

**Mitigating Roadway Deicing Salt Runoff:
Utilizing Environmental Containment Socks to Sequester Na⁺ and Cl⁻**

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Abstract

Roadway deicing salts are applied annually in Virginia and other states impacted by winter weather on a large scale. The United States Environmental Protection Agency (EPA) estimates that the damage caused by road salt costs approximately \$5 billion dollars per year in repairs for cars, trucks, roads, and bridges. Improvements have optimized winter maintenance activities, such as better application equipment and technology, utilization of precise weather forecasting, and the implementation of new highway deicing strategies. Nevertheless, additional methods are necessary to combat the detrimental effects caused by deicing practices if the broader ecological consequences of deicing salts are to be adequately addressed. One such method is phytoremediation; however, the plant species that are able to uptake sodium (Na^+) and chloride (Cl^-) ions from the soil do not grow during the winter when road salt application is most frequent. In addition, implementation is limited as phytoremediation cannot be used on impervious surfaces like parking lots where salt can drain freely into sewers and drains. In these situations, environmental containment socks (ECS) are a novel way to filter pavement runoff before it enters a drainage system and ultimately the environment. In this experiment, multiple ECS filled with various Na^+ and Cl^- binding materials (biofilters) were exposed to a 100mM solution of NaCl to evaluate the potential of this technology at adsorbing salt runoff. The results demonstrate that both Na^+ and Cl^- can be retained by biofilters and suggests that they can be prevented from entering into the environment. Further studies are needed with ECS in the field to evaluate their potential for recycling sodium and chloride ions for potential reuse.

Table of Contents

List of Tables.....	4
List of Figures.....	4
Introduction	6
Purpose.....	8
Literature Review.....	9
Road Salt Application.....	9
Efforts to Mitigate Road Salt Pollution.....	13
Economics.....	17
Road salt Impacts.....	20
Vegetation & Soil	20
Infrastructure.....	24
Waterways.....	25
Human Health.....	28
Biofilter Materials.....	30
<i>Cannabis sativa</i> L.....	30
Biochar.....	31
Materials & Methods.....	32
Results & Discussion.....	37
Conclusions.....	45
Literature Cited.....	47
Tables	61
Figures.....	62

List of Tables

- 1) Table 1. Weights of biofilters for treatment and control groups.....33
- 2) Table 2. Saline runoff filtration capabilities of a 10kg biofilter ECS.....40

List of Figures

1) Figure 1. Diagram of road salt runoff transport methods and resulting impacts.....	13
2) Figure 2. Photo of road-side trees impacted by road salt runoff.....	20
3) Figure 3. Cross section of hemp stalk.....	30
4) Figure 4. Photos of experiment materials.....	33
5) Figure 5. Photos of ECS control and treatment groups in bins.....	34
6) Figure 6. Photos of ECS treatment group placed and submerged in bin containing 100 mM NaCl solution.....	35
7) Figure 7. Test results detailing sodium and chloride binding abilities of various biofilters.....	38
8) Figure 8. Graph comparing sodium and chloride binding abilities of various biofilters.....	38
9) Figure 9. Conceptual diagram of NaCl reclamation and reuse system.....	42
10) Figure 10. Photo of ECS in front of parking lot drain.....	43

Introduction

The use of sodium chloride rock salt (NaCl) as a deicer on roads has risen dramatically over the past century (U.S. Geological Survey, 2022). Consequently, numerous environmental and societal impacts continue to arise, such as long-term salinization of freshwater resources, corrosive damage to infrastructure, degradation of soil and aquatic ecosystems, and negative impacts to human health (Tiwari & Rachlin, 2018; Jackson & Jobbágy, 2005; Stranko *et al.*, 2013; D'itri, 1992; Kaushal *et al.*, 2005; Siegel, 2007; Löfgren, 2001). Despite the negative effects road salts have on our infrastructure and environment, their use is absolutely necessary because they prevent ice from accumulating on roads, facilitating the function of the U.S. economy by minimizing the risk of vehicle accidents (D'itri, 1992). Although other forms of chemical deicers are utilized, NaCl is the primary compound of choice because it is affordable and readily available (D'itri, 1992; New Hampshire Department of Environmental Services, 2016).

Once salt is applied, the dissociated Na⁺ and Cl⁻ ions can infiltrate the surrounding environment through a myriad of transport mechanisms, ultimately increasing the salinity of soil, streams, lakes, wetlands, and stormwater management ponds (Jones *et al.*, 2015; Kaushal *et al.*, 2005). Sodium is a positively charged cation that usually attaches to the negative charged sites within the soil profile, whereas the chloride anion is much more mobile and can readily infiltrate down into groundwater supplies (New Hampshire Department of Environmental Services, 2016; D'itri, 1992; Siegel, 2007). While both elements contribute to saline runoff impacts, chloride is often the focus of studies that investigate the effects of road salt application because it is the anion of most deicing salts and stays in solution once dissociated, usually moving with water flow in a watershed. The continuous rise in chloride concentrations in freshwater resources has been linked to the use of road salt, leaving some experts to predict

that current aquatic wildlife will not be able to survive in the next 50 years if this trend continues (Dugan *et al.*, 2017).

Although there have been significant improvements in the efficiency of road salt storage and application that have lowered its entrance into the environment, additional technologies that capture and remove NaCl are required to significantly address this complex and global issue. Alternative methods such as phytoremediation of salt-affected soils and the recycling of saline runoff show tremendous potential; however, additional methods are required to more deliberately sequester NaCl runoff. A complementary sequestration strategy that would benefit areas where phytoremediation is not feasible involves placing natural Na⁺ and Cl⁻ binding materials (biofilters) into environmental containment socks (ECS) to filter salt runoff in paved/impermeable areas. To test this novel idea, an experiment was conducted that evaluated the efficacy of biochar and two types of hemp fiber at adsorbing Na⁺ and Cl⁻ particles in an aqueous solution. These biofilters were placed into ECS and then submerged in a 100 mM solution of NaCl. Biochar had the highest adsorption potential, but all biofilters were effective at binding to Na⁺ and Cl⁻ and thus offer means to filter road salts in runoff. The experiment conducted in this study was done with the intention of identifying an alternative technology to sequester NaCl runoff and thus mitigate its effects on the environment. I will discuss the practice and economics of road salt application, its impacts on vegetation, soil and aquatic ecosystems, infrastructure, and human health. In addition, information regarding the chosen material placed inside the ECS will be detailed as well.

Purpose

This research project examined the binding capacity of 4 biofilters placed in a 100 mM solution of NaCl. The objective of this paper was to investigate novel ways to keep Na⁺ and Cl⁻ out of the environment, with special emphasis on testing different biofilter materials in environmental containment socks to see which are best suited for sequestering sodium and chloride. Hemp fibers were selected as a biofilter material for ECS was due to their absorptive and adsorptive properties (Stevulova *et al.*, 2014; Stevulova *et al.*, 2015; Cigasova *et al.*, 2014; Reh & Barbu, 2017; Nguyen *et al.*, 2009). Research regarding hemp's potential in biocomposites for construction applications noted its hydrophilic behavior (Stevulova *et al.*, 2015). In addition, *Cannabis sativa* L. has been shown to be an excellent candidate for phytoremediation of various toxic pollutants and can be grown in environments with high NaCl concentrations (Varga *et al.*, 2022; Alufasi *et al.*, 2020; Ahmad *et al.*, 2015; Husain *et al.*, 2019; Citterio *et al.*, 2003). Biochar was also selected as a biofilter as it has long been recognized for its cation and anion adsorption properties (Zhao *et al.*, 2019).

The target population for this experiment are all those who are impacted by the annual application of road salts and the broader problems it contributes to such as salinization of soils and freshwater bodies, and the corrosion of infrastructure.

The long-term goal of this paper is to develop a product that can be utilized in a comprehensive strategy to capture and recycle NaCl salt. The ECS technology can be deployed in areas where phytoremediation cannot be possible such as parking lots. ECS could then be collected and transported to a facility that reclaims NaCl from saturated biofilter materials. Recaptured NaCl can be redistributed, saving highway agencies money while simultaneously lowering the amount of NaCl exposure to the environment. Phytoremediation technology using candidate halophytes along highways are also part of this strategy, and potentially these plants can be turned into biochar for use in ECS.

Literature Review

Road salt Application

The first usage of sodium chloride (NaCl) as a deicing compound dates back to 1938 in New Hampshire (Stranko *et al.* 2013). Over the next 20 years, road salt application rose dramatically due to the expanding US highway system and the “bare-pavement” policy employed by most urban and suburban areas (D’itri, 1992; Fitch *et al.*, 2005). The “bare-pavement” policy was a winter maintenance standard promoted by the Salt Institute - a lobbying group for the use of salt in all aspects - that advocated for the application of deicers until the roadways were clear of snow and ice, thus resulting in “bare-pavement”. While the benefits to drivers were hard to dispute, this often led to an overapplication of deicing salts to achieve this standard (McConnell & Lewis, 1972). Consequently, by the 1950s the adverse environmental and infrastructural impacts of road salt runoff started to be documented in reports from the Federal Highway Administration, EPA, and Transportation Research Board, causing a revision in the deicing practices employed by each state’s DOTs (D’itri, 1992). For example, a revision that many authorities have made to the previously mentioned “bare-pavement” policy is that it is no longer a standard for all roads, but instead for roadways with high levels (> 10,000 vehicles) of traffic (Kelly *et al.*, 2012).

Further, improved equipment, reduced application rates, the use of abrasives, and sophisticated weather tracking have contributed to the stabilization of salt use over the past few decades (Dao *et al.*, 2019; U.S. Geological Survey, 2022); however, rising salt concentrations in soil and groundwater continue to be observed and contribute to the long-term salinization of these ecosystems (Jackson & Jobbágy, 2015; Dugan *et al.*, 2017; Kaushal *et al.*, 2005). Although alternative deicing applications continue to be investigated, it is unlikely any will replace the estimated 12-24 million tons of sodium chloride applied on roadways in the United States annually (Rippy *et al.*, 2022; Tiware & Rachlin, 2018; Jackson & Jobbágy, 2015).

Sodium chloride continues to be the most widely used chemical deicing compound because it is inexpensive, easy to handle, store, and spread (D'itri, 1992; New Hampshire Department of Environmental Services, 2016). It is typically derived from mined rock salt which the US produced 40 million tons of in 2021, 42% of which was for highway deicing (Siegel, 2007; U.S. Geological Survey, 2022). Once mined, it is distributed and stored in various storage facilities throughout a certain state until its application; for instance, VDOT stores more than 350,000 tons of salt in its 300 storage facilities (Fitch *et al.*, 2008). A few decades ago, most roadway salt was stored in uncovered stockpiles outside which unsurprisingly resulted in runoff with chloride concentrations up to 13,500 mg/L in some cases (Ohno, 1990). Today, most salt stockpiles are located inside covered buildings with impermeable loading pads and nearby collection basins for runoff to drain into (Fitch *et al.*, 2008). Although runoff collected from these impermeable sites had lower levels of contamination relative to uncovered sites, chloride concentrations still exceeded state and federal guidelines by 1,000 mg/L in one study (Fitch *et al.*, 2005). To make matters worse, these large stockpiles require the addition of anti-caking compounds like ferrocyanide to prevent the granular NaCl from clumping so it spreads evenly, potentially killing aquatic life if runoff concentrations from stockpiles are high enough (Siegel, 2009; Tiwari & Rachlin, 2018; Ohno, 1990; Jones *et al.*, 2015).

In the 28 states that use chemical deicers NaCl is undoubtedly the most common, although alternative deicers such as calcium chloride (CaCl_2), magnesium chloride (MgCl_2), potassium chloride (KCl), and calcium magnesium acetate (CMA) are used as well (D'itri *et al.*, 1992; Valleau *et al.*, 2022; Renshaw, 2021). Both CaCl_2 and MgCl_2 are used when temperatures are lower than 15F (NaCl loses its effectiveness at this point) and are considered safer than NaCl (Tiwari & Rachlin, 2018; New Hampshire Department of Environmental Services, 2016). While both have their benefits, CaCl_2 is five times more expensive than NaCl and both CaCl_2 and MgCl_2 still contribute to chloride pollution (Tiwari & Rachlin, 2018; Renshaw, 2021). CMA is a mixture of limestone and acetic acid that works very similarly to NaCl but is less corrosive and

more environmentally friendly (Renshaw, 2021; New Hampshire Department of Environmental Services, 2016). Sustainable agricultural by-products of corn, beet, molasses, and alcohol have been shown to be effective prewetting treatments in winter maintenance programs, meaning that they do not melt snow and ice but rather slow the development of ice crystals on pavement (New Hampshire Department of Environmental Services, 2016). Finally, some deicing agencies have implemented the use of abrasives like sand to improve traction along roads and reduce salt application (Jones *et al.*, 2015; Fortin *et al.*, 2014).

The amount and type of road salt applied varies between states, as each has their own unique highway system, winter maintenance policies, and Total Maximum Daily Load (TMDL) concentration targets (Dao *et al.*, 2019; S. Rittay, Personal Communication, March 28, 2022; A. Roller, Personal Communication, March 31, 2022). TMDL, or Total Maximum Daily Load, is a regulatory term in the U.S. Clean Water Act that “calculates the maximum amount of a pollutant allowed to enter a waterbody so that the waterbody will meet and continue to meet water quality standards for that particular pollutant” (USEPA, 2022); the chloride concentration threshold for aquatic life in impaired waterbodies that TMDLs are assigned to is 230 mg/L (Siegel *et al.*, 2007; Dugan *et al.*, 2017). Although there are differences in each state’s winter maintenance program, most rely on weather variables like snowfall and road temperature to determine maintenance activities like road salt application (Dao *et al.*, 2019; S. Rittay, Personal Communication, March 28, 2022; A. Roller, Personal Communication, March 31, 2022). Ultimately, a range of factors determine how much NaCl is applied at a given point, including: (1) the type of weather event, (2) event intensity, (3) pavement temperature (the colder the pavement, the more salt applied), (4) vehicle traffic conditions, (5) number of lanes on highway, and (6) width of highway lanes (S. Rittay, Personal Communication, March 28, 2022).

Once applied, Na⁺ and Cl⁻ ions enter into the environment through a number of ways, including: (1) splash and spray by highway traffic, (2) direct infiltration into soil and groundwater via runoff (3) drainage into water bodies via ditch, culvert, or pipe, and (4) transformation into

airborne salt spray by vehicle traffic and dispersion via wind (Figure 1)(D'itri, 1992; Dugan *et al.*, 2017; Lakoba *et al.*, 2020; Tiwari & Rachlin, 2018; Siegel, 2007). An alternative way to estimate how much saline runoff will enter the environment is the amount of impervious surface area (ISA) like roads, parking lots, and bridges that road salts are applied to. Elvidge *et al.* (2004) calculated that the total ISA% of the US (Alaska and Hawaii excluded) to be 112,610 km², nearly the size of Ohio. At the time of this study, it was estimated that ISA% within the US was expected to increase more than 15,000 km with the construction of new roads and buildings. Furthermore, Kaushal *et al.* (2005) noted that this increase will likely be concentrated in areas near surface waters like rivers and lakes as these are areas where urban growth is likely to occur. Recent data as late as 2019 estimates that the ISA% of the US to be 157,800 km², a 40% increase in almost 2 decades (Dewitz, 2019). The US Geological Survey's (USGS) National Land Cover Database (NLCD) is an important tool utilized in road salt studies because it tracks urban population growth that is then correlated with anthropogenic impacts on the environment (Xian *et al.*, 2019; Wickham *et al.*, 2020). Geographic areas of watersheds can be estimated using assessment units as accurate as 1m²; for example, the Chesapeake Bay Region's area was calculated at 262,000 km² (Wickham *et al.*, 2018).

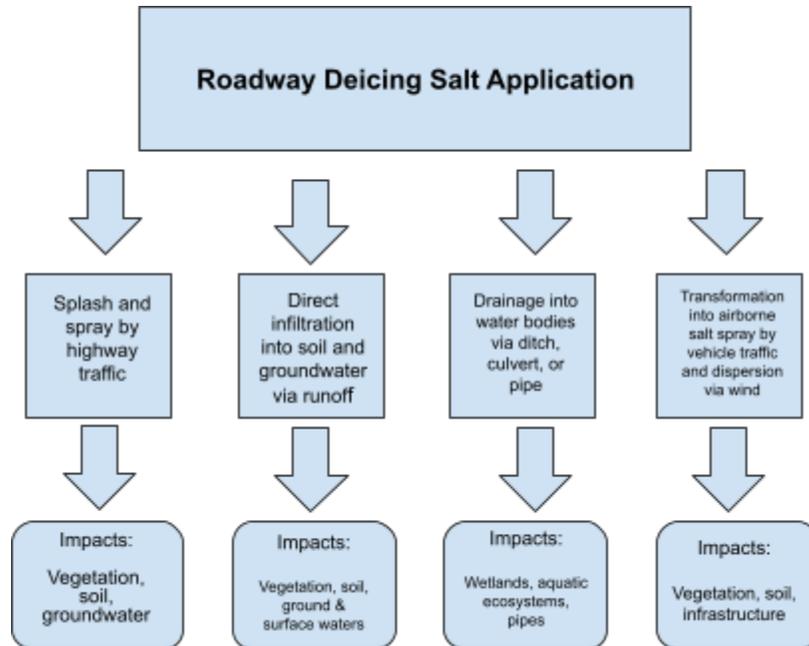


Figure 1. Diagram of road salt transportation pathways and subsequent impacts.

Efforts to Mitigate NaCl Pollution

One potential solution to road salt pollution is phytoremediation. Phytoremediation is the intentional use of hyperaccumulator plant species to remove various contaminants from soils (Alufasi *et al.*, 2020; Rippey *et al.*, 2022; Renshaw, 2021; Litalien & Zeeb, 2020).

Phytoremediation is becoming increasingly recognized as a way to remediate soils impacted by anthropogenic processes like road salt application because it is cost effective and environmentally friendly (Rippey *et al.*, 2022). Hyperaccumulators can be defined as any plant species that accumulate particular metals and/or metalloids in biomass at concentrations 100 to 1000 times greater than normal, and hyperaccumulation is a prerequisite for phytoremediation (Reeves *et al.*, 2007). It is important to note that phytoremediation capabilities are not solely dependent on the plant species; in fact, several factors within the soil like pH, organic matter (OM) content, CEC, moisture, and temperature influence the accumulation potential of plants (Alufasi, 2020). The first step in implementing this technology is identifying which plant species

are able to grow in the environment with the contaminated soil and accumulate a large portion of the hazardous element(s) in their biomass. In the context of this paper, phytoremediation of soil with elevated levels of sodium chloride or other salts can be done through using halophytes, commonly referred to as salt-loving plants (Renshaw, 2021; Litalien & Zeeb, 2020; Rippy *et al.*, 2022). Not all halophytes can be considered hyperaccumulators of salt however, as some deal with salt stress through exclusion via root filtration mechanisms (Renshaw, 2021). Recent studies by Rippy *et al.* (2022) demonstrate the ability of the Virginian-native cattail plant (*Typha latifolia*) to sequester a significant amount of Na⁺ and Cl⁻ ions in areas where deicing salts are applied. They estimate that removal of cattails after one season could sequester between 0.4% to 2.4% of the sodium and 0.8% to 14.3% of the chloride added to the Northern Virginia highway sites that they were grown in. One disadvantage to phytoremediation is the relative slowness of a plant to fully sequester pollutants, which can be due to shallow root penetration and low biomass production (Alufasi, 2020; Renshaw, 2021). In addition, the list of candidate halophytes for road salt extraction is further limited due to the fact that most plant species are dormant during the winter when salt levels are the highest (Rippy *et al.*, 2022). In response to these challenges, efforts have been directed to identify and study plant species with a high production capacity as the amount of pollutant removed is proportional to the amount of plant growth (Alufasi, 2020; Renshaw, 2021). *Cannabis sativa* L. has great potential in phytoremediation as it has rapid growth, a deep rooting system, and high tolerance to drought and heavy metals (Ahmad *et al.*, 2015; Husain *et al.*, 2019; Citterio *et al.*, 2003). Numerous studies have shown the potential of hemp to grow and tolerate soils contaminated with heavy metals. Husain *et al.* (2019) grew 6 different varieties of industrial hemp in 4 different soil types: 2 being from contaminated coal land mine soils in Pennsylvania and 2 from commercial potting soils. The results indicate that there was no significant difference in seed germination rate and plant height among all hemp varieties; furthermore, the concentration of nickel was 2.54 times greater in the leaves of hemp grown in coal mine soil. Hemp's ability to uptake heavy metals from soil is

partially due to the presence of stress tolerance genes (Ahmad *et al.*, 2015; Citterio *et al.*, 2003). A potential problem with growing hemp along roadsides for phytoremediation is that it is high enough to block vision, and most varieties are not suited to grow during the winter when phytoremediation efforts are most needed.

Alternative solutions to alleviate road salt damage to soils are through applying various amendments like gypsum and biochar. Biochar can adsorb and retain sodium ions, while gypsum can replace sodium ions with calcium ions. Both have been shown to help improve soil structure and make it more conducive to plant growth in agricultural settings. Additionally, gypsum can help leach excess sodium out of the soil, further reducing its concentration within the soil profile (Rippy *et al.*, 2022; Makoi & Verplancke, 2010).

While phytoremediation is often associated with plants, some definitions expand the concept to include the microorganisms linked with hyperaccumulator species like mycorrhizae and bacterial endophytes (Rippy *et al.*, 2022). Mycorrhizae are fungi that form a symbiotic association with the roots of certain plant species, whereas bacterial endophytes are beneficial plant bacteria that thrive inside certain plant species. Mycorrhizae have been shown to mitigate the osmotic stress and foliage burn in plants growing in saline soils by maintaining complimentary K^+/Na^+ ratios, increasing root water transport, and improving antioxidant capacity (Kushwaha *et al.*, 2020; Augé *et al.*, 2014). Similarly, bacterial endophytes can promote growth in saline soils by promoting the production of phytohormones and enzymes, improved micronutrient adsorption, and by activating resistance genes against plant pests and pathogens (Kushwaha *et al.*, 2020). Kears *et al.* (2019) isolated a strain of *Halomonas* and *Bacillus* endophytes that were found associated with halophytic plants in saline Utah soils and inoculated the roots of alfalfa seedlings grown in soil containing 1% NaCl. Their results show that the presence of these root endophytes stimulated the growth of the alfalfa seedlings whereas growth would be severely limited normally. Newer solutions to combat salt damage to roadside vegetation and agricultural crops in the form of biological applications such as

mycorrhizae and bacterial endophytes are certainly feasible and already occurring; however, further trials in the field are necessary to evaluate the extent to which these microorganisms can benefit salt-impacted plants (Abolfazli & Strom, 2021; Augé *et al.*, 2014; Kearl *et al.*, 2019; Kushwaha *et al.*, 2020).

One of the technologies state DOTs have implemented to lower the amount of road salt usage is in the form of liquid salt brine (Renshaw, 2021; Claros *et al.*, 2022; Duffin, 2021). Numerous studies have shown that usage of salt brine is more efficient and less environmentally damaging than traditional solid deicing chemicals (Claros *et al.*, 2022; Duffin, 2021). Salt brine applications can be via direct liquid treatment or a combination of brine and solids (Claros *et al.*, 2022). Following a treatment, a thin layer of brine is created between the pavement and snowfall, preventing ice from bonding. When applied properly, liquid brine can reduce solid salt usage by 23% and increase the ease of snow and ice removal by equipment (Claros *et al.*, 2022; Duffin 2021). Typically, salt brine is made from saturating salt until a 20%-25% NaCl solution is achieved; however, alternative methods exist to obtain liquid brine (Claros *et al.*, 2022). Fitch *et al.* (2008) was able to determine that by capturing saline overflow from salt storage ponds, VDOT could capture sufficient volumes of solution to meet the majority of its brine production needs.

Engineers have devised a variety of protection strategies in an effort to combat the ravages of salt on infrastructure. For example, all federal highways use epoxy-coated reinforcing steel, and bridges are now built with protective membranes on bridge decks, denser concrete, latex-modified concrete, and the application of surface sealants. While these improvements have hastened the rate of infrastructural deterioration, most highways are built to last about 20 years. In fact, the technology exists to construct roads that withstand saline runoff infiltration and the corresponding cracking and corrosion for 50 years, even in environments with harsh winter climates. The reason why road ways are not designed to this level of quality is a

matter of political judgment, as engineers and policymakers have determined that roads that last for 20 years are the most cost effective (D'itri *et al.*, 1992).

Economics

In order to fully understand the implications of roadway deicing it is necessary to discuss the economics of the practice. Economics helps us understand the positive and negative impacts of alternatives and the trade-offs involved so governments can effectively design environmental policies. In regards to road salt application, the costs can be differentiated between direct and indirect costs. Direct costs refer to the cost of the chemical deicing agent, equipment and labor to spread it, and maintenance costs. Indirect or “real” costs relate to the maintenance of infrastructure such as bridges and roadways, vehicle corrosive damage, damage to vegetation, soil and aquatic biota, as well as human health (D'itri, 1992; Vitaliano, 1992). Direct costs are relatively straightforward although it is important to note that the data reported to quantify these costs comes from public record and does not reflect the total amount of road salt applied from private citizens and businesses, which is estimated to be at least equivalent to the amount applied by the public sector (Lakoba *et al.*, 2020; Dindorf *et al.*, 2014). Indirect costs are much more difficult to quantify and vary depending on the fiscal value assigned to variables in economic models.

The average cost per pound of NaCl rock salt is \$56, although this number fluctuates depending on distance between salt mines and shipping facilities (U.S. Geological Survey, 2022; D'itri *et al.*, 1992). As of 2021, the total value of NaCl salt used or consumed was \$2.5 billion; highway deicing accounted for 42% of this, meaning that \$1.05 billion was spent just to acquire road salt (U.S. Geological Survey, 2022). VDOT has 10,800 vehicles for snow and ice removal and 2,500 crew members on payroll, although they do hire contractors to support winter operations. Another direct cost is the disposal of saline stormwater from salt-storage detention ponds when they overflow. In Virginia this cost was \$.13 per gallon, which totals to about \$7.8

million each year when the average yearly rainfall is accounted for. VDOT's total budget for the 2021-2022 winter season was \$211 million; in addition, they estimate that the average operations cost for a statewide snowstorm to be approximately \$11 million per day (VDOT, 2021). Collectively, the total annual cost for winter maintenance programs in the United States is estimated to be \$2.3 billion (Dao *et al.*, 2019).

In regards to indirect costs, the economist Donald Vitaliano (1992) estimates that for every ton of NaCl applied, there is a cost of \$800 for repair and maintenance of roads and bridges, vehicle corrosion costs, and loss of aesthetic value through roadside tree damage. He noted that bridges were especially susceptible to deterioration from road salt, and estimated that the United States spends "\$2.5 billion per year to repair and rehabilitate interstate and arterial bridges." It should be noted that values have changed since the time of his publication, as the EPA (2022) has stated that the cost to infrastructure damages alone is \$5 billion per year. Further, a study conducted by an environmental consulting firm calculated that for every ton of NaCl applied on bridges, there is a cost of \$1,460 to the environment and surrounding infrastructure (Dindorf *et al.*, 2014).

Increased chloride concentrations in the groundwater can accelerate galvanic corrosion and dezincification of private wells and water plumbing, affecting property values for both old and new homeowners. The deterioration of the public water infrastructure by road salts has caused many states to replace public wells and construct new municipal water lines, the latter of which cost the Town of Orleans in New York an estimated \$13.2 million (Pieper *et al.*, 2018).

One indirect cost that is difficult to quantify is salt damage to roadside vegetation. Although the loss of aesthetic value has been quantified to some extent, the real cost to replace and reestablish trees damaged by road salt was found to be \$10,000 per lane mile in the Adirondack region of New York (D'itri, 1992; Dindorf *et al.*, 2014). Farmland located in close proximity to roads where NaCl is applied are likely to have high soil salinity. Soil salinization degrades soil structure and ultimately impacts crop yield, which puts a financial burden on the

farmer. For instance, assume the farmer's total input cost is \$300/ha. If the market price and growing conditions are fair, the farmer realizes a gross return of \$600/ha, earning \$300/ha in profit. Now, if this same soil's salinity levels are increased by NaCl runoff, yield diminishes, resulting in a gross return of \$400/ha, leaving only \$100/ha in profit (Munns & Gilliam, 2015). In addition, among 65,000 acres of Colorado farmland analyzed over a 3 year period, the estimated loss due to decreased crop performance from soil salinization was \$4.3 million annually (Houk *et al.*, 2006). Although the value of fiscal damage varies, one farm in Canada was awarded \$100,000 for crop damage and loss of farmland value due to road salt damage (Casey, 2015). While NaCl runoff is not the only cause of the land salinization issue, it is important to recognize that it is a substantial anthropogenic source that contributes to this complex, global problem that costs an estimated \$27.3 billion annually (Qadir *et al.*, 2014).

Depending on the source, estimates of damage to infrastructure, vehicles, vegetation, human health and the environment due to road salt were found to range from \$803 to \$3,341 per ton of NaCl (Dindorf *et al.*, 2014). Assuming an annual application of 20 million tons, a conservative estimate would equal approximately \$16 billion, whereas an aggressive estimate would equate to approximately \$66 billion in annual damages.

So if the cost damages associated with chemical deicing are so high, why do we apply them in the first place? Certainly the cheapest cost alternative would be to not apply deicers at all; however, the cost implications of not applying them are even higher. Effective deicing reduces winter car accidents by approximately 88% and decreases the costs associated with these events by 10%. Furthermore, the snow and ice build-up by winter weather events would immobilize vehicle movement on roadways causing schools, businesses, institutions, and potentially entire cities to close, costing a total of \$300 to \$700 million per winter storm (Dindorf *et al.*, 2014). A cost-benefit analysis conducted by 2 independent consulting firms found that for every dollar of damage caused to vehicles, infrastructure, and the environment, there was \$6.30 to \$18.10 worth of benefit. While some would argue that this cost-benefit ratio is stronger today

with the improvements made to lower the adverse cost impacts of road salt, others claim that the steadily rising concentrations of Na^+ and Cl^- in the environment will have continuous compound damage that will outweigh the benefits (Siegel et al., 2007). With this in mind, society has made both an economic and moral choice to deice roads because the potential risks to transportation safety from not doing so would be too high (D'itri, 1992).

Road Salt Impacts

Vegetation & Soil

The application of NaCl is recognized as a major contributor to the decline of numerous roadside tree and grass species. While the majority of studies concerning NaCl effects on plant life pertain to roadside vegetation, field crop impacts have been evaluated as well. Furthermore, the overall health of roadside vegetation is strongly interrelated with the physical and chemical properties of the soils that facilitate their growth, which is also impacted by Na^+ and Cl^- , particularly sodium (Mills *et al.*, 2020). Accordingly, the decline in one often means the decline in the other; therefore, understanding how NaCl affects roadside plant species and soils is crucial as their presence serves as a filter between roadways and the broader environment where Na^+ and Cl^- ultimately ends up (New Hampshire Department of Environmental Services, 2016).



Figure 2. Photo of road salt damage to roadside trees. Retrieved from USDA Forest Service.

Although this section focuses on the effects that excessive Na^+ and Cl^- have on vegetation, it is important to recognize the functions these elements have within the plant as nutrients prior to reaching toxic concentrations. Interestingly, both elements have roles in the

operation of plant stomata which regulates gas exchange and influences photosynthesis. Chloride can act as the counterbalance to the potassium ion in operating stomata, whereas the sodium ion can actually substitute as the potassium ion in some plant species. Apart from this, it is the electrons supplied from the chloride atom that are used to break the bonds holding water molecules together during photosynthesis. Sodium can help support the osmotic activities in the cells of some - not all - plant species, which is why it is not considered an essential nutrient (Lowenfels, 2020). At excessive levels, NaCl has been shown to disturb the osmotic balance within plant cells, reduce root growth, and inhibit water adsorption (Environment Canada, 2001; Litalien & Zeeb, 2020; D'itri *et al.*, 1992). Symptoms usually present themselves as marginal and/or full necrosis on plant leaves and can result in the impairment and death of roadside plant species (Figure 2) (Kotheimer, 1967; Litalien & Zeeb, 2020). Furthermore, Na⁺ and Cl⁻ toxicity symptoms aren't always noticeable to the human eye, Li *et al.* (2022) demonstrated that a 100mM solution of sodium chloride reduces production of plant secondary metabolites such as caffeine by as much as 48%.

Roadside vegetation is exposed to sodium and chloride through direct contact on leaf and branch surfaces as salt is splashed or sprayed via wind and vehicle traffic, and through root absorption as NaCl runoff enters into the soil (D'itri *et al.*, 1992; New Hampshire Department of Environmental Services, 2016; Willmert *et al.*, 2018; Baltrėnas *et al.*, 2006). Plants that are in close proximity (1-10 m) to the road are at the highest risk of damage from salt pollution; however, airborne salt spray can reach plants as far as 500 m from the point of application (Chung, 1981). The factors that influence salt injury to vegetation include (1) temperature, (2) amount of salt applied, (3) soil saturation, (4) amount of runoff, and (5) growth stage and salt tolerance of the plant species (Kotheimer, 1967). Certain grass species like perennial ryegrass, fescue grass, and meadow grass have all been shown to be resistant to salt, with ryegrass having the highest resistance qualities (Baltrėnas *et al.*, 2006). Conversely, salt sensitive plant species have been damaged by chloride runoff concentrations as low as 30 mg/L (Environment

Canada, 2001). Decades-long salt application has resulted in the death of roadside tree species like the Paper Birch (*Betula papyrifera*) (Willmert *et al.*, 2018).

Due to their positive charge, sodium ions tend to remain in the soil as they are often adsorbed out of solution onto soil exchange sites (Siegel, 2007; Tiwari & Rachlin, 2018). Conversely, chloride ions are highly mobile partially due to their negative charge which prohibits adsorption onto soil mineral surfaces, and also because it does not readily enter into redox or biochemical reactions (D'itri *et al.*, 1992; Siegel, 2007). Consequently, chloride moves downward through the soil much more rapidly than sodium where it often reaches the groundwater.

NaCl runoff travels horizontally and vertically through and across the soil, the distance of each influenced by the soil characteristics and environmental conditions (D'itri *et al.*, 1992). As previously mentioned, the amount of exchange sites within the soil profile impacts the movement of sodium. For example, soils with high clay and/or organic matter content are more likely to bind to sodium and retain it within the soil profile (Siegel, 2007). Furthermore, these soils also have greater water holding capacity, which has also been shown to retain salt. A study by Toler & Pollock (1974) noted that silt-based soils have greater surface area, retained more water, and thus had higher salt concentrations than sand-based soils that lost moisture more rapidly. Another factor that affects the distance of road salt runoff and infiltration into the soil is the time of year. During the winter, when some soils are frozen, vertical movement is prohibited whereas lateral movement is enhanced significantly (D'itri *et al.*, 1992).

It is well known that NaCl is capable of deteriorating the structure of soil. This is caused by the sodium ion reacting with soil exchange sites, where it displaces other cations like magnesium, calcium, and potassium. Not only does the leaching of these cations - particularly calcium - impact soil structure, but it also negatively impacts plant fertility since these essential nutrients are no longer available for plant uptake (Willmert *et al.*, 2018). In addition, salt usually increases the alkalinity of soil, but has also been shown to mobilize H⁺ ions from soil exchange

sites and make the soil more acidic as well (Łuczak *et al.*, 2021; Löfgren, 2001). In regards to the latter, lowering of the pH below 5.5 increases the availability of aluminum, which can be toxic to plants (Delhaize & Ryan, 1995). Evelin *et al.* (2019) explains that high concentrations of Na^+ in the soil interferes with K^+ uptake by plant roots. In addition, salt can inhibit soil bacteria responsible for maintaining soil structure, which impacts erosion control and increases sediment runoff (New Hampshire Department of Environmental Services, 2016).

Effects of salt accumulation in irrigated desert soils is another major concern that has been studied extensively, although salinization of irrigation water and desert soils is separate from roadway deicing salt pollution. Salinization of soils can be attributed to both natural and anthropogenic processes (Renshaw, 2021). A soil is classified as saline when its electrical conductivity (EC) exceeds 4 dD/m at 25 C, which is approximately 40mM NaCl (Varga *et al.*, 2022). When compared to normal farmland, crops grown in salt-impacted soils like wheat, rice, sugarcane, and cotton experience a yield loss of 39%, 45%, 48%, and 63%, respectively (Qadir *et al.*, 2014).

Vegetation and soil along roadways act as a natural buffer area between pollutants like road salt and the environment. Tree mortality and vegetation loss is caused directly by impacts of salt on plants and decline in soil fertility (Willmert *et al.*, 2018). It is typical for roadside soils (<10m from road) to have different physical and chemical characteristics than soil 15m to 150m away from the road (Willmert *et al.*, 2018; source). High levels of salt in the soil have direct effects on soil chemistry, structure, and microbial populations as well as indirect effects on groundwaters and vegetation (New Hampshire Department of Environmental Services, 2016; Tiwari & Rachlin, 2018). As vegetation and soil degradation continues, it compromises the retention of pollutants in stormwater runoff to the environment (New Hampshire Department of Environmental Services, 2016).

Infrastructure

Damages to infrastructure such as bridges, buildings, roadways, vehicles, and pipelines have long been recognized as consequences of highway deicing practices (D'itri *et al.*, 1992; Vitaliano, 1992; Mahima *et al.*, 2018; Popoola *et al.*, 2014; Houska, 2007). Vehicle damage was one of the first aspects of road salt damage that caused car companies to improve the materials used in automobile manufacturing to better resist the deteriorating effects of salt (D'itri *et al.*, 1992). Infrastructure damage is caused by corrosion, which is a “physicochemical interaction between a metal and its environment which results in changes to the properties of the metal which may often lead to impairment of the function of that metal”. In other words, it is the degradation of a material by its reaction with the environment (Popoola *et al.*, 2014). Corrosive damage due to road salt is most prominent on bridge and roadways, although salt can travel great distances and damage buildings and underground pipelines as well (D'itri *et al.*, 1992; Houska, 2007; Mahima *et al.*, 2018; Betts & Boulton, 1993; Pieper *et al.*, 2018).

In order for corrosion to occur, the surface of a material must come into contact with water and air. This process is expedited when the water solution contains electrolytes, which is the case in deicing chemical runoff like NaCl. This is not the only way in which salt affects corrosion however, as the hygroscopic nature of NaCl allows it to pull water out of the air, making corrosion possible at lower humidity levels and at longer durations. In addition, chloride ions can deteriorate the oxide layer that protects the surfaces of metals susceptible to corrosion (Houska, 2007). There are several types of corrosion but the most damaging form that accounts for 90% of corrosive damage is crevice corrosion, which is a form of localized attack that occurs in narrow areas where oxygen access is limited such as in the mechanical joints of engineering structures (Betts & Boulton, 1993; D'itri *et al.*, 1992).

Bridges are recognized as the most vulnerable to road salt damage because they are subject to fluctuations in temperature due to exposed undersides and have thin layers of concrete covering the reinforcing steel in their decks. When road salt is applied to bridges, the aqueous solution that develops penetrates through the cracks in the surface resulting in the

breakdown of these concrete layers and of the steel bars critical for structural integrity. This translates to bridge deck flexing and the deterioration of concrete surfaces (Vitaliano, 1992). Salt bridge decks are estimated by the Transportation Research Board (TRB) to last only 15 to 18 years where salt is used, versus 100 years or more in no-salt or low-salt environments. Vehicles and structures with paint are also susceptible to saponification, which is a type of corrosive damage that results in paint separation from a surface (D'itri *et al.*, 1992). Deicing salt can penetrate soil to depths at which pipelines and other utilities are located, leading to their deterioration and impacting our drinking water. A study by Pieper *et al.* (2018) showed that high levels of chloride in water “increased galvanic corrosion and dezincification of plumbing materials, resulting in increased metal leaching and pipe wall thinning.”

Waterways

Deicing activities have led to the increased salinization of manmade and natural water systems (Rippy *et al.*, 2022; Siegel, 2007; Tiwari & Rachlin, 2018; Lakoba *et al.*, 2022; Abolfazli & Strom, 2021; Dugan *et al.*, 2017; Hintz & Relyea, 2019; Jackson & Jobbágy, 2005; Jones *et al.*, 2015; Valteau *et al.*, 2022; D'itri *et al.*, 1992). Dissolved Na⁺ and Cl⁻ ions in runoff are estimated to impact 37% of the contiguous drainage area in the US and contribute to the long-term salinization of waterways (Lakoba *et al.*, 2020). Freshwater salinization is the human-facilitated increase in the concentration of total dissolved solids within freshwater ecosystems and is often monitored by tracking conductivity levels in surface and ground waters within a watershed (Lakoba *et al.*, 2020; Fortin *et al.*, 2014; Siegel, 2007; Valteau *et al.*, 2022). Once dissolved, salt ions like chloride cannot be removed from the water using natural processes, which makes desalination of freshwater prohibitively expensive (Rippy *et al.*, 2022; New Hampshire Department of Environmental Services, 2016; D'itri *et al.*, 1992). Elevated Na⁺ and Cl⁻ levels in surface and ground waters negatively affect the water quality and functioning of aquatic ecosystems, having both immediate and long-term impacts to biotic community

structure, diversity, and productivity (Dugan *et al.*, 2017; Tiwari & Rachlin, 2018; Hintz & Relyea, 2019; Jones *et al.*, 2015; Valleau *et al.*, 2022). Current surface and groundwater salinization trends threaten the supply of potable water for human consumption over the next century (Kaushal *et al.*, 2005; Jackson & Jobbágy, 2005).

The vast majority of research regarding road salt runoff involves its entrance and transportation pathways within the environment. After road salt is applied, the dissociated sodium and chloride ions can flow directly into surface waters alongside roadways or move through the subsurface environment into the groundwater (Rippy *et al.*, 2022; Siegel, 2007). The extent to which NaCl enters surface and groundwaters depends on the amount of salt applied, climate, surface and subsurface conditions, soil characteristics, ISA%, and location of application within a given watershed (Kaushal *et al.*, 2005; Siegel *et al.*, 2007; New Hampshire Department of Environmental Services, 2016). Around lakes, ISA% greater than 1% have been correlated with increased chloride levels and 27% of large lakes within the United States have this feature (Dugan *et al.*, 2017).

Chloride levels in surface and groundwaters are the main indicator of salinisation from road salts (Lakoba *et al.*, 2020; Dugan *et al.*, 2017). Watersheds can retain up to 90% of chloride from road salt application (Kelly *et al.*, 2005, 2018; Novotny *et al.*, 2009). The concentration of chloride is usually attributed to the increasing ISA% from urbanization; however, the varying presence and flow rate of the streams that feed into larger waterways in a watershed also have an effect (Lakoba *et al.*, 2020). Typically, greater concentrations of chloride are observed during the winter when deicing chemicals are applied although recent research has documented chloride concentration increases in feeder streams during the summer (Jackson & Jobbágy, 2005). Variation in chloride trends during periods when deicing doesn't occur could be due to: (1) salt retention in the soil and groundwater, (2) presence and amount of road salt collection sites, (3) precipitation events, and (4) temperature (Lakoba *et al.*, 2020). Understanding the relationship between stream flow and ISA% on chloride levels is important

for making deicing management decisions. Few studies have evaluated the relationship between ISA%, stream flow, and watershed management on chloride concentrations over time (Lakoba *et al.*, 2020).

As NaCl drains through the soil, Na⁺ ions liberate calcium, magnesium, potassium, hydrogen, and aluminum ions from soil exchange sites, causing these cations to leach into groundwater (Siegel, 2007). Increased concentrations of these elements can lead to the death of numerous aquatic plant and animal species; furthermore, chloride toxicity has been found to increase when associated with potassium and magnesium (Siegel, 2007; Tiwari & Rachlin, 2018). Salinization of large water bodies like lakes can interfere with ecosystem processes, damage aquatic wildlife and plants, and change the composition of plant and animal species. The chloride concentration threshold for aquatic life is 230 mg/L (Siegel *et al.*, 2007; Dugan *et al.*, 2017). Documentation of ecological impacts from chloride concentrations ranging from the 100s to 1,000s mg/l have been documented (Tiwari & Rachlin, 2018). Cl⁻ ions from NaCl can prevent lake turnover by interfering with the thermal stratification of these water bodies. Thermal stratification refers to the change in the water temperature at different depths of the lake due to the density difference between warmer and cooler water. When too much chloride is present in a lake, the ions sink to the bottom layer where they accumulate and prevent mixing with the oxygen-rich top layer. Prevention of this mixing leads to low oxygen levels and the release of heavy metals and phosphorus from lake sediments, which can stress and eliminate aquatic life (Tiwari & Rachlin, 2018). A study by Abolfazli & Strom (2021) correlated the addition of road salt to increased flocculation within a lake ecosystem, resulting in the degradation of the benthic population in the waterbody.

Human Health

The annual input of NaCl for deicing practices contributes to the rising salinization levels in the world and has broad impacts on human health and civilization as a

whole. Once salt is applied, the saline runoff penetrates into the soil, eventually making its way into the groundwater where a large amount of irrigation and drinking water is pulled from (Siegel, 2007). Sodium's infiltration into the soil profile leads to the mobilization and leaching of other toxic elements into drinking water, and chloride has been shown to form complexes with hazardous elements causing the now soluble compounds to enter the groundwater as well (Bäckström *et al.*, 2004). As Na^+ and Cl^- concentrations in surface and groundwaters rise, the potability of our drinking water obtained from these resources is at risk. Over the past 20 years, Na^+ levels in the Potomac River have increased from 10 mg/L to 16 mg/L, with occasional surges above the EPA's advisory level of 20 mg/L for individuals with high blood pressure (Olivo, 2022).

The health of those dependent on the water supply, especially anyone with sodium-sensitive diets or medical conditions such as heart disease are at a higher risk in these circumstances. Excess sodium can lead to hypertension, a condition 25% of the adult U.S. population has that can lead to cardiac disease, renal disease, eye damage, and stroke (Siegel, 2007). The EPA's drinking water threshold for chloride is 250 mg/L, and although chloride in drinking water is considered non-toxic to humans, that does not mean it cannot cause health problems. High levels of chloride in groundwater can lead to corrosion of drinking water infrastructure, resulting in increased metal leaching into drinking water (Pieper *et al.*, 2018). Indeed, Na^+ and Cl^- going into the environment can cause other hazardous elements to leach into groundwater supplies as well. This has been apparent for a while now, as the 1972 *Science* publication by Feick *et al.* (1972) showed that Na^+ and Cl^- triggers mercury release in the bottom of waterbody sediments. Bäckström *et al.*, (2004) noted that higher levels of cadmium, copper, lead, and zinc - all toxic elements to humans - were present during the winter months. Their presence was strongly correlated with the electrical conductivity and chloride levels in the water sampled, both of which are a direct result of NaCl application. The mechanisms responsible for the increase of these elements by NaCl are ion exchange, lowered pH, and chloride complex

formation (Feick *et al.*, 1972; Bäckström *et al.*, 2004; McNaboe *et al.*, 2017). Out of 956 drinking water wells sampled from a township in southeastern New York, mean concentrations for Na⁺ and Cl⁻ were 30 mg/L and 62 mg/L, respectively. The highest concentrations for Na⁺ (860 mg/L) and Cl⁻ (1800 mg/L) were from wells in close proximity to the road. This shows that the distribution of salt in groundwater is uneven and influenced not only by ISA% but also various landscape factors like well distance and elevation relative to the road (Kelly *et al.*, 2018). Increased competition of adsorption sites and formation of complexes by higher levels of Na⁺ and Cl⁻ have also been correlated with elevated radium levels in aquifers and wells, often exceeding the EPA standard of 5 pCi/L. Further, mobilization of radon gas - which is recognized as the second leading cause of lung cancer - is influenced by NaCl runoff as the solubility of the harmful gas decreases with increasing salinity in an aquifer or well (McNaboe *et al.*, 2017).

The increasing salinity of soils could lead to a decline in crop production, which has long-term potential consequences to civilization if managed improperly. Increased salinization of soils - particularly agricultural soils - by natural and anthropogenic processes like roadway deicing pose a great threat to food security as growing populations will increase food demand. Increased salinization of agricultural soils contributed to the collapse of the Mesopotamia Civilization (Olivo, 2022). Furthermore, increasing salinity in freshwater ecosystems can threaten the stability of aquatic fish species that are a source of food (Siegel *et al.*, 2007; Tiwari & Rachlin, 2018).

Review of Biofilter Materials

Hemp

Cannabis sativa L. is a multipurpose annual plant species that has been used for thousands of years in various industrial, religious, and recreational practices. Commonly referred to as hemp, this plant can be observed growing in various parts all around the world as it can adapt to a wide variety of climates and soil types (Kumar *et al.*, 2017).

Each component of the hemp plant's stalk has unique properties that allow for different

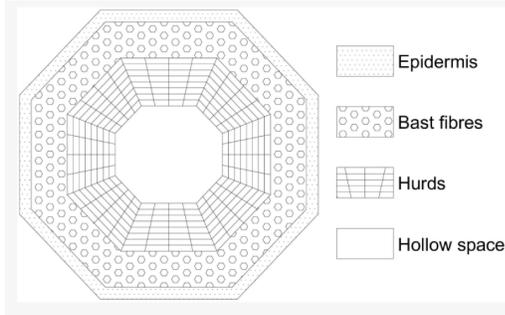


Figure 3. Diagram of cross section of hemp stalk. Obtained from Stevulova *et al.*, 2014.

applications (Figure 3). The inner core fibers - commonly referred to as “hurd” - are highly absorbent and can be used for animal bedding and construction materials. The contrasting bast fibers are long, sturdy, string-like in appearance and can be used to make paper and textile products (Stevulova *et al.*, 2014; Stevulova *et al.*, 2015). Hurd fibers represent the

majority of the hemp stalk by weight (60-80%), whereas the bast fibers represent the remaining 20%-40%. The majority of bast fibers are composed of cellulose (57%-77%), with a smaller portion of hemicellulose (9%-14%) and lignin (5%-9%). Hurd fibers consist of less cellulose than bast fibers (40%-48%) and relatively more hemicellulose (18%-24%) and lignin (21%-24%) (Stevulova *et al.*, 2014; Reh & Barbu, 2017; Nguyen *et al.*, 2009).

The anatomical structure of hemp allows it to absorb large amounts of water - up to five times its own weight (Stevulova *et al.*, 2014; Reh & Barbu, 2017). The high porous structure of hemp is one explanation for its absorption and adsorption capabilities (Stevulova *et al.*, 2014; Stevulova *et al.*, 2015; Zhao *et al.*, 2019).

Biochar

Biochar is a porous, charcoal-like product that is produced during the oxygen-limited pyrolysis of biomass from a variety of feedstocks (Rippy *et al.*, 2022; Zhao *et al.*, 2019; Tan *et al.*, 2017). The earliest use of biochar was likely by indigenous communities in the Amazon Basin, who incorporated it into the soil to increase fertility (Eden *et al.*, 1984). Biochar has been the focus of many studies as it offers a solution to combat global warming via carbon sequestration, can improve soil quality via increased CEC, sequesters contaminants, and a source of renewable energy production (Laird, 2008; Ogawa *et al.*, 2006; Vávrová *et al.*, 2022).

Its broad potential applications in agriculture and waste management can be attributed to its physical and chemical properties like pH, CEC, high porosity, and large surface area (Tan *et al.*, 2017; Zhao *et al.*, 2019).

Research by Zhao *et al.* (2019) demonstrated that the adsorption capabilities of biochar are determined by the type of feedstocks utilized, finding that sewage sludge-based biochar had better adsorption abilities than charcoals derived from agricultural and wood biomass waste. Wood biomass waste had the lowest adsorption abilities. In addition, the temperature conditions of the pyrolysis reaction also influenced the resulting biochar's adsorption abilities; high temperature led to increased porosity and surface area compared to lower temperatures.

Biochar amendment application to saline soils offers a potential method to sequester not just NaCl but several heavy metals (Rippy *et al.*, 2022; Zhao *et al.*, 2019). Awan *et al.* (2020) reported that biochar absorbed Na⁺ from irrigation water, lowering the sodium adsorption ratio (SAR) of soils and thus improving crop growth. Therefore, biochar's potential in mitigating the negative impacts of road salt runoff is tremendous as it can be utilized to directly sequester runoff and it can be incorporated into the soil during winter months until halophyte species are able to uptake Na⁺ and Cl⁻ into their biomass.

Materials and Methods

The materials used for this experiment were: a measuring bowl and cup, scale, scissors, a thermometer, zip ties, 5-gallon buckets, 3 plastic bins, 4 environmental containment socks, biochar, hemp hurd, and hemp fiber. Also, water was gathered from Slate Creek in Wilderville, Oregon. The environmental containment socks were supplied by New Pig Corp (Tipton, PA), which sells a wide variety of ECS designed to do a range of tasks like containing industrial spills and runoff from construction sites (Figure 4, Panel A). The original material was removed, replaced with experimental biofilters, then resealed. The biochar was obtained from an

amendment wholesaler in White City, OR; the biomass used to create the biochar was green waste in the form of tree cuttings from a nearby arborist. The hemp hurd was donated by Old Dominion Hemp, a Virginia-based distributor of high-quality hemp fibers for small animal bedding (Figure 4, Panel B). The hemp bast fiber was sourced from Gordon Jones at the Southern Oregon Research and Extension Center, Central Point, OR. Biochar was selected as a potential NaCl binder because of its high porosity, high surface area, and surface charge (Zhao *et al.*, 2019). Hemp hurd and bast fibers were selected due to their high absorbency and porosity (Stevulova *et al.*, 2014; Stevulova *et al.*, 2015; Cigasova *et al.*, 2014; Reh & Barbu, 2017; Nguyen, 2009; Yang *et al.*, 2019; Vávrová *et al.*, 2022) and because *Cannabis sativa* L. is a good candidate for phytoremediation (Husain *et al.*, 2019; Ahmad *et al.*, 2015; Alufasi *et al.*, 2020; Citterio *et al.*, 2003).



Figure 4. Photos of Experimental Materials. Panel A) Environmental Containment Socks, Panel B) Hemp Hurd, Panel C) Mix of Biofillers, Panel D) Weighed Hemp Hurd, Panel E) Weighed Biochar, Panel F) Weighed Hemp Bast Fiber

A scale was used to weigh out treatments (Figure 4, Panels D-F). Table 1 details the treatment and corresponding weight.

Table 1. Weights of Biofilter Sections for Control and Treatment Groups

Treatment Number	Biofilter	Weight
1A, 2A	Biochar	585 g
1B, 2B	Hemp Hurd	250 g
1C, 2C	Hemp Bast Fiber	152 g
1D, 2D	Biochar, Hemp Hurd, Hemp Bast Fiber	172 g 55 g 55 g

In preparation for the experiment, zip ties were used to divide the 4 ECS into 2 sections for 8 treatments. Biofilter materials were weighed and placed into a corresponding section of an ECS. For sections 1D and 2D, biochar, hemp hurd, and hemp fiber were mixed thoroughly then added (Figure 4, Panel C). Of these 8 treatments, half would be placed into water without any NaCl added (control group) and the other half would be placed into a 100 mM solution of NaCl (treatment group); the treatment group was divided into 2 bins (sections 2A and 2B in 1 bin, sections 2C and 2D in the other) (Figure 5). The untreated control group was included to show treatment differences and determine the extent of sodium chloride binding by the four treatments.



Figure 5, Panel A. Control group (Sections 1A-1D) in bin.



Figure 5, Panel B. Sections 2C and 2D of treatment group in bin with NaCl application.

Samples were placed in a bin and soaked in the NaCl solution for 24 hours to ensure full hydration and complete binding (Zhao *et al.*, 2019) and a weighted 5-gallon bucket was placed on top of the ECS to fully submerge them in the solution (Figure 6, Panels A & B). A thermometer was placed in one of the bins to measure the temperature of the solution. After 24 hours, the ECS were removed from the bins and layed out to dry prior to shipping. The ECS and their contents were sent to Dr. Gregory Welbaum at 301-C Saunders Hall in Blacksburg, VA, to be analyzed. Upon arrival, the ECS were still wet so Dr. Welbaum placed them in a drier at 60 °C for 48 hours. The samples were then sent to the soils lab at NC State University because the Virginia Tech Soils Lab now processes soils only.



*Figure 6, Panel A.
Sections 2A and 2B of
treatment group in NaCl
solution.*

*Figure 6, Panel B. Weighted
5-gallon bucket placed on
treatment group.*

Treatments 1A and 2A were labeled by the NC lab as ‘combustion/thermal-by-products’ and given the waste code ‘CSO’ which means ‘ash, mixed or other’, whereas treatments 1B, 2B, 1C, 2C, 1D, and 2D were labeled as ‘non-composted raw materials’ and given the waste code ‘NCR’ which means crop residue. Prior to the analysis, a subsample (~250 cm³) was weighed (Mettler PM4800; Mettler-Toledo, Hightstown, NJ), dried overnight (12-24 hr) at 80 °C, reweighed, and ground with a stainless steel grinder (Intermediate Wiley Mill; Arthur H. Thomas Co.; Philadelphia, PA) to pass through a 20-mesh (1-mm) screen (adapted from Hoskins *et al.*, 2003). Samples were wet-ashed using an open-vessel HNO₃ microwave digestion system (MARS & MDS2100 microwaves; CEM Corp.; Matthews, NC) (Campbell and Plank, 1992). A 0.5- g, dried/ground aliquot of sample was digested in 10 mL 15.6N HNO₃ for 5-30 minutes in a microwave, and then the prepared sample volume was brought to 50 mL with deionized water prior to measurement. After ashing, total Na⁺ concentration was determined with an inductively coupled plasma (ICP) spectrophotometer (Optima 3300 DV ICP emission spectrophotometer,

Perkin Elmer Corporation; Shelton, CT) at 580.982 nm following Donohue and Aho (1992) and adapted from USEPA (2001). Total concentration of chloride was determined by the thiocyanate displacement method (Zall *et al.*, 1956; Skalar Analytical 1995b) with an autoflow spectrophotometer analyzer (San++ Segmented Flow Auto-Analyzer, Skalar Instruments; Breda, The Netherlands) following a deionized water (1 g/25 mL), 30-minute extraction on a reciprocating shaker (Wrist Action Model 75; Burrell Corp. Pittsburgh, PA) (McGinnis *et al.*, 2013).

Results & Discussion

The most effective biofilter was biochar, followed by the mix, bast fiber, and finally hemp hurd (Figure 7 & 8). Biochar was the only biofilter that exceeded 10,000 mg Na⁺/kg and absorbed almost 20,000 mg Cl⁻/kg (Figure 7). Na⁺ and Cl⁻ ions captured by biochar were 12,700 mg/kg and 19,800 mg/kg, respectively (Figure 7). The mixture of biochar and hemp fibers was second in Na⁺ and Cl⁻ sequestration after the biochar-only section, with 8,890 mg/kg and 13,600 mg/kg, respectively (Figure 7). Hemp bast fibers sequestered 8,590 mg Na⁺/kg and 12,000 mg Cl⁻/kg (Figure 7). The hemp hurd biofilter absorbed the least Na⁺ (6,920 mg/kg) and Cl⁻ (10,500 mg/kg; Figure 7). The relatively high adsorption capabilities of both hemp fibers are impressive as neither underwent any chemical modification or pyrolysis process. High adsorption is likely due to hemp fibers' high porosity level and surface reactivity (Zhao *et al.*, 2019). The superior

Na⁺ and Cl⁻ sequestration abilities of the bast fibers could be because it has a higher ratio of cellulose to hemicellulose and lignin, and suggests the properties of cellulose facilitate ion binding (Stevulova *et al.*, 2014; Reh & Barbu, 2017; Nguyen *et al.*, 2009). Additional trials with different varieties of hemp bast fiber would be interesting to see if there is a specific variety that is better suited for NaCl sequestration; if possible, varieties should be selected based on a higher ratio of cellulose to hemicellulose and lignin.

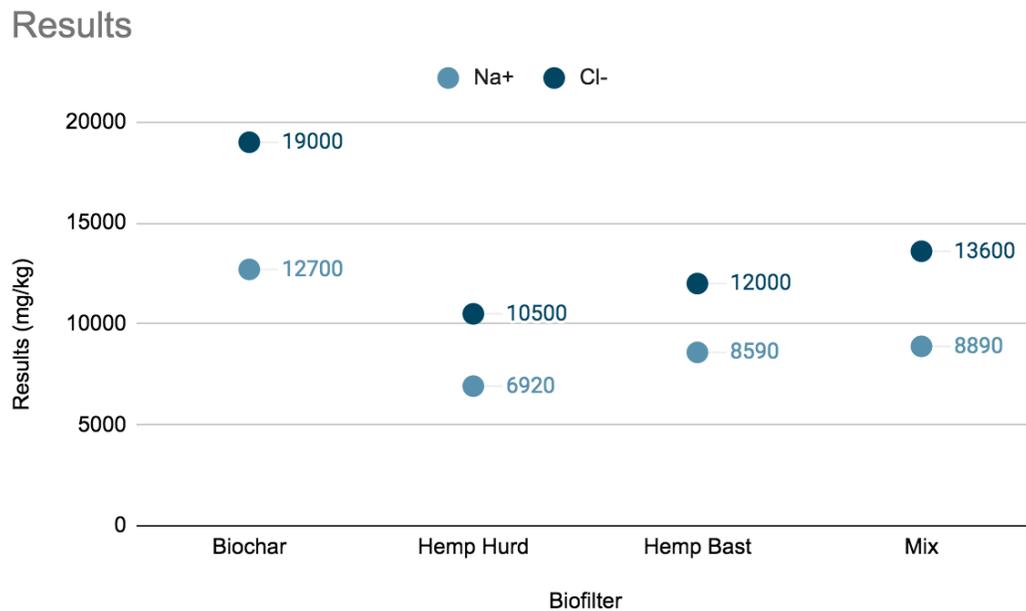


Figure 7. Analysis results. Values are expressed in mg/kg on a dry-weight basis.

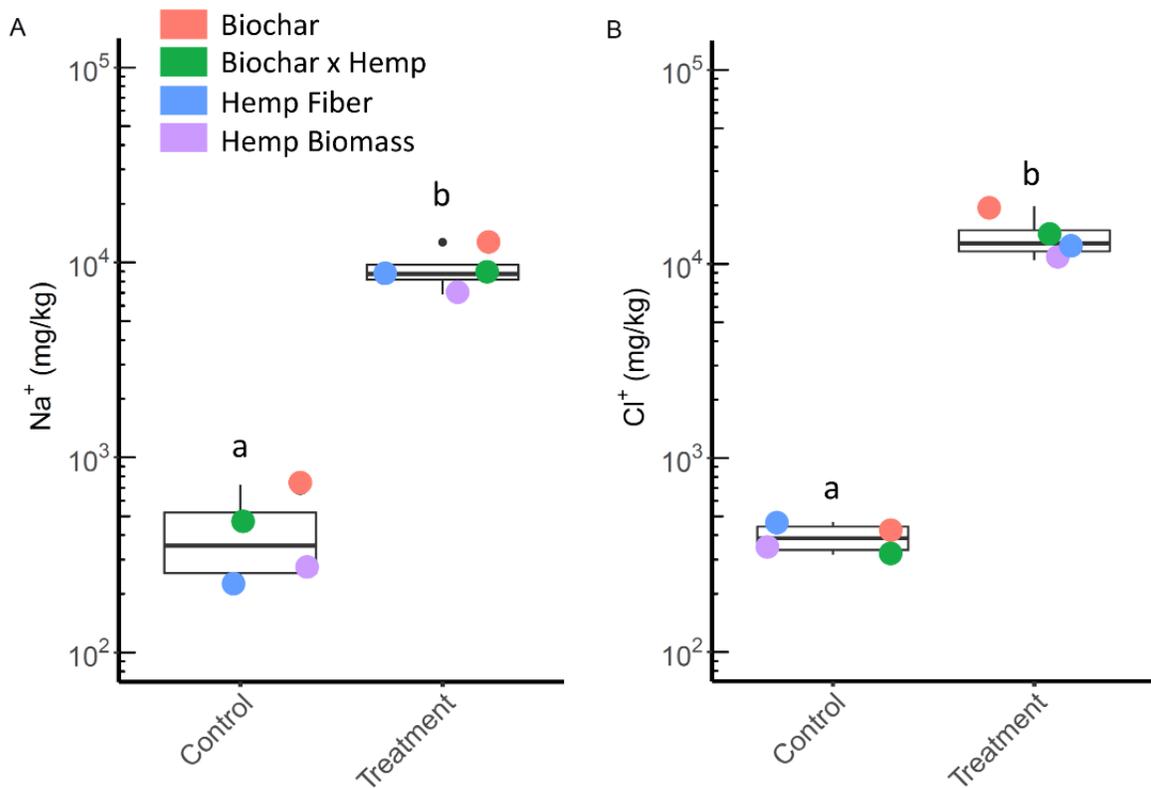


Figure 8. Graph showing comparison of ECS biofillers ability to bind to sodium and chloride particles. Panels A & B show the results from both groups for Na^+ and Cl^- , respectively. "Control" indicates ion content from ECS not exposed to NaCl and corresponds with the "a" on both panels, whereas "Treatment" indicates ion content from ECS exposed to NaCl and corresponds with the "b" on both panels. The colored dots correspond to the results from a specific biofiller treatment. The box around dots represents the average variation amongst ECS treatments, as the lab took multiple samples from each section of ECS. Note that hemp hurd (sections 1&2 B) is labeled as "Hemp Biomass", hemp bast fiber (section 1&2 C) is labeled as "Hemp Fiber", and mix (sections 1&2 D) is labeled as "Biochar x Hemp". Adapted from Rippy et al., 2022.

Another way to state the results is by stating how much Na^+ and Cl^- 1 kg of a given biofilter can bind to by weight. Binding capabilities of 1 kg of: biochar is 12.7 g Na^+ and 19.8 g Cl^- , hemp hurd is 6.92 g Na^+ and 10.5 g Cl^- , hemp bast 8.59 g Na^+ and 12 g Cl^- , mix is 8.89 g Na^+ and 13.6 g Cl^- . From this an estimation of how much road salt runoff an ECS can filter

before it is theoretically saturated with Na⁺ and Cl⁻ ions can be determined assuming a 100 mM solution of NaCl. Sodium filtration capabilities of biochar, hemp hurd, hemp bast, and mix are 12.7, 6.92, 8.59, and 8.89 liters, respectively. Chloride filtration capabilities of biochar, hemp hurd, hemp bast, and mix are 19.8, 10.5, 12, 13.6 liters, respectively. Table 2 lists how many liters various biofilters could theoretically filter in novel applications of ECS in areas with NaCl runoff. Based on the literature, estimations of NaCl concentrations (mg/l) of runoff are used to approximate how many liters ECS could filter (Smith & Granato, 2010; Bennett & Linstedt, 1978; Fitch *et al.*, 2005). Furthermore, because the full volume of ECS were divided to ensure there were enough sections for a control and treatment, an estimation of 10 kg of biofilter per ECS is assumed.

Table 2. Amount of NaCl runoff various 10kg ECS could filter in liters. Multiple concentrations of NaCl in runoff were listed based on literature findings. Filtration values were calculated by averaging the amount of Na⁺ and Cl⁻ an ECS could filter based on analysis results.

Biofilter	NaCl 1,000 mg/l	NaCl 5,000 mg/l	NaCl 10,000 mg/l	NaCl 20,000 mg/l
Biochar Filtration	323.8	64.8	32.4	16.2
Hemp Hurd Filtration	174	34.8	17.4	8.8
Hemp Bast Filtration	207.4	41.5	20.8	10.4
Mix Filtration	224.5	44.9	22.5	11.2

ECS containing biochar have the greatest potential to filter NaCl runoff. Assuming the lowest runoff concentrations of 1,000 mg/l NaCl, a 10kg biochar ECS could filter 323.8 liters (~85 gallons) before maximum binding capacity would be reached. In comparison, the highest

concentration of 20,000 mg/l would require the same ECS to be replaced after 16.2 liters (~4 gallons) of runoff.

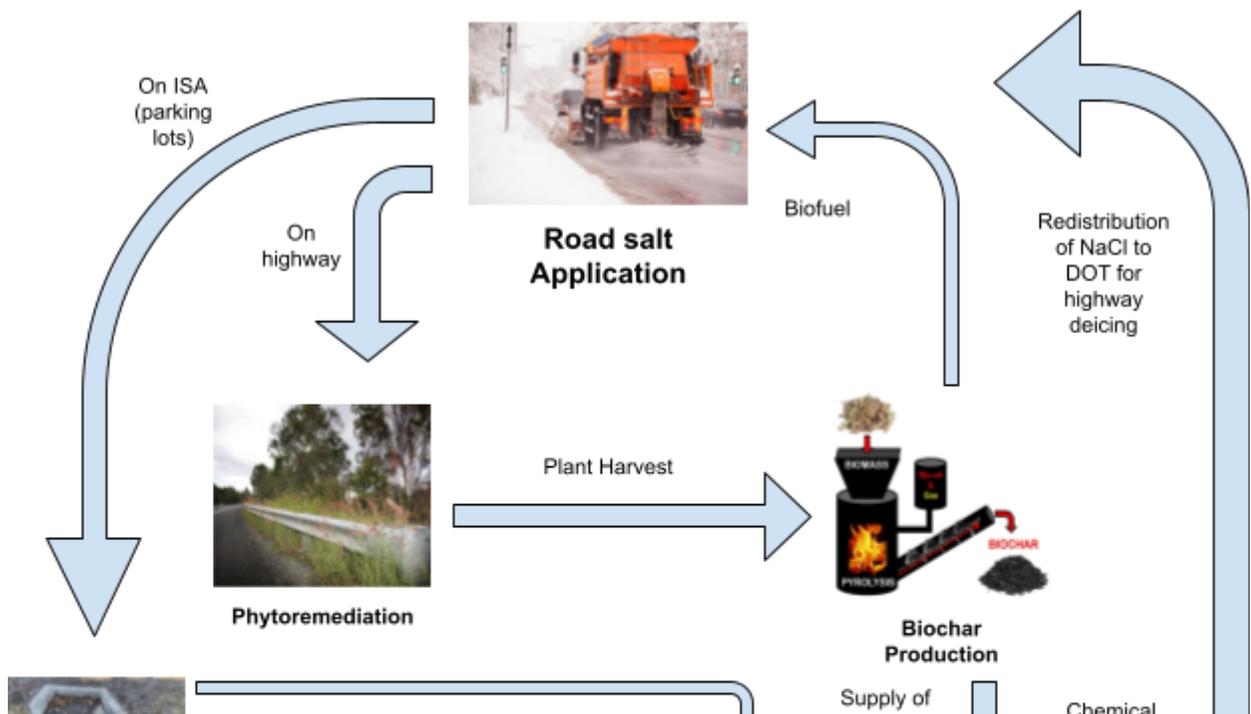
System to Reclaim and Reuse NaCl

In order to solve the complex issue of road salt pollution, a multifaceted effort is required. A potential system to sequester a portion of saline runoff using an approach that utilizes phytoremediation of road salt-affected soils in combination with use of ECS in strategic locations where phytoremediation cannot be implemented or totally effective (Figure 9). Taking the results of this experiment into account, biochar would be the best choice for biofilter selection in ECS. