.37 (0.02)	0.03 (0.01)	823 (16)
BS	349 (3)	63.1 (8.0)	0.98 (0.47)	76 (1)	11 (2)	0.38 (0.01)	0.04 (0.01)	832 (39)
MU	615 (367)	37.6 (14.1)	0.56 (0.12)	74 (3)	10 (1)	0.37 (0.02)	0.04 (0.01)	1079 (181)
BC + BS	355 (71)	58.4 (12.3)	0.89 (0.14)	69 (4)	14 (3)	0.39 (0.03)	0.07 (0.03)	941 (120)
BC + MU	801 (481)	38.1 (23.5)	0.58 (0.30)	80 (2)	9 (1)	0.43 (0.01)	0.06 (0.02)	904 (52)
BS + MU	1085 (719)	27.1 (21.2)	0.63 (0.33)	72 (4)	13 (1)	0.35 (0.01)	0.05 (0.01)	1024 (70)
p-value	0.268	0.701	0.478	0.541	0.535	0.241	0.155	0.622

1: TTR, Time to Runoff; 2: RO, Runoff Proportion; 3: SED, Sediment Production; 4: GWC, Gravimetric Water Content; 5: MWD, Mean Weight Diameter.

Table 3: Pearson's correlation coefficients (r values) of observed soil properties under the various examined treatments.

	TTR	RO	SED	Infiltration Rate	% Sand	% Clay	Available Water	GWC	MWD	Bulk Density
TTR	1									
RO	-0.72 ***	1								
SED	-0.66 ***	0.87 ***	1							
Infiltration Rate	0.72 ***	-0.98 ***	-0.84 ***	1						
% Sand	0.14	-0.38	-0.45 *	0.32	1					
% Clay	0.16	0.12	0.26	-0.06	-0.73***	1				
Available Water	0.34	-0.24	-0.16	0.21	0.29	0.10	1			
GWC	0.16	-0.06	-0.06	0.13	0.17	0.33	0.53 *	1		
MWD	0.44 [1]	-0.71 ***	-0.59 **	0.73 ***	0.17	0.02	0.33	0.25	1	
Bulk Density	-0.02	0.34	0.37	-0.31	0.01	-0.09	-0.31	-0.10	-0.36	1

Correlations significant at * $p < 0.05$, ** $p < 0.01$, *** $p < 0.005$, [1] $p < 0.06$

