Center for Environmentally Sustainable Transportation in Cold Climates

Evaluating the Potential Effects of Deicing Salts on Roadside Carbon Sequestration

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13. ABSTRACT

This project sought to document patterns of road deicing salts and the effects of these salts on the amount of carbon being sequestered passively along Montana Department of Transportation roads; it was designed collaboratively with a related roadside project that tested three different highway right-of-way management techniques (mowing height, shrub planting, disturbance) to determine whether they have the capacity to increase soil organic carbon.

Our sampling did not reveal elevated salt levels at any of the nine locations sampled at each of the three I-90 sites. The greatest saline concentrations were found at the sample locations farthest from the road. This pattern was consistent across all three sites. The range of soil organic matter (SOM) was broad, from ~1% to >10%. Generally, SOM values were lowest adjacent to the road and highest farthest from the road. We found no or weak evidence of a relationship between our indices of soil salinity and SOM levels, with electrical conductivity, exchangeable calcium, and cation exchange capacity. Results imply that if road deicing salts are altering patterns of roadside SOM and potential carbon sequestration, this effect was not captured by our experimental design, nor did deicing salts appear to have affected roadside vegetation during our most recent sampling effort.

Our findings highlight the value of experimentally separating the multiple potentially confounding effects of winter maintenance operations on roadside soils: roads could focus the flow of water, salts, and sands to roadside soils. How these types of mass inputs to roadside soils might influence medium- or long-term carbon dynamics remains an open question, but their fuller characterization and possible flow paths will be essential to clarifying the role of roadside soils in terrestrial soil organic carbon sequestration strategies.

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METRIC (SI*) CONVERSION FACTORS

APPROXIMATE CONVERSIONS TO SI UNITS

Symbol	When You Know	Multiply By	To Find	Symbol
		LENGTH		
in	inches	25.4	millimeters	mm
ft	feet	0.3048	meters	m
yd	yards	0.914	meters	m
mi	Miles (statute)	1.61	kilometers	km
		AREA		
in^2	square inches	645.2	millimeters squared	cm^2
ft^2	square feet	0.0929	meters squared	m^2
yd^2	square yards	0.836	meters squared	m^2
mi^2	square miles	2.59	kilometers squared	km^2
ac	acres	0.4046	hectares	ha
		MASS (weight)		
oz	Ounces (avdp)	28.35	grams	g
lb	Pounds (avdp)	0.454	kilograms	kg
T	Short tons (2000 lb)	0.907	megagrams	mg
		VOLUME		
fl oz	fluid ounces (US)	29.57	milliliters	mL
gal	Gallons (liq)	3.785	liters	liters
ft^3	cubic feet	0.0283	meters cubed	m^3
yd^3	cubic yards	0.765	meters cubed	m^3

Note: Volumes greater than 1000 L shall be shown in m^3

Symbol	When You Know	Multiply By	To Find	Symbol
		TEMPERATURE (exact)		
ºF	Fahrenheit temperature	5/9 (ºF-32)	Celsius temperature	ºC
		ILLUMINATION		
fc	Foot-candles	10.76	lux	lx
fl	foot-lamberts	3.426	candela/m^2	cd/cm^2
		FORCE and PRESSURE or STRESS		
lbf	pound-force	4.45	newtons	N
psi	pound-force per square inch	6.89	kilopascals	kPa

These factors conform to the requirement of FHWA Order 5190.1A *SI is the symbol for the International System of Measurements

APPROXIMATE CONVERSIONS FROM SI UNITS

Symbol	When You Know	Multiply By	To Find	Symbol
		LENGTH		
mm	millimeters	0.039	inches	in
m	meters	3.28	feet	ft
m	meters	1.09	yards	yd
km	kilometers	0.621	Miles (statute)	mi
		AREA		
mm^2	millimeters squared	0.0016	square inches	in^2
m^2	meters squared	10.764	square feet	ft^2
km^2	kilometers squared	0.39	square miles	mi^2
ha	hectares (10,000 m^2)	2.471	acres	ac
		MASS (weight)		
g	grams	0.0353	Ounces (avdp)	oz
kg	kilograms	2.205	Pounds (avdp)	lb
mg	megagrams (1000 kg)	1.103	short tons	T
		VOLUME		
mL	milliliters	0.034	fluid ounces (US)	fl oz
liters	liters	0.264	Gallons (liq)	gal
m^3	meters cubed	35.315	cubic feet	ft^3
m^3	meters cubed	1.308	cubic yards	yd^3

Symbol	When You Know	Multiply By	To Find	Symbol
		TEMPERATURE (exact)		
ºC	Celsius temperature	9/5 ºC+32	Fahrenheit temperature	ºF
		ILLUMINATION		
lx	lux	0.0929	foot-candles	fc
cd/cm^2	candela/m^2	0.2919	foot-lamberts	fl
		FORCE and PRESSURE or STRESS		
N	newtons	0.225	pound-force	lbf
kPa	kilopascals	0.145	pound-force per square inch	psi

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

EXECUTIVE SUMMARY

The transportation sector produces the greatest emissions of the greenhouse gas carbon

dioxide (CO_2) of any sector in the United States. Two key strategies to reduce the atmospheric

carbon (C) concentration caused by anthropogenic emissions include capturing and storing CO_2

in either the earth's biosphere (terrestrial soil organic carbon sequestration, TSOCS) or in

geologic formations deep underground (geologic sequestration). The main focus of terrestrial

sequestration is soil organic carbon (SOC), since soils hold a greater mass of carbon than the

earth's atmosphere and forests combined. The transportation sector, particularly in snowy

regions, also uses deicing salts for winter maintenance operations. These salt additions could

contribute to the area of salt-affected soils, which is nearly 1 billion hectares globally (Wong et

al. 2010). The extent to which the long-term addition of salts to roadside soils might affect soil

carbon dynamics in these settings remains relatively unexplored.

This project sought to document the potential effects of road deicing salts on the amount

of carbon being sequestered passively along Montana Department of Transportation roads. The

project was designed collaboratively with a related roadside project that tested three different

highway right-of-way management techniques (mowing height, shrub planting, disturbance) to

determine whether they have the capacity to increase SOC.

Our field study did not reveal elevated salt levels at any of the nine locations sampled

(three distances [0 m, 1 m, and ~13–17 m composite samples] at each of the three Interstate

Highway-90 [I-90] sites). The highest soil salt level observed (as indexed by electrical

conductivity in a 1:1 soil:water extract) was 0.21 deciSiemens per meter (dS/m), or

approximately $1/20^{th}$ the threshold typically regarded as indicative of "saline" soils (4 dS/m;

Mavi and Marschner 2017), at a sample location farthest from the road. This pattern was

consistent across all three sites (3F, BC, and BP), with average EC values ($\pm$1SD, n=3) for the sample points farthest from the road typically triple the EC values for the sample points closest to the road (0.17$\pm$0.04 vs. 0.05$\pm$0.01 dS/m).

Other indices of salinity such as exchangeable calcium and sodium were broadly consistent with this trend. Over the same sample locations (0 m, 1 m, and ~15 m), cation exchange capacity (CEC), which generally reflects the abilities of negatively charged colloids such as clays and organic matter to retain and exchange cations, shifted from 6.7 to 8.3 to 19.1 cmol_+/kg. This pattern is largely consistent with a shift from soils under the strong influence of sand-sized (typically, 0.05–2 mm) particles, perhaps due to either highway construction or winter maintenance aggregate (sand) applications, which have close to 0 CEC. For reference purposes, at the Arthur H. Post Farm just 8 km southwest of I-90 between Bozeman and Belgrade (re: between transect locations), EC levels ranged from 0.05–0.11 dS/m in the upper 30 cm, while CEC values averaged ~18 cmol_+/kg (Romero et al. 2017).

The range of soil organic matter (SOM) was broad, from ~1 to >10%. In general, these values were lowest adjacent to the road and highest farthest from the road, though this finding could also reflect the soil dilution effect caused by sands used during winter maintenance operations and concentrated in samples closest to (0 m) the road, and which have essentially 0% SOM.

We found no or weak evidence of a relationship between our indices of soil salinity and SOM levels, with EC, exchangeable calcium, and cation exchange capacity all having R^2 values of <0.5 with respect to associated SOM. These R^2 values were less than 0.07, with the exclusion of a single sample that showed relatively elevated EC (0.21 dS/m), exchangeable calcium (2700 mg/kg), and cation exchange capacity (18.1 cmol_+/kg). The Arthur H. Post Farm showed SOC

levels of 13.1 (0–10 cm) to 7.8 (20–30 cm) g/kg (Romero et al. 2017), which amount to an estimated 2.6% to 1.6% SOM, assuming SOM is double SOC.

Taken together, our results imply that if road deicing salts are altering patterns of roadside SOM and potential carbon sequestration, this effect was not captured by our experimental design, nor did deicing salts appear to affect roadside vegetation during our most recent sampling effort in late spring/early summer 2018 (Ament et al. 2018). Evidence from the literature (e.g., Wong et al. 2010, Amini et al. 2016, and Mavi and Marschner 2017) points to the possibility of salts, such as road deicing salts, to affect soil carbon dynamics, but clarifying those types of effects in Montana will require experimental additions of deicing salts to roadside soils, together with long-term monitoring of soil processes at treated and untreated plots.

Our findings highlight the value of experimentally separating the multiple potentially confounding effects of winter maintenance operations and management practices on roadside soils. Roads could focus the flow of water, salts, and sands to roadside soils. How these types of mass inputs to roadside soils might influence medium- or long-term carbon dynamics remains an open question, but their fuller characterization and possible flow paths will be essential information for clarifying the potential roles of roadside soils in TSOCS strategies.

CHAPTER 1.0   INTRODUCTION

Across North America, snow- and ice-control products such as deicers are used during winter highway maintenance operations to cost-effectively improve safety and increase mobility for the driving public. Deicers work by lowering the freezing point of water to aid in the removal of snow and ice from roadways. Commonly used deicers include sodium chloride and magnesium chloride, as liquids and solids, but may also include calcium chloride, agriculturally derived co-products (e.g., beet juice), acetates, formates, and glycols or glycerols. Following application of deicers to highways, deicers can run off or be plowed off or splashed off the road onto roadside right-of-ways (ROWs). Once off the road, deicers can impact plants, soil, ground and surface water, and aquatic and terrestrial fauna.

A professional paper by Cote (2015) and an associated Center for Environmentally Sustainable Transportation in Cold Climates (CESTiCC) funded project (*Exploring management options to increase roadside carbon storage*) investigated the use of experimentally manipulated roadside soils and vegetation for enhanced carbon sequestration in the highway ROW. Because deicing salts can impact roadside vegetation and soils (Černohlávková et al. 2008), and salts have shown mixed results on carbon storage in soils (e.g., salinity increasing carbon storage [Olsen et al. 1996] and salinity decreasing carbon storage [Lal 2001]), one of the goals of this project was to assess whether chloride-based snow- and ice-control products impact potential carbon sequestration rates in the near-road environment, and if so, the magnitude of this influence. If road deicers reduce carbon sequestration rates, large swaths of highway ROWs may be unsuitable for carbon sequestration purposes (Cote 2015).

According to past research, most deicing salts sprayed off the roadway land within 100 ft (30.5 m) of the road edge, covering a large extent, if not all, of the ROW. Additionally, highway

agencies have estimated that 5% to 10% of plants within 100 ft (30.5 m) of the road edge in these high-use sections are affected by deicers (Viskari and Karenlampi 2000, Bryson and Barker 2002). Our goals were to (i) quantify deicer application rates, (ii) measure cation and anion concentrations of deicers used by the Montana Department of Transportation (MDT) in roadside soils (specifically but not limited to sodium and chloride), and (iii) assess soil organic carbon in the same roadside soils, to determine if winter maintenance operations affect carbon sequestration potential in the ROW.

CHAPTER 2.0   BACKGROUND

2.1 Roadway Deicers

Roadway deicers can have many impacts on the near-road environment, as shown in Figure 1 (Fay et al., 2015). Since the 1960s, there has been a steady increase in the use of chloride roadway deicers for winter maintenance operations, along with an awareness of and research into the impacts from their use (Hawkins 1971, Roth and Wall 1976, Paschka et al. 1999, Ramakrishna and Viraraghavan 2005). Chloride salts, primarily sodium chloride (NaCl), magnesium chloride ($MgCl_2$), and calcium chloride ($CaCl_2$), are the most commonly used freezing-point depressants in snow and ice control because of their relative low cost, ease of use, and generally low risk for applicators and road users. In 2017, the United States used ~19 million tons of NaCl in winter maintenance operations (USGS, 2018).
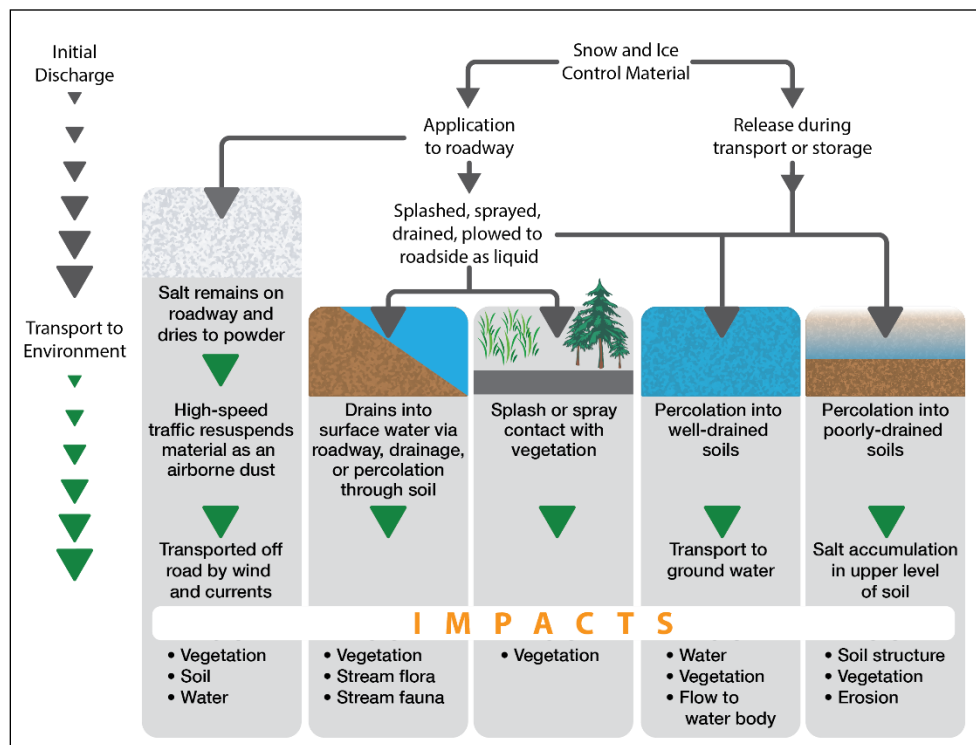
Figure 1. Flow chart showing the transport of snow- and ice-control products off the road, the environments affected, and potential impacts.

Chloride ions (Cl⁻) are conservative, which means that once dissolved they do not degrade in the environment, but instead remain in solution (Ministry of Environment 2011). The environmental impacts of chloride roadway deicers vary based on factors unique to each formulation and the location of application. According to Ramakrishna and Viraraghavan (2005, pg. 54), the degree and distribution of chloride-based deicer impacts in the highway environment are defined by spatial and temporal factors, such as draining characteristics of the road and adjacent soils, amount and timing of materials applied, "topography, discharge of the receiving stream, degree of urbanization of the watershed, temperature, and precipitation," and dilution and adsorption into soil. Recent research confirms that repeated applications of chloride deicers may adversely affect the surrounding soil and vegetation, water bodies, aquatic biota, and wildlife (Buckler and Granato 1999, Venner Consulting and Parsons Brinckerhoff 2004, Levelton Consultants Ltd. 2007). Not all chloride in ecosystems originates from deicer use, of course, and approaches have been developed to quantify sources of chloride to the environment (Sherwood 1989, Mullaney et al. 2009).

Aerial drift of deicers resulting from vehicular splash, plowing, and wind has also been observed to impact vegetation adjacent to roadways. Nicholson and Branson (1990) showed that deicer particulates deposited on the road could be removed and re-suspended by vehicular traffic. Wet conditions, increased vehicle speed, wind currents, and updrafts generated by vehicular traffic can cause redistribution of deicers off the roadway into the adjacent environment (Kelsey and Hootman 1992). Deicing particles have been shown to travel up to 130 ft and have been observed up to 330 ft from the roadway (Lumis et al. 1973, Blomqvist and Johansson 1999, Trahan and Peterson 2007). Kelsey and Hootman (1992) observed sodium deposition within 400 ft of a toll way and sodium-related plant damage within 1,240 ft of the toll way. Field tests have

shown that 20% to 63% of the NaCl-based deicers applied to highways in Sweden were carried through the air, with 90% of them deposited within 65 ft of the roadside (Blomqvist and Johansson 1999). Given that typical ROWs on Class 1 highways are 75 ft wide (FHWA 2015), about 80% of applied deicers could be redeposited in the ROW.

Deicer migration into soils adjacent to roadways can cause the swelling and compaction of soil, increase electrical conductivity, and lead to loss of soil stability by means of dry-wet cycling, osmotic stress (greater salinity lowers the osmotic potential of soils), and mobilization of nutrients (Environment Canada 2010). Factors that affect the concentration of deicers in soil are the texture of the soil, as well as its water content, cation exchange capacity, permeability, and infiltration capacity (D'Itri 1992). Lundmark and Jansson (2008, pg. 215) modeled "the spread of deicing salt from the road to surroundings, deposition in the roadside environment and the subsequent infiltration into roadside soil." With increasing distance from the road, the field observations confirmed a general decrease in the chloride content of soil, with supporting evidence in soil physical properties, vegetation properties, and snow characteristics.

Common deicer exposure mechanisms to plants include increased concentrations in soil solution that can result in uptake by plant roots, or accumulation on foliage and branches from splash and spray (TRB Special Report 235 1991). Hanes et al. (1970) described the three major effects of salt on plant growth. First, salt can increase soil salinity and alter the osmotic pressure gradient, inhibiting the uptake of water by plant roots. Second, salt accumulations can occur in plant tissues. Third, specific salts can induce ionic imbalances, causing plant injury symptoms such as desiccation and leaf burn. Deicing salt exposure resulting from spray within 33 to 65 ft of the road was demonstrated to cause a greater severity of foliar damage than uptake through the soil alone (Hofstra and Hall 1971, Viskari and Karenlampi 2000, Bryson and Barker 2002). On

primary highways within 100 ft of the road, highway agencies estimate that 5% to 10% of the

plants in high-use sections are affected by deicers, and report that shrubs and grasses can tolerate

increased concentrations better than trees (Viskari and Karenlampi 2000, Bryson and Barker

2002). Plants with broader leaves are generally affected more than plants with narrow leaves

(TRB 1991). Needle necrosis (death), twig dieback, and bud kill have been associated with areas

of heavy deicing salt usage, with foliage downwind from and facing roads more heavily affected

than foliage upwind from or facing away from roads (Hofstra and Hall 1971, Lumis et al. 1973,

Sucoff et al. 1976, Pedersen et al. 2000). The slope of the roadside adjacent to the treated

roadway is also an important variable in defining the extent of plant injury from deicer treatment,

with vegetation showing effects up to 20 ft away on flat surfaces, 40 to 55 ft away for steep

downslopes, but only 10 ft for steep upslopes (TRB 1991).

2.2 Soil Organic Carbon

Global anthropogenic emissions of carbon dioxide (CO_2) totaled 32.5 billion tons

(gigatons [Gt]) in 2017, with the U.S. contributing about 15% of that total (IEA 2018,

http://www.globalcarbonproject.org). In the U.S., the transportation sector will soon be

responsible for generating more greenhouse gases (GHGs) than the electricity generation sector.

In 1990, for example, the ratio of emissions from electricity generation to transportation was

1.24 (total emissions ~6.4 Gt CO_2), but by 2017 (total emissions ~6.5 Gt CO_2), that ratio had

dropped to 1.01 (US-EIA 2018, Houser and Marsters 2018). Within the transportation sector,

vehicles on roads are responsible for 84% of these emissions. Between 1990 and 2010, GHG

emissions in the transportation end-use sector increased more in absolute terms than any other

end-use sector (industrial, agriculture, residential, commercial). Thus, it is incumbent on the

transportation sector to explore means of reducing net GHG emissions as best it can to play its

part in the overall national effort to reduce atmospheric CO_2 concentrations through a number of natural (i.e., dissolution in the oceans) and human-driven processes.

Two key strategies to reduce the atmospheric carbon (C) concentration caused by anthropogenic emissions include capturing and storing CO_2 in either the earth's biosphere (terrestrial sequestration) or in geologic formations deep underground (geologic sequestration) (Lal 2008). The main focus of terrestrial sequestration is soil organic carbon (Lal 2010), since soils have more carbon than the atmosphere and forests combined (Donovan 2013).

One relatively unexplored opportunity for terrestrial carbon sequestration is the vegetation and soils along the nations' roads. The nation's transportation infrastructure comprises over 4 million miles (6.4 million km) of public roads in the U.S., managed by various entities including federal, state, county, tribal, and metropolitan agencies. Public roadsides are easily accessible, and state, tribal, and local transportation agencies could benefit in two ways: reduced costs for vegetation management and increased revenue if sequestered carbon were sold on carbon offset markets (Proudfoot et al. 2015).

A recent, complementary CESTiCC project by Ament et al. (2018) outlined three prior estimates of the potential for roadside carbon sequestration by FHWA (2010), Ament et al. (2014), and Harrison (2014). If roadsides along federal highways and federal lands encompass ~7 million hectares (~17 million acres), this area could sequester ~37 million metric tons of CO_2 (~10 million metric tons of C) or <0.001% of annual U.S. emissions.

Though interest in the potential for roadside vegetation and soils to capture and store carbon has grown recently, little is understood about how increasing use of de-icing salts might affect soil organic carbon dynamics. For example, in a recent review, one study (Setia et al. 2011) found that increased EC (salinity) reduced soil respiration rates, which should increase soil

organic carbon storage, while another study (Wong et al. 2010) reported decreased soil organic

carbon storage with increased salinity and sodicity (predominance of sodium).

2.2.1 The Carbon Cycle

The carbon cycle is composed of the fluxes of carbon among five main carbon reservoirs

(listed in decreasing amounts of C: oceanic pool, geologic pool (includes fossil fuels), the soil

pool that includes soil organic carbon (SOC) and soil inorganic carbon (SIC), atmospheric pool,

and the biologic pool (Lal 2008 and Figure 2 [inorganic carbon is not shown]). The soil and

biologic pools combined are commonly termed the terrestrial C pool.
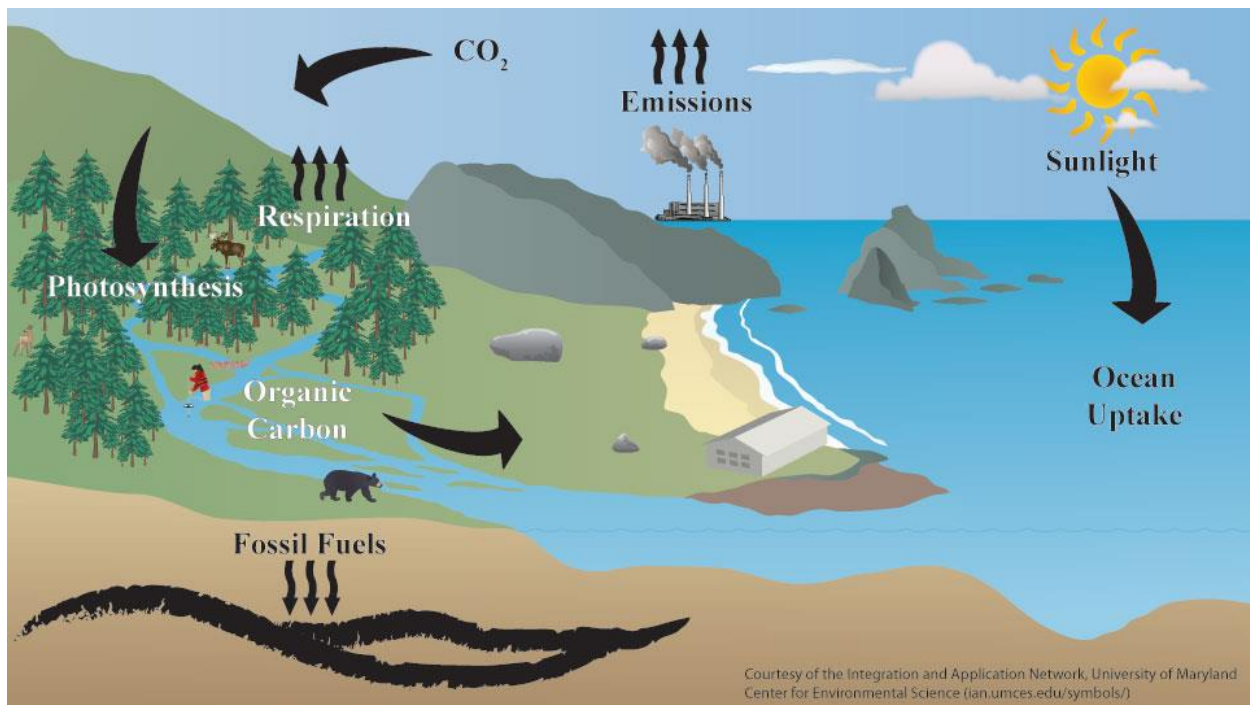
Figure 2. Carbon cycle. Image courtesy of University of Maryland.

The terrestrial carbon cycle (Figure 2 and Figure 3) captures carbon dioxide from the

atmosphere primarily via plant photosynthesis and stores it in the soil through root residues and

excretions, via dead microorganisms, and in wood and other durable plant parts above ground.

Carbon is lost to the atmosphere via plant (autotrophic) and soil microbial (heterotrophic)

respiration of carbohydrate carbon back to carbon dioxide.

Belowground carbon is comprised of both SOC and SIC, though our focus is on SOC. Soil organic carbon is primarily derived from humus, which is decomposed plant and animal matter, also known as soil organic matter (SOM), which is about half SOC, with the remaining mass comprised primarily of structural oxygen. Over half of the carbon in the terrestrial pool resides in soils (Figure 3). As roots and microscopic animals die each year, carbon can accumulate in the soil. Across arid and semiarid landscapes in general, and Montana in particular, SIC can also accumulate as salts such as calcium carbonate buildup. It is not uncommon in some Montana soils to have SIC:SOC ratios >10 in the upper 12 inches of a soil profile (Hartshorn et al. 2017).
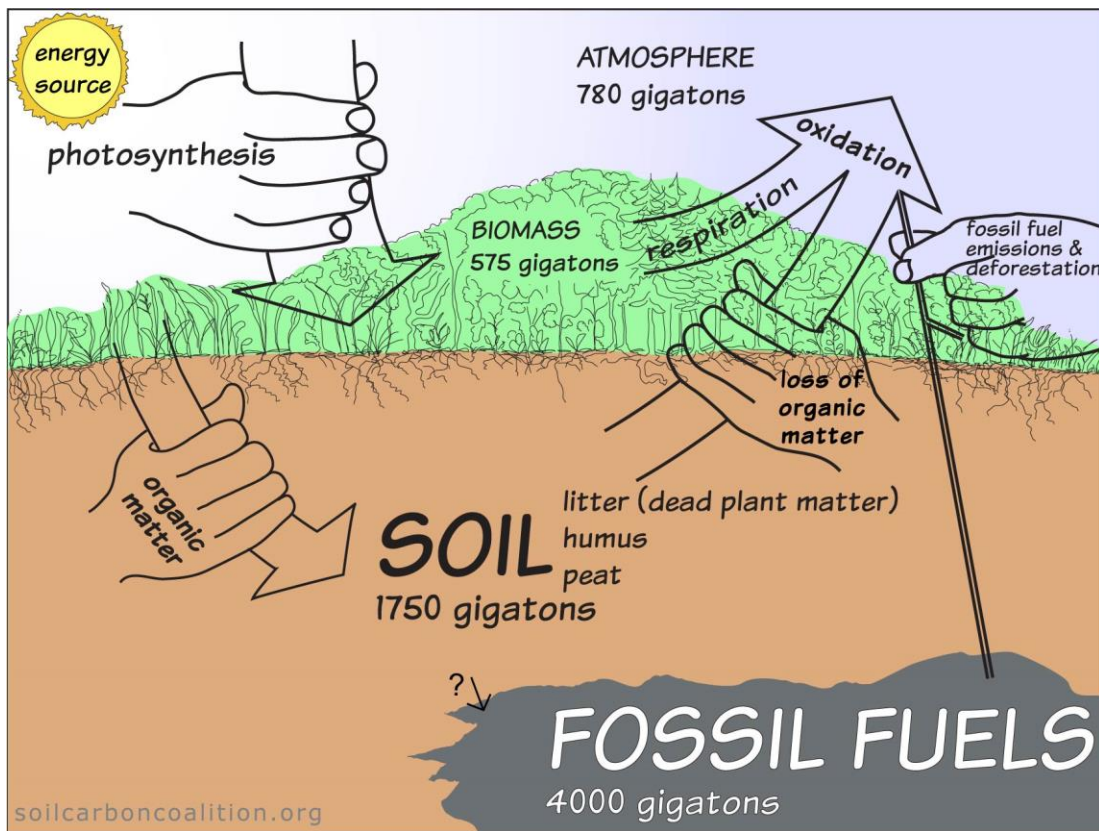
Figure 3. Carbon (not CO_2), distribution in terrestrial ecosystem compartments: soil, fossil fuels, biomass, and the atmosphere. Figure courtesy of the Soil Carbon Coalition.

Figure 3 also shows transfers between compartments as arrows. For reference, 1 gigaton is equivalent to 1,000 million metric tons. Between the hand labelled "loss of organic matter" and the atmosphere indicates where soil respiration measurements were performed for this project, to identify transfers of carbon from soil back to the atmosphere. The net gain or transfer of atmospheric carbon into soils and plants of the terrestrial system is called biological carbon sequestration. Biological carbon sequestration into other pools such as the geologic pool can require millennia; biological carbon sequestration into the oceanic pool occurs on a daily basis. Much of the focus of increasing terrestrial carbon stores is to locate it in soils. Carbon sequestration occurs when SOC increases over time, although some can also be stored above ground, primarily in forests.

Soil organic carbon is a complex mix of organic carbon compounds of biological origin, from plants and animals in various states of decomposition (Xiao 2015); SOC originally comes from atmospheric CO_2 captured by plants through photosynthesis. The amount of SOC is a balance of carbon inputs and carbon losses. For carbon storage, or sequestration (or long-term storage), the goal is to increase the amount of carbon in slow and passive decomposition groups/pools. Amending soil, such as with compost, aids in slowing the decomposition of SOC. Factors that affect carbon sequestration include soil type, initial content of SOC, depth of the soil profile, and climate conditions.

Care should be taken when comparing the SOC levels for a single site over time, or two locations at one time, because as SOC increases relative to mineral soil material (say, from 1% to 2%), this addition of lower-density organic soil material often ends up reducing the density of the soil. Thus, a soil with 10% SOC (~20% SOM) will often be of much lower density (~1 Mg/m^3) than a second soil with only 1% SOC (~2% SOM; ~1.5 Mg/m^3). Best practices for SOC

comparisons include always reporting mass of SOC per unit volume of soil, and ideally, use of an equivalent soil mass (ESM) approach (e.g., Wendt and Hauser 2013). Even ESM approaches to shifts in SOC, however, can lead to errors ranging from -87% to +54% compared with methods that account for simultaneous shifts in both soil volume and soil mass (Sollins and Gregg 2017).

2.3 Road Salts and SOC

Study of the relationship between salt in the soil and carbon sequestration is limited. Specifically, a link between desertification of dryland and emissions of CO_2 from soil and vegetation to the atmosphere has been found (Lal 2001). Desertification or degradation of soil and vegetation has affected ~1 billion ha of soils and ~2.6 billion ha of rangeland, at a rate of 5.8 million ha per year. The main causes are accelerated soil erosion by water and wind and salinization, reduced soil quality and effective root depth, decreased vegetative cover, reduced biomass productivity, and low or variable rainfall (Lal 2001). The consequences of desertification are reduced soil carbon and the transfer of carbon from the soil to the atmosphere (the opposite of sequestration). Lal (2001) notes that restoration of eroded and salt-affected soils can help sequester carbon and recommends identification and implementation of policies to facilitate restoration. While this research provides insight into the potential impacts of salt on carbon sequestration, the findings may not apply outside of drylands.

Work by Lazar (2016) found a link between sodium from deicing salts, irrigation water, and deposition from oceanic aerosols and water extractable dissolved organic carbon (DOC), which could have implications on carbon storage. A preliminary literature search was unable to find any studies that have looked directly at the effects of deicing salts used for winter maintenance on ROW carbon sequestration rates or impacts from salts on SOC, though an intuitive conceptual model would be that, as salinity is increased in roadside soils, primary plant

14

productivity (and therefore carbon sequestration) should be reduced. To our knowledge, no empirical evidence of threshold salinity concentrations on soil carbon dynamics has been reported.

According to the FHWA, in the past, U.S. DOT highway ROWs were typically 50 to 75 ft wide; for new multi-lane highways, ROWs are estimated to be 150 to 300 ft (FHWA 2015). This is a large area (~3.4 million unpaved acres [FHWA 2010]) that could be used for carbon sequestration (Cote 2015). Along Montana's four road types (National Highway System [NHS] Interstate, NHS non-interstate, primary, and secondary), ~166,000 acres are used for ROWs. Nearly half of this area (75,000 acres) has a land cover designation of "plains grassland," one of six different "Montana climax vegetation ecosystem types" associated with different rates of net ecosystem exchange (NEE), a measure of net carbon exchange between an ecosystem's plants and soils and the atmosphere (Ament et al. 2014). The Montana vegetation ecosystem type with the greatest NEE is "riparian," with an estimated (average±1 standard deviation) NEE of 0.9±1.0 metric tons of carbon per acre per year. However, Cote (2015) only identified ~7,000 ROW acres mapped with this vegetation type, and it would be unrealistic to expect much of Montana's semiarid ROW acreage to ever mimic these sequestration rates. Two of Montana's vegetation types shared the lowest NEE: intermountain and Plains grasslands, each with an estimated NEE of 0.4±0.5 Mg C acre^{-1} yr^{-1}. Cote (2015) estimated the extent of ROW mapped as intermountain grasslands as ~40,000 acres. Because the Montana vegetation type "shrub grassland" has a 25% higher average NEE (0.5±0.5 Mg C acre^{-1} yr^{-1}) but only characterizes ~13,000 ROW acres across Montana, Cote (2015) recommended exploring the extent to which manipulation of ROW grasslands to shrublands experimentally might lead to increased ROW soil carbon storage. Table

1 summarizes this experimental design, which was investigated in the complementary field project by Ament et al. (2018).

Table 1. Experimental design by Cote (2015) for the modification of ROW from grassland to shrubland to potentially increase carbon sequestration.

Current ROW type	Current type NEE	Current C sequestration	Possible ROW type	Possible type NEE	Possible C sequestration	Possible-current C seq.
Grassland (115,000 acres)	0.4 Mg C acre^{-1} yr^{-1}	46,000 Mg C yr^{-1}	Shrubland (13,000 [current]+115,000 [previously grassland] acres)	0.5 Mg C acre^{-1} yr^{-1}	64,000 Mg C yr^{-1}	18,000 Mg C yr^{-1}

Work by Ament et al. (2018) developed estimates of total ROW acreage owned by the MDT, which were obtained through Geographic Information System (GIS) analysis and combined with Montana vegetation data in order to determine acreage estimates according to vegetation type. Estimates of annual carbon sequestration from a network of global carbon measurement sites were applied to vegetation classes and used to estimate annual MDT ROW carbon sequestration. It was found that ROW owned by MDT consists of 70% grasslands and sequesters 75,292 metric tons of carbon per year. One management option analyzed was to convert existing MDT ROW grasslands into shrublands, since this type of physiognomic transformation has been found to increase SOC in a variety of studies. Our analysis found that conversion of existing grassland to shrubland, outside of the MDT ROW management zone, could increase carbon sequestration by 16,200 metric tons of carbon per year. This amount of annual carbon sequestration is equivalent to the removal of 12,505 passenger vehicles from the road per year, or the energy use of 5,420 homes per year.

Our initial conceptual model envisioned a four-step sequence: (i) the initial accumulation of deicing salts in roadside soils could lead to increased dispersion of clays and SOM, which

16

could, in turn, (ii) prevent or slow infiltration of roadside runoff, leading to (iii) an eventual increased accumulation of salts, with (iv) a resulting reduction in plant productivity and decline in soil organic carbon storage.

2.4 Site Considerations

The segment of Interstate Highway 90 (I-90) studied in our experiment is an east–west 4-lane highway that gradually rises in elevation between Three Forks, MT (elevation ~4000 ft) in the western terminus of the study area, to its highest point at Bozeman Pass (elevation ~5700 ft) at the eastern terminus of the study area, over a distance of ~38 miles. The experiment designed by Ament et al. (2018) elected to establish three test areas along this elevational gradient to determine if climate differences, primarily soil moisture and air and soil temperatures, have any influence on C sequestration rates. Derner and Schuman (2007), for example, showed that total precipitation, as indexed by mean annual precipitation, was an important factor for soil C flux to go from negative (more SOC respired from soil to atmosphere than CO_2 photosynthesized from atmosphere and ultimately contributing to the formation of SOM) to positive in western U.S. rangelands. Our original expectation was that the greater effective precipitation (the balance of precipitation inputs and evapotranspiration losses) of the highest-elevation site, Bozeman Pass (BP), would lead to greater SOC sequestration rates, relative to the driest, lowest-elevation site, Three Forks. Not only is the BP field site likely to receive the most precipitation, and therefore support the greatest above- and belowground productivity, but it should also experience the lowest evapotranspiration rates, making the soils wetter, therefore slowing decomposition rates. Globally, the ecosystems with the greatest accumulations of SOM (and thus SOC) tend to be the wettest ecosystems (Raich and Schlesinger 1992). Fortuitously, our BP site is only about 200 m from a continuously operated road weather information service (RWIS) station with real-time data (http://rwis.mdt.mt.gov/), one of 73 such stations across Montana.

The typical ROW along a highway is a disturbed site reclaimed after highway construction. Thus, ROW soils are not natural in the context of undisturbed soils, from a soil taxonomic perspective. These ROW soils tend to be relatively poorly developed (Entisols and Inceptisols are the formal U.S. taxonomic orders for the least developed soils), may represent a diverse collection of material sources from various locations, and may not be uniform. This stretch of interstate highway was only completed decades ago, relatively recently in soil development/geological time. The last portion of I-90 was completed in 1987, ~15 miles east of the project area. Soil development in this relatively semi-arid climate is slow.

In South African grasslands, soil C stocks were influenced more by soil parent material than by land use (Mills et al. 2005), indicating the important role of the parent material in C sequestration. Thus, our roadside experiment could not benefit from natural soil profiles. Rather, the three sites for the experimental plots are set in soils that represent disturbed profiles still recovering from construction, with weak horizon development. Therefore, it is likely that the C storage capabilities of the three sites have been influenced by factors other than the elevational gradient along which they were originally established in 2016.

CHAPTER 3.0   METHODOLOGY

3.1 Literature Review

A literature search was conducted to further explore the relationship between chloride-based deicers used in winter maintenance operations and the impacts of these deicers on SOC. We identified no active or completed research projects on this topic. Information from the literature review was used to better understand the complexity of evaluating the potential effects of deicing salts on roadside carbon sequestration, to design the field sampling techniques, to develop methods used to capture the data, and in the results.

In this task and throughout the project, we communicated with the MDT to determine the type of snow- and ice-control products applied on the road segments adjacent to the test plots, the application rates used, and the quantity of material applied annually for that road segment as far back in time as quality data allowed.

3.2 Field Sites Description

The three transects are located along a 61 kilometer (km) (38 mile [mi]) section of I-90 from Three Forks, MT, east to Bozeman Pass. The project secured an Encroachment Permit from MDT to conduct our experiments in the I-90 ROW.

We sought to distribute the experiment equivalently based on elevation, not on distance (mileage), within the study area. Thus, the three transects are located west of the Logan exit of I-90 near Three Forks at 4,102 feet [ft]), west of the Bear Canyon exit at 4,956 ft, and near Bozeman Pass, 5,713 ft (Figure 4). From west to east along the I-90 corridor, each successive transect site is approximately 800 ft higher in elevation than the previous site. The Logan/Three Forks site (3F) and the Bear Canyon (BC) site were located on the north side of I-90, and the Bozeman Pass (BP) site was located on the south side of the highway. The transects were mostly

on level areas, with some portions of each transect having slight slopes (<5%); all slopes face northward (azimuth of approximately 0 degrees). The field sites were located within the ROW, but outside the "active zone" or managed portion of the ROW, where maintenance of the safe zone, sight lines, mowing, and weed management are a priority.
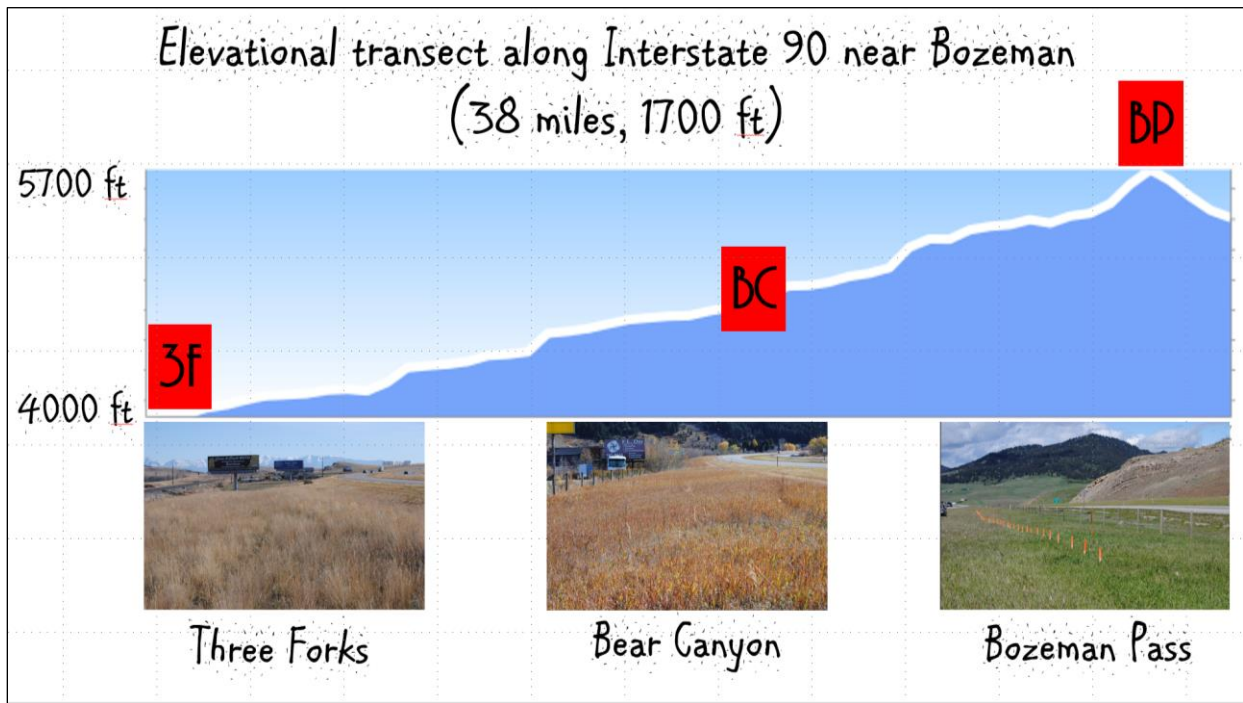
Figure 4. Location of the three transects along I-90 in Montana: 3F (Three Forks), BC (Bear Canyon), and BP (Bozeman Pass).

Three additional salt transects ranging in length from 42 to 55 ft (~13 to ~17 m) were established and sampled perpendicular to each of the three original 60-plot transects (one per each of the three sites). These transects were used to test whether road deicing salts might have accumulated within roadside soils, and if so, the extent to which a relationship could be discerned with SOC levels, which were estimated with an empirical relationship with SOM.

Soils were characterized along three transects at each of the three sites previously established and initially characterized in 2016 (Ament et al. 2018) (Figure 5).
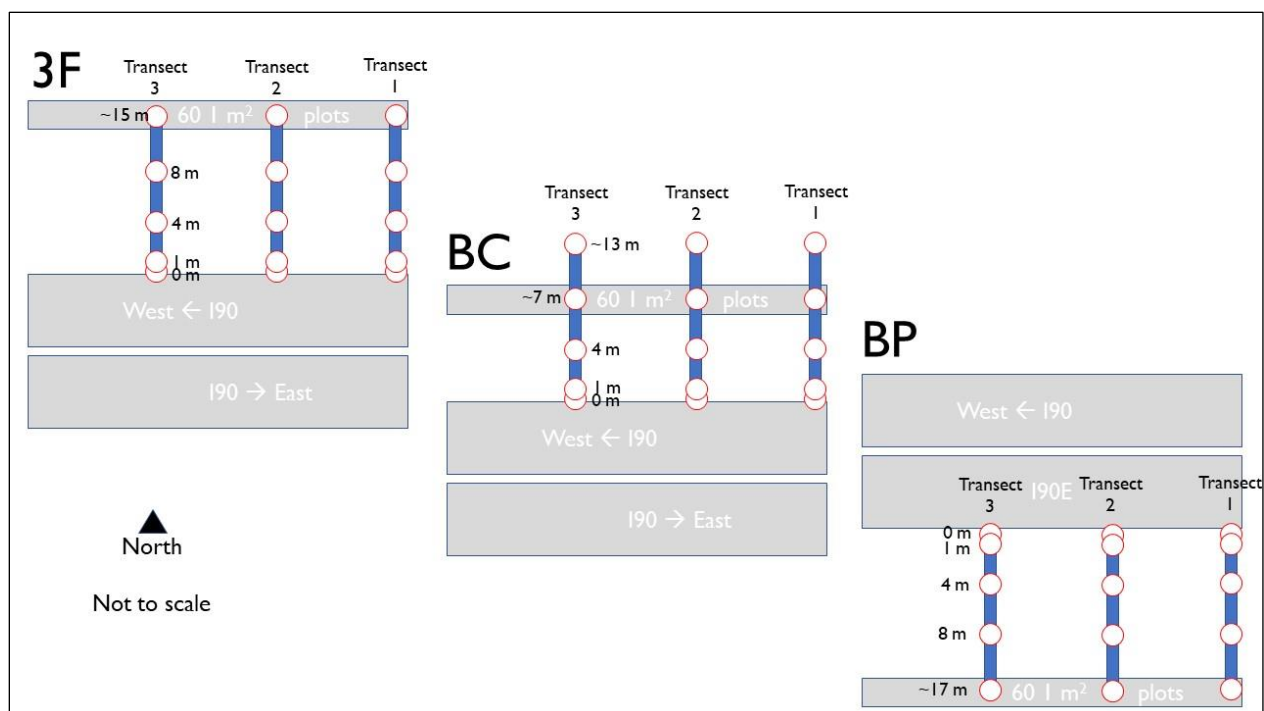
Figure 5. Layout of the three sample transects (blue vertical rectangles) with the soil sample locations (red circles), located along I-90 near Bozeman, MT. The gray rectangles located within the transects represent transects established by Ament et al. (2018).

From each transect, soils were sampled at 0, 3.2, and 13.1 ft (0, 1, and 4 m) from the roadside along transects established perpendicular to both the road and along a 120 m long transect (paralleling the road located approximately 50, 23, and 56 ft (15, 7, and 17 m) from the road edge, respectively) (Figure 5). The three salt transects at each field site intersected the 2016 transect at plots 1, 25, and 49, corresponding to distances of 2, 50, and 98 m from the eastern end of each 2016 transect; and were located approximately 157 ft (48 m) apart (Ament et al. 2018). Additional soil samples were collected from the 3F site at approximately 25 and 50 ft (8 and 15 m) from the road; for the BC site at approximately 23 and 43 ft (7 and 13 m); and for the BP site at approximately 26 and 56 ft (8 and 17 m). At 3F and BC, the samples collected farthest from the road were collected on the roadway side of the fencing that runs continuously along the highway. At the BP field site, the farthest samples coincided with the 2016 transect, which lies

between the highway fence and a frontage road to the south. The total number of soil samples is 45 (3 sites $\times$ 3 transects per site $\times$ 5 samples per transect).

Details associated with the different management treatments (mow height, addition of compost, planting of shrubs, etc.) for each of the three sites are described in the final report for the related project (Ament et al. 2018).

3.3 Soil Characterization Methods

Soil respiration measurements were obtained from nearly all of the 180 plots at various times throughout the year to clarify seasonal differences as well as treatment differences (Ament et al. 2018). Details related to soil respiration surveys can be found in Ament et al. (2018).

3.3.1 Soil Organic Matter Measurements

Changes in SOM for the three types of management techniques explored for roadside soils were reported in Ament et al. (2018). These findings, analyzed using standard loss-on-ignition procedures (Chatterjee et al. 2009) in 2016 ($n = 720$), and the 560 samples analyzed in 2018 ($n = 560$) are presented in Figure 6 to provide context for the 45 soil salt samples not previously characterized (Figure 7).

Nine additional soil samples were collected and composited by distance from the paved highway surface for independent measurements of SOM from Agvise Laboratories (https://www.agvise.com/) to provide quality control of the Montana State University (MSU) measurements.

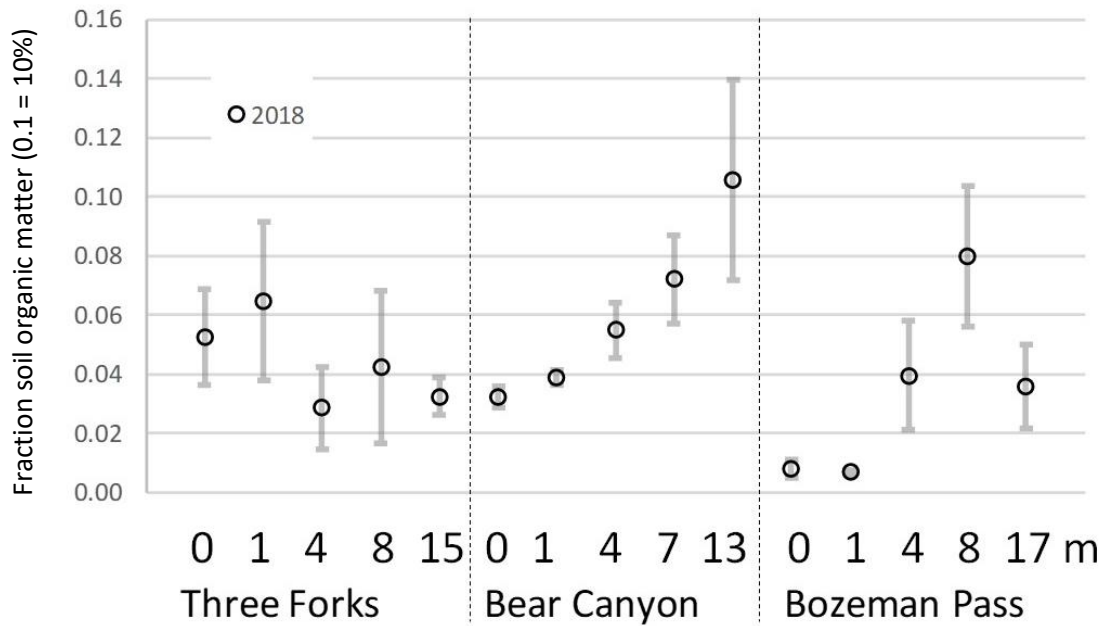**Figure 6.** Average SOM (mass basis; 0.1 is equivalent to 10%) as a function of distance from the roadway (I-90) (x-axis) from the three study site salt transects.
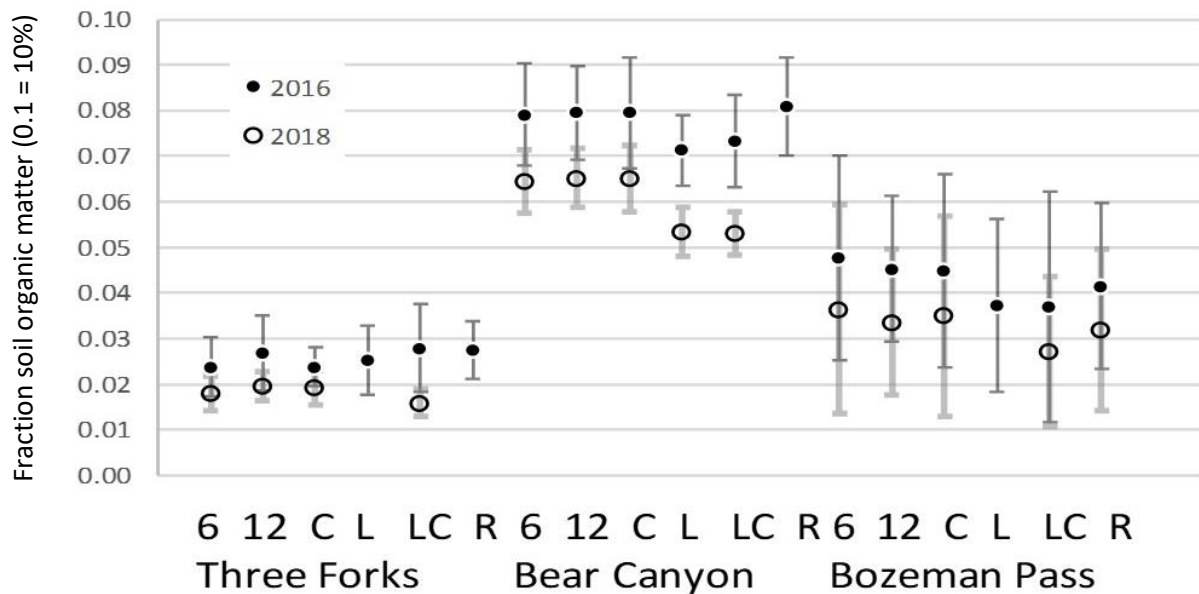

Figure 7. Average SOM (mass basis; 0.1 is equivalent to 10%) from vegetation transects from the three study sites. The six initial treatments are 6-inch mow (6), 12-inch mow (12), control (C), legume (L), legume control (LC), and rabbitbrush shrub (R).

<u>3.3.2</u> Soil Salinity

The nine additional soil samples collected and composited by distance from the paved

highway surface were analyzed for six salinity- and/or sodicity-related measurements by Agvise

Laboratories (https://www.agvise.com/):

1. Soluble salts on a 1:1 soil:water dilution as a proxy for electrical conductivity (units deciSiemens per meter [dS/m]);

2. Ammonium acetate-extractable exchangeable calcium (units mg/kg or $cmol_+/kg$);

3. Ammonium acetate-extractable exchangeable magnesium (units mg/kg or $cmol_+/kg$);

4. Ammonium acetate-extractable exchangeable potassium (units mg/kg or $cmol_+/kg$);

5. Ammonium acetate-extractable exchangeable sodium (units mg/kg or $cmol_+/kg$)*;

6. Cation exchange capacity (CEC; units $cmol_+/kg$)*.

[*Note: The quotient of analyte #5 and #6 can be used to index sodicity as the exchangeable

sodium percentage. For example, if the exchangeable sodium result were 87 mg/kg (the

maximum value across all samples; from locations ~3 ft [1 m] south of the interstate at BP), it

would convert to 3.8 $mmol_+/kg$, since the atomic weight of sodium is 23 g/mole or 23

mg/mmole. This converts to 0.38 $cmol_+/kg$, yielding an exchange sodium percent (ESP) of 6%

(for the corresponding CEC of 6.2 $cmol_+/kg$ [100*0.38/6.2]). Sodic soils are those with

ESP>15%.]

<u>3.4</u> Biomass and Vegetation Sampling

A census of vegetation from all 180 plots was taken in 2016 as well as in 2018, with 0.1

m^2 subplots destructively harvested each time to index aboveground biomass per plot (Ament et

al. 2018). The remaining characterization methods (aboveground biomass inventories, soil

respiration rates, and estimates of soil carbon residence times [in years]) are detailed in the

complementary report (Ament et al. 2018). Details related to aboveground biomass and

aboveground vegetation can be found in Ament et al. (2018).

CHAPTER 4.0   RESULTS

4.1 Soil Organic Matter

Our sampling across 45 locations in 2018 did not reveal consistently clear trends in SOM with distance from the paved highway (**Figure 6**), though for two sites (BC, BP), SOM tended to increase with increasing distance from the paved surface. At each of our five sample points ($n =$ 45), SOM measurements run in duplicate ranged from ~1% to >10%. Values for SOM were generally lowest adjacent to the road and highest farthest from the road. This trend could reflect the soil dilution effect from sand used on the road during winter maintenance operations. Sands were found to be concentrated in samples closest to the road (0 ft, 0 m), and sand has negligible SOM. This trend of increasing SOM with distance from the road was most clearly expressed at the BC site, though variability also increased sharply with distance from the road.

In 2018, 560 soil samples were analyzed from the vegetation transects paralleling the highway; these results are provided in Figure 7 for context.

A comparison of SOM levels along the three vegetation transects parallel to the highway (Figure 7) with SOM levels from the nine salt transects (Figure 6) perpendicular to the highway is illustrative. The salt transects clearly show a broader range of SOM concentrations as high as 0.105 or 10.5% (Figure 6). Note that the vegetation transects may not have adequately characterized some of the spatial variability in roadside SOM levels. As just one example, the range in vegetation transect SOM was relatively narrow (5.5–6.5%) across the three broad management practices explored, but along the three perpendicular salt transects at the same BC site, SOM ranged more widely from ~3% to >10%.

4.2 Soil Salinity and Exchangeable Cations

No elevated salt concentrations were found at any of the nine locations sampled at the three study sites (Table 2, Figure 8). The highest soil salt level observed (reported as electrical conductivity in a 1:1 soil:water extract) was 0.21 deciSiemens per meter (dS/m), or approximately 1/20th the threshold typically regarded as indicative of "saline" soils (4 dS/m; Mavi and Marschner 2017). This soil salt level occurred at a sample location farthest from the road. This pattern was consistent across all three sites (3F, BC, and BP), with average EC values (±1SD, n=3) for the sample points farthest from the road typically triple the EC values for the sample points closest to the road (0.17±0.04 vs. 0.05±0.01 dS/m, respectively; Figure 8).

Table 2. Summary of soil SOM and salinity/sodicity measures by distance from the paved surface across the three test sites (3F, BC, BP) along I-90. Data from Agvise Laboratories.

Sample	Distance (m)	%SOM	EC (1:1, dS/m)	xCa ppm	xMg ppm	xK ppm	xNa ppm	CEC (cmol+/kg)	Exchangeable sodium percentage
BC1	0	2.3	0.05	1042	205	87	26	7.3	2
BC2	1	3.1	0.07	1253	282	124	45	9.1	2
BC3	13-17	6.1	0.21	2700	434	325	30	18.1	1
BP1	0	0.5	0.04	667	177	59	57	5.2	5
BP2	1	0.4	0.05	787	207	69	87	6.2	6
BP3	14	3.2	0.16	3535	408	310	35	22.0	1
3F1	0	3.4	0.06	1217	116	102	35	7.5	2
3F2	1	4.4	0.08	1538	15.5	160	30	9.7	1
3F3	14	2.8	0.14	3025	195	175	18	17.3	0
3F4	dup of 3F1	3.6	0.08	1078	103	87	31	6.6	2

From Table 2, the exchangeable sodium percentage (ESP) was estimated as the quotient of exchangeable sodium and cation exchange capacity (CEC). The BC1 sample is a composite of three samples collected 0 ft (0 m) from the paved surface, one from each of the salt transects perpendicular to the highway. The BC2 sample is a composite ~3 ft (1 m) from the paved surface, and the BC3 sample is a composite collected ~42–56 ft (13–17 m) from the paved surface. One duplicate was analyzed (re: 3F4=3F1) for quality-control purposes. Coefficients of

variation (SD/average) were generally low, ranging from 4% (SOM) to 20% (EC), with an average of 10%.
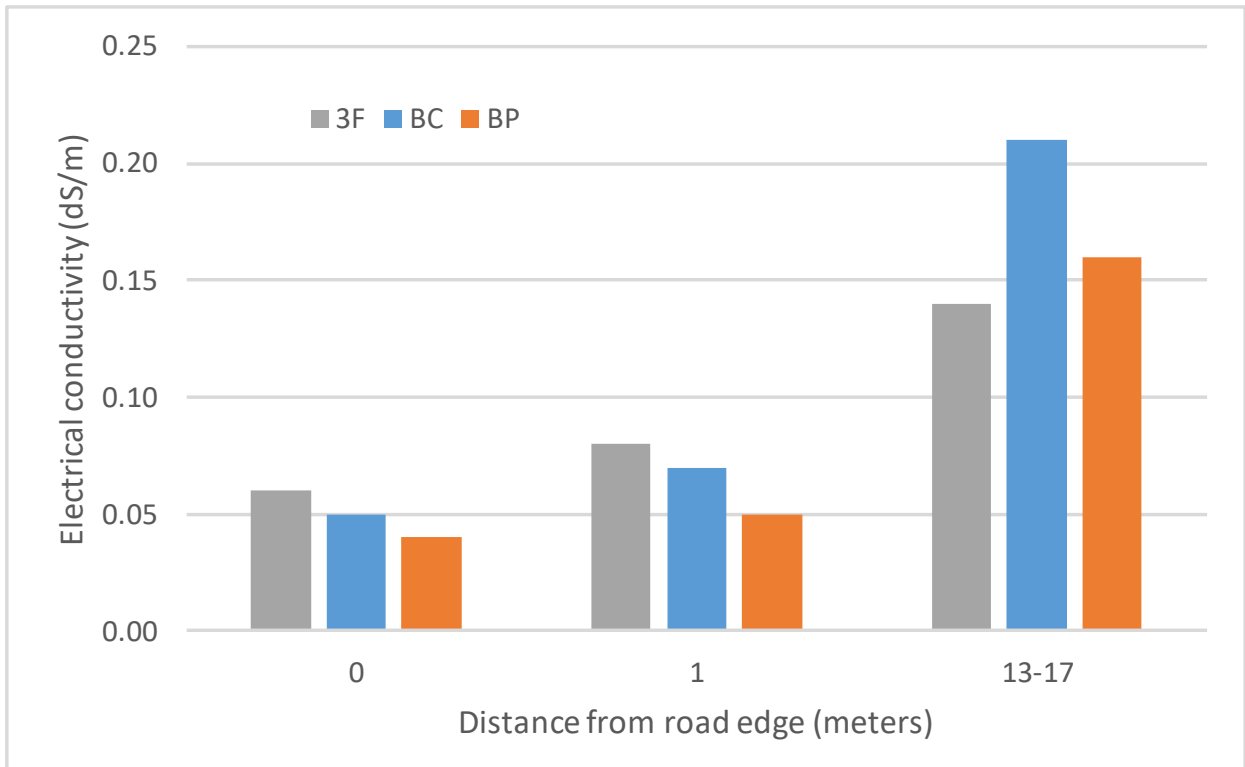
Figure 8. Electrical conductivity (dS/m) (y-axis), or salt concentration, with distance from the roadway (x-axis) at the three study sites.

Other indices of salinity such as exchangeable calcium are broadly consistent with this trend, ranging from 975±281 parts per million (ppm) nearest the road to 3087±421 ppm farthest from the road (Figure 9, Table 2). Exchangeable sodium values were uniformly low, ranging from 18 to 87 ppm across all nine composite samples analyzed. When averaged across distance from the road, exchangeable sodium levels averaged (±1SD) 39±16 ppm for soils nearest the roadside versus 28±9 ppm for soils farthest from the road (Table 2, Figure 10).

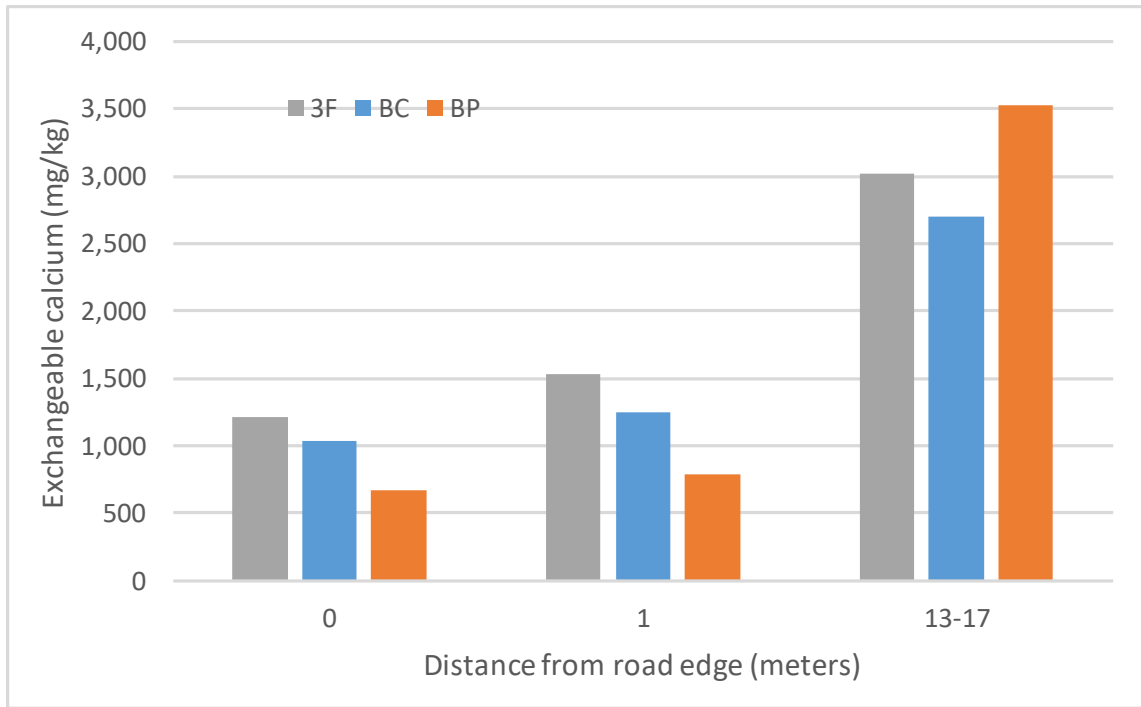**Figure 9.** Exchangeable calcium (mg/kg) (y-axis) with distance from the roadway (x-axis) at the three study sites.
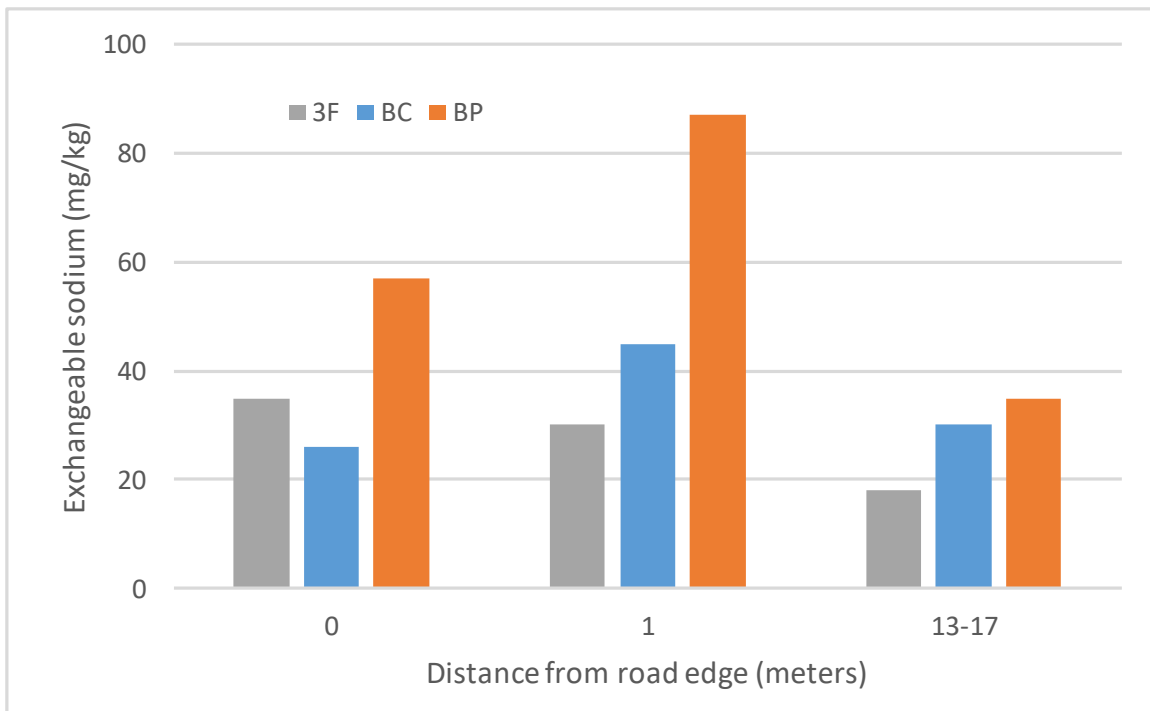

Figure 10. Exchangeable sodium (mg/kg) (y-axis) with distance from the roadway (x-axis) at the three study sites.

Two ratios of cations were explored for insights into roadside salt patterns. First, the

average levels of the divalent cation calcium (Ca^{+2}) to that of monovalent cation sodium (Na^{+1})

across distance from the road was determined. The Ca:Na ratios were 25, 22, and 112 for the 0

m, 1 m, and 13–17 m sample points from the road, respectively. The ratio shows a shift for the

farthest soils to relatively greater calcium abundance on the CEC, a common finding for

southwestern Montana soils. Second, ratios of the monovalent cation potassium (K^{+1}) to sodium

were 2.1, 2.2, and 9.8 with increasing distance from the road. These results show a similar trend

of relatively greater potassium abundance at the soils farthest from the road. However, it is

important to note the overwhelming abundance, in an absolute sense, of calcium relative to

potassium.

Cation exchange capacity (CEC) provides information on the ability of negatively

charged colloids such as clays and organic matter to retain and exchange cations. From the three

sample distances of 0 m, 1 m, and 13–17 m from the road, CEC shifted from 6.7 to 8.3 to 19.1

$cmol_+$/kg, respectively (Table 2). This pattern is generally consistent with a shift from soils under

the strong influence of sand-sized (typically, 0.05–2 mm) particles and may be associated with

either highway construction or winter maintenance operations sand applications, which have

negligible CEC. At the Arthur H. Post Farm (an agricultural research center unaffected by

deicing salts or sand applications) just 8 km southwest of I-90 between Bozeman and Belgrade,

EC levels ranged from 0.05–0.11 dS/m in the upper 30 cm of the soil sampled, while CEC values

averaged ~18 $cmol_+$/kg (Romero et al. 2017).

4.3 Comparison of MSU and Agvise SOM Results

The MSU SOM results were aggregated by distance from the paved surface and

compared with the Agvise composite SOM results (Figure 11). Agreement between the

independent sets of results was strong, with $R^2>0.94$, though the MSU samples showed a slight

tendency to be greater than the 1:1 line, and this offset grew with increasing SOM. We could not

derive measures of within-sample variances for the composite sample results from Agvise (note

the absence of horizontal error bars), but the larger error bars associated with the higher-SOM

MSU results could be shrunk with additional sample preparation efforts, including the grinding

of the soil samples prior to heating for loss-on-ignition.
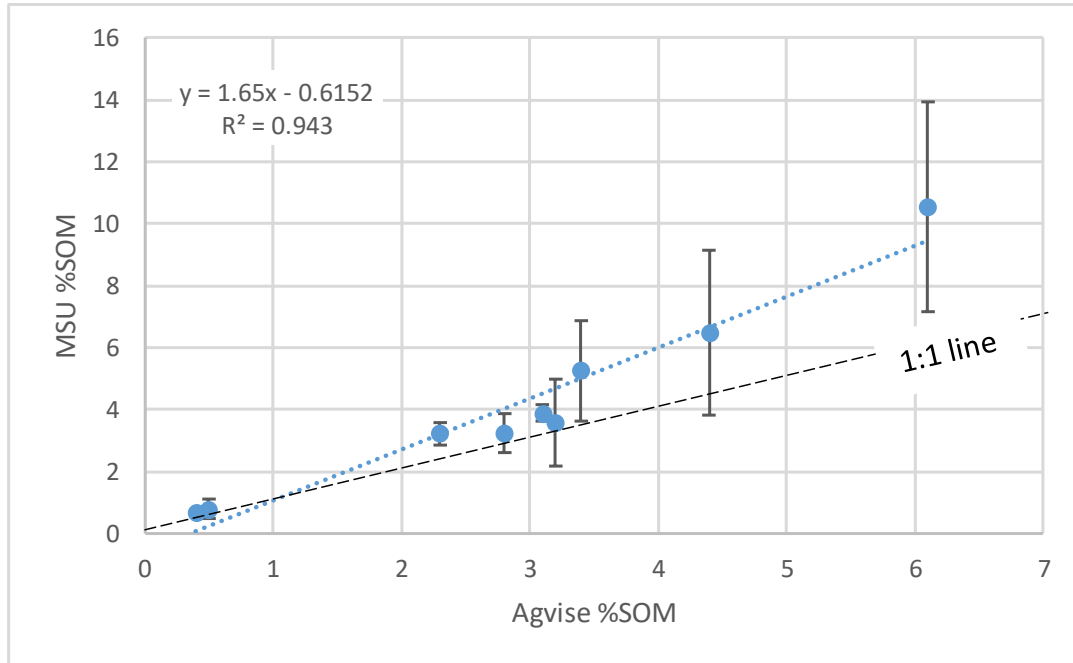
Figure 11. Comparison of MSU and Agvise Laboratory SOM results.

4.4 SOM and Salt Relationship

We found no or weak evidence of a negative relationship between our indices of soil

salinity and SOM levels, with EC and CEC all having R^2 values <0.5 with respect to SOM

(Figure 12). These R^2 values are likely inflated, as they were less than 0.07 with the exclusion of

a single sample (10.5% SOM) that also showed *relatively* elevated EC (0.21 dS/m),

exchangeable calcium (2,700 ppm), and cation exchange capacity (18.1 cmol$_+$/kg). For reference

purposes, the Arthur H. Post Farm just southwest of I-90 showed SOC levels ranging from 13.1

(0–10 cm) to 7.8 (20–30 cm) g/kg (Romero et al. 2017), which would translate to an estimated 2.6 to 1.6% SOM, assuming SOM is double SOC.

Evidence from the literature (e.g., Wong et al. 2010, Amini et al. 2016, and Mavi and Marschner 2017) points to the potential for salts, such as road deicing salts, to affect soil carbon dynamics. Further clarifying those types of effects in Montana will require experimental additions of deicing salts to roadside soils, together with long-term monitoring of soil processes at treated and untreated plots. For example, Wong et al. (2010) reported a number of studies that examined both measures of salinity (EC) and sodicity (ESP), as well as SOC. It is clear from the plotted data (Figure 13) that, while there may be a negative relationship between the indices of salinity or sodicity and SOC, these relationships are weak at best ($R^2 \leq 0.30$).

In fact, when the Wong et al. (2010) dataset is restricted to just those ESPs within the range of values from this study (e.g., ESP≤6), the R^2 of the ESP relationship with SOC is reduced to $R^2 \leq 0.03$, showing no relationship between salt indices and SOC either in the literature or at the three study sites along I-90 in southwestern Montana (Figure 12 and Figure 14).
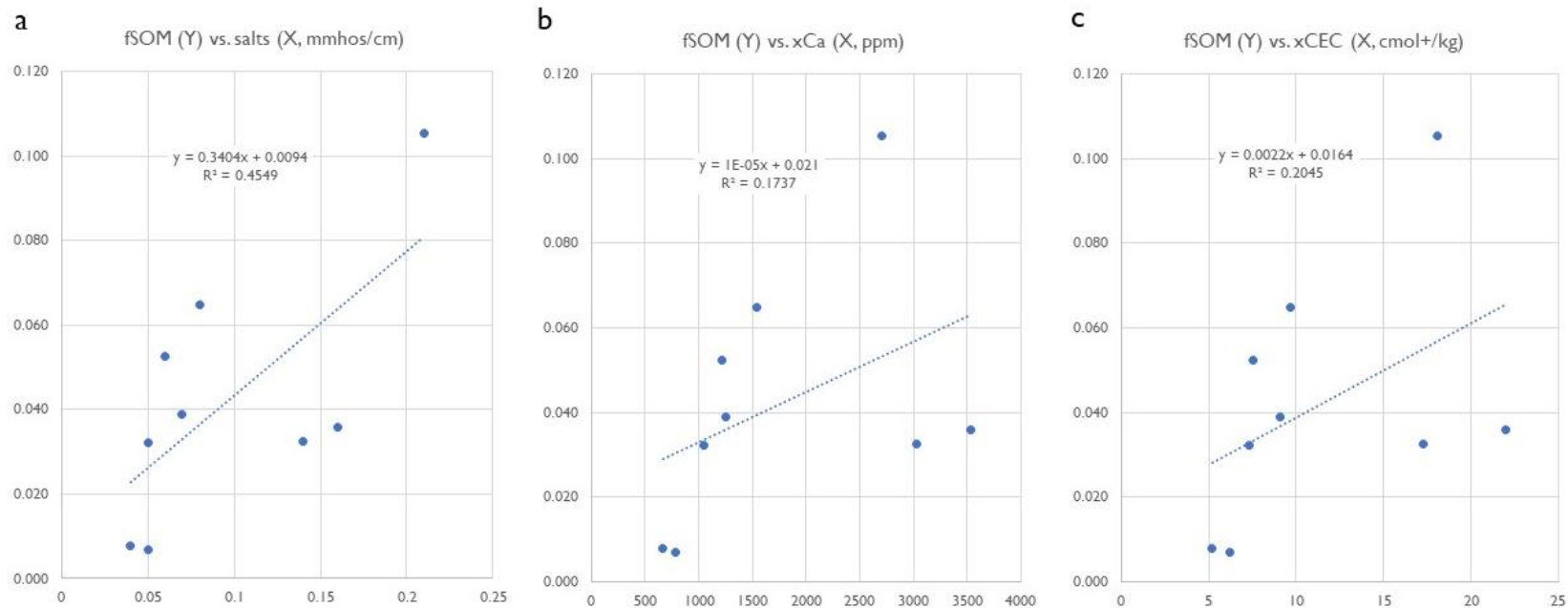
Figure 12. Relationships of salt indices (x-axes) and soil organic matter (y-axes): (a) electrical conductivity (dS/m; x axis) vs. SOM; (b) exchangeable calcium (mg/kg; x axis) vs. SOM; and (c) cation exchange capacity (CEC [cmol+/kg]; x axis) vs. SOM.
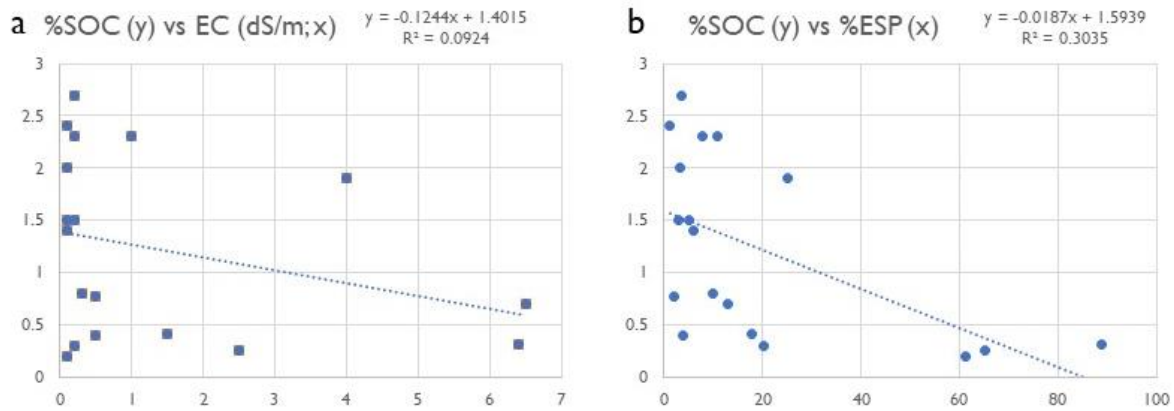
Figure 13. Literature-derived relationship between indices of salinity or sodicity and SOC (Wong et al. 2010, Table 1): (a) electrical conductivity (EC) vs. SOC; and (b) exchangeable sodium percentage (ESP) vs. SOC. A threshold of ESP>15% is often used to distinguish non-sodic from sodic soils.
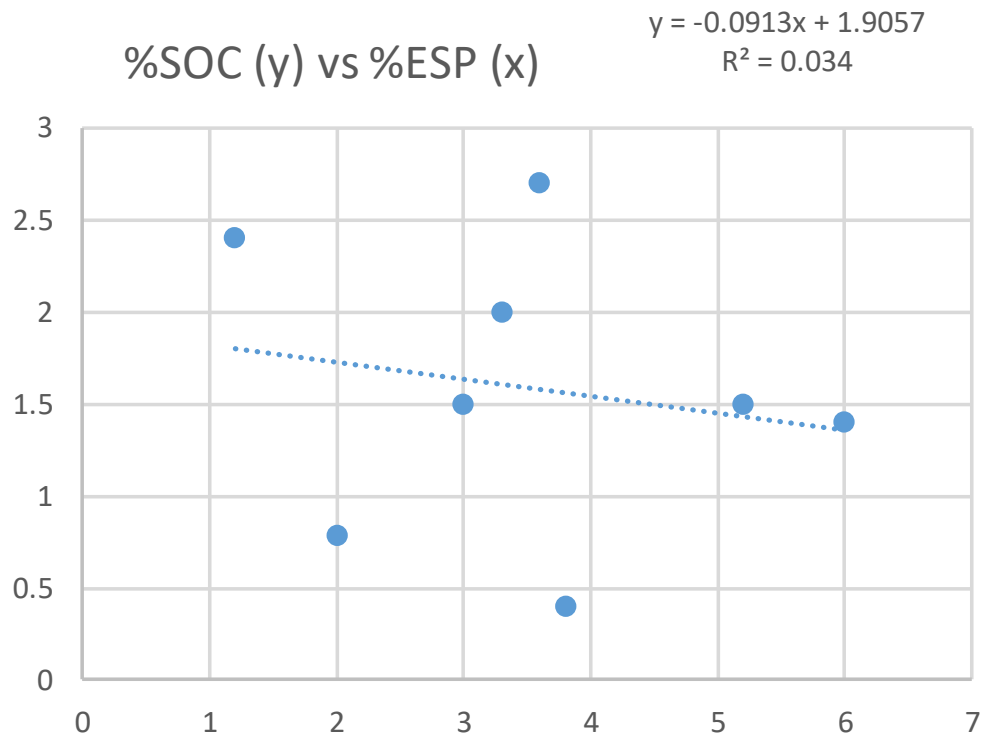
Figure 14. Truncated representation of the published relationship between exchangeable sodium percentage and SOC (Wong et al. 2010, Table 1). All values with ESP >6% have been censored in this graph; to see all ESP values, refer to Figure 13b (Wong et al. 2010).

4.5 Deicers Applied by MDT

The MDT conducts winter maintenance operations on the I-90 corridor that passes

through the three field sites; it uses magnesium chloride ($MgCl_2$) liquid to anti-ice the highway

34

during winter maintenance operations. Anti-icing refers to the application of liquid product to the road surface before the storm begins, helping to prevent snow and ice from bonding to the pavement surface and aiding in snow removal by plowing. Typical application rates for $MgCl_2$ range from 30 to 60 gallons per lane mile, applied in a stream in the wheel paths. Historically, MDT applied $MgCl_2$ at slightly higher rates and as spray to cover the entire pavement surface.

The MDT also uses sand for winter traction, which has 15% rock salt added by weight. The salt not only aids in the workability of the sand, but also adds a minor deicing aspect to the sand/salt blend. The MDT has begun pre-wetting the sand/salt blend as it is applied with 8 to 15 gallons of $MgCl_2$ per yard sand/salt. Pre-wetting aids in keeping the product on the road. Both anti-icing and pre-wetting have been identified as best practices in winter maintenance operations to reduce costs, reduce chloride use, and improve efficiency.

CHAPTER 5.0   CONCLUSIONS

Taken together, our research results imply that if road de-icing salts are altering patterns of roadside SOM and potential carbon sequestration, this effect was not captured by our experimental design nor did de-icing salts appear to have affected roadside vegetation during our sampling effort in late spring/early summer 2018. Evidence from the literature (e.g., Wong et al. 2010, Amini et al. 2016, and Mavi and Marschner 2017) points to the potential for salts, such as road de-icing salts, to affect soil carbon dynamics, but clarifying those types of effects in Montana will require experimental additions of de-icing salts to roadside soils, together with long-term monitoring of soil processes at treated and untreated plots.

These findings highlight the value of experimentally separating the multiple confounding effects on roadside soils of winter maintenance operations and practices, such as but not limited to the following:

(i)   Roads could focus the flow of water (snowmelt and runoff) to roadside soils, artificially irrigating these soils, boosting plant productivity and therefore the potential for terrestrial soil organic carbon sequestration (TSOCS);

(ii) Roads could focus the flow of salts to roadside soils, depressing non-saline-soil- or non-sodic-soil-adapted plants, and as a result, lowering the potential for TSOCS; and

(iii) Roads could focus the flow of sand to roadside soils, lowering water-holding capacity, which could lower the potential TSOCS by lowering plant productivity, but also lowering cation exchange capacity, which could increase the possibility of leaching of any roadside salts, which could, in turn, boost TSOCS by increasing non-salt- or non-sodic-adapted plant productivity.

An additional factor to be considered in each of these potential winter maintenance operations consequences for roadside soils would be the local "geomorphology" associated with the specific sections of highway being monitored. For example, does the highway slope toward the roadside soils being monitored, and if so, how steep is that slope? What is the upslope contributing area for every sample point being monitored? Even small (<1%) slopes for impervious surfaces such as interstate highways can strongly influence hydrologic flow paths. Do sloping portions of highways have aspects that lead to greater or lesser sublimation:snowmelt ratios? Higher sublimation:snowmelt ratios could concentrate deicing salt and sand inputs.

Our initial conceptual model required the initial detection of salts in roadside soils adjacent to where deicing salts are used. Because we were unable to detect elevated salts in these soils at our existing three TSOCS project sites, we were not able to refine this conceptual model. Future work might target specific stretches of highway where deicing salts are used with greater frequency, at greater application rates, or both. Absent these findings, future investigations might also employ roadside soils under the more carefully controllable conditions of a greenhouse to explore the direct and indirect mechanistic connections between deicing salts and broader SOC dynamics. Future work could explore questions such as the following: To what extent does the application of deicing salts alter the hydrologic properties of soils? Are these relationships linear or nonlinear? In a semiarid climate like that of southwestern Montana, are roadside plant communities generally more salt- and sodium-tolerant?

Nevertheless, this study has clarified that a fuller characterization of the mass inputs (water, salts, sands) and the possible flow paths of that mass would shed light on the impacts of highway management practices on roadside soils. These data should be a critical consideration in future evaluations of the roles that roadside soils could play in integrated TSOCS strategies.

CHAPTER 6.0   REFERENCES

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APPENDIX A: GLOSSARY

Abbreviations, Acronyms and Definitions

C	carbon
$CaCl_2$	calcium chloride
CEC	cation exchange capacity
CO	carbon monoxide
CO_2	carbon dioxide
CO_2E	carbon dioxide equivalent – a measure to compare various greenhouse gases based on their global warming potential (GWP), usually expressed in metric tons
CCS	carbon capture and storage = carbon sequestration
EC	exchange capacity
FHWA	Federal Highway Administration
FLMA	federal land management agency
GHG	greenhouse gas
GIS	geographic information system
MDT	Montana Department of Transportation
$MgCl_2$	magnesium chloride
MSU	Montana State University
NaCl	sodium chloride
NEE	net ecosystem exchange of CO_2
NHS	National Highway System
ROW	right-of-way
SIC	soil inorganic carbon
SOC	soil organic carbon
SOM	soil organic matter
TCSOCS	terrestrial soil organic carbon sequestration

Conversion Factors

Multiply	By	To obtain
centimeter (cm)	0.3937	inch (in.)
meter (m)	3.281	foot (ft)
hectare (ha)	2.471	acre (ac)
gram (g)	0.03527	ounce (oz)
kilogram (kg)	2.205	pound (lb)
mile (mi)	1.609	kilometer (km)

Carbon Compound Weight Conversion Factors

1 g CO = 0.43 g C
1 g CO_2 = 0.27 g C
1 g CO_2E (CO_2 emission equivalent) = 0.27 g C

Equivalents

1 megagram (Mg) = 1 million (10^6) grams = 1,000 kilograms = 1 megagram = 1 metric ton (Mt)
1 gigagram (Gg) = 1 billion (10^9) grams = 1,000 metric tons = kiloton (kMt)
1 teragram (Tg) = 1 trillion grams (10^{12}) = 1,000,000 metric tons = megaton (MMt)
1 petagram (Pg) = 1 quadrillion (10^{15}) grams = 1,000,000,000 metric tons = gigaton = (BMt)