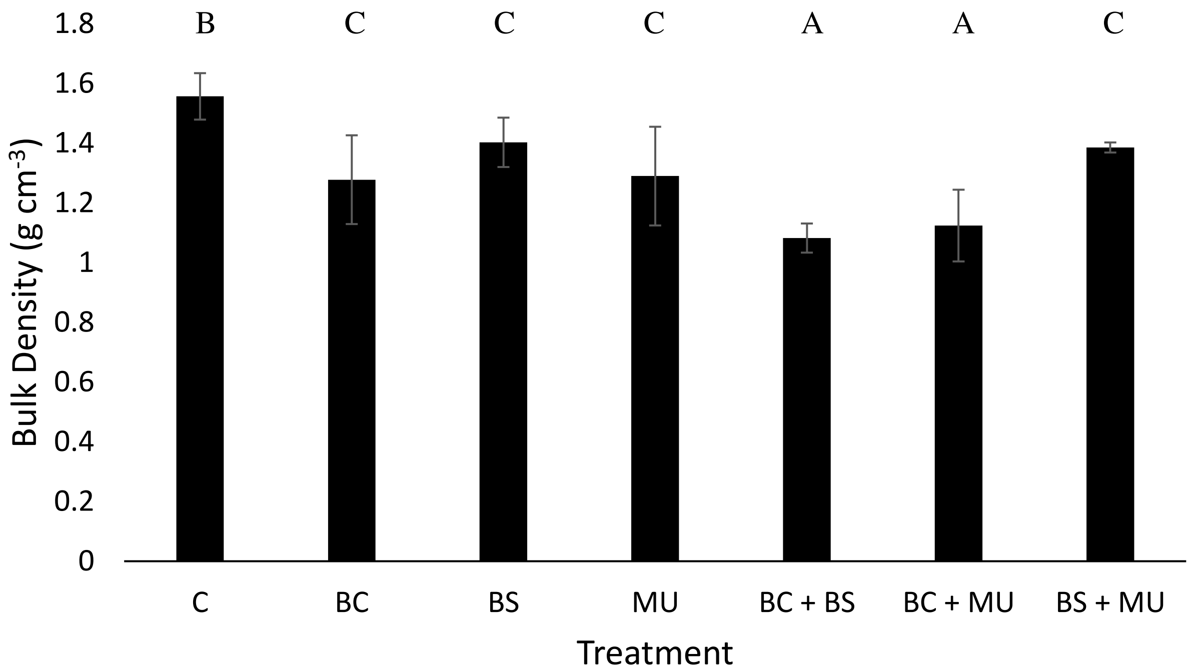

Figure 1: Mean bulk density (0-10 cm) measured in June 2016 from soil amendment treatments applied across three soil restoration sites located within the Roosevelt National Forest, Colorado. Treatments include control (C), biochar (BC), Biosol (BS), wood straw mulch (MU), biochar + Biosol (BC-BS), biochar + mulch (BC-MU), and Biosol + mulch (BS-MU). Error bars represent standard error. Letters above bars indicate groups of statistical similarity ($p < 0.05$) using Tukey-adjusted pairwise comparisons ($n = 3$).

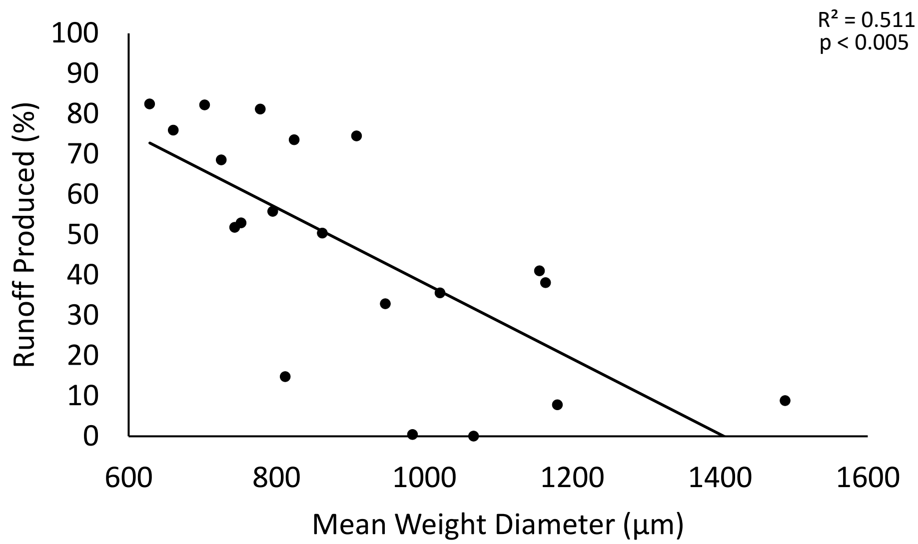

Figure 2: Relationship between soil aggregate stability (mean weight diameter) and the proportion of simulated rainfall occurring as runoff in soils across three soil restoration sites located within the Roosevelt National Forest, Colorado.

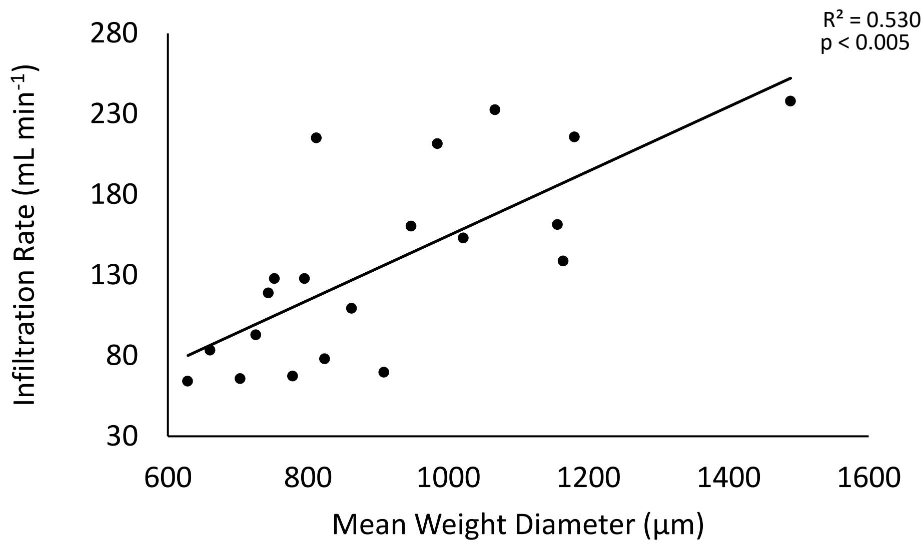

Figure 3: Relationship between soil aggregate stability (mean weight diameter) and infiltration rate (mL min^{-1}) of simulated rainfall in soils across three soil restoration sites located within the Roosevelt National Forest, Colorado.

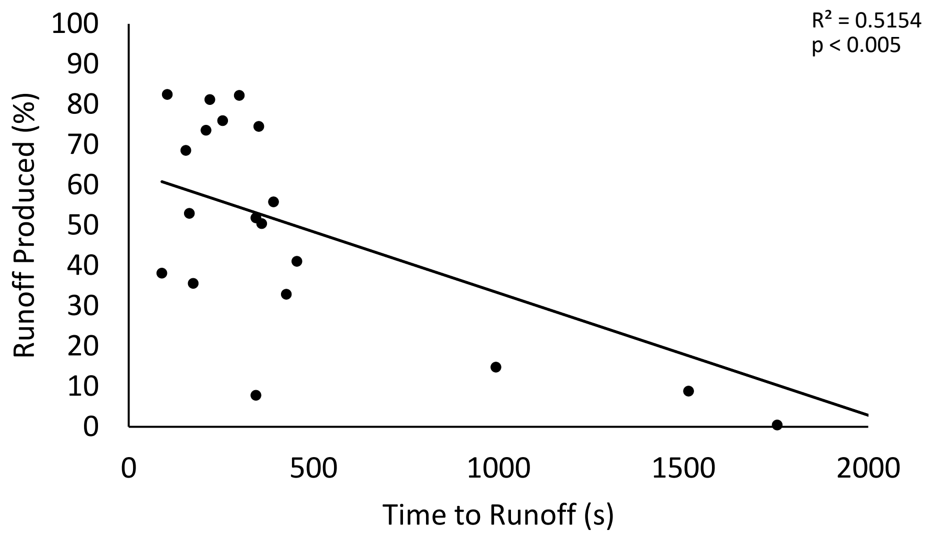

Figure 4: Relationship between the time to runoff (s) after precipitation begins versus proportion of simulated rainfall that does not infiltrate in soils across three soil restoration sites located within the Roosevelt National Forest, Colorado.

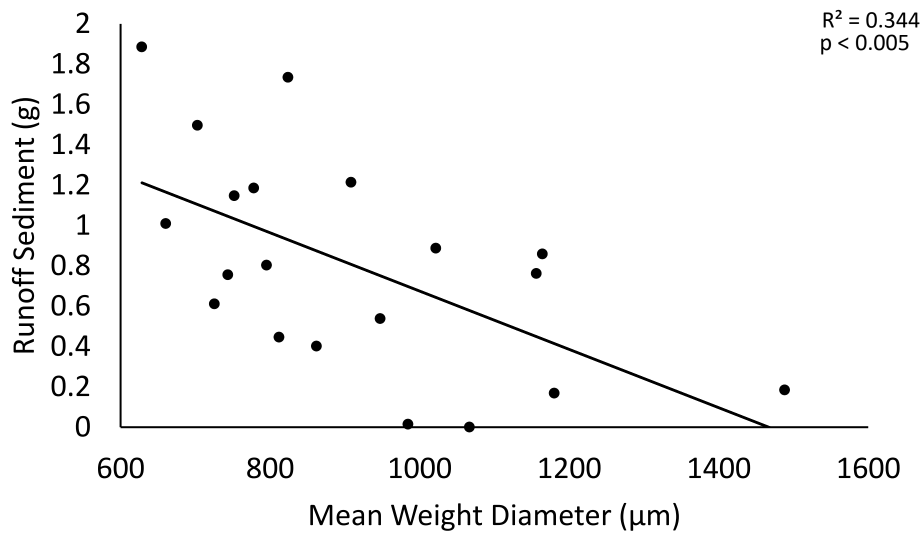

Figure 5: Relationship between aggregate stability (mean weight diameter) vs. the mass of sediment (g) observed in runoff in soils across three soil restoration sites located within the Roosevelt National Forest, Colorado.

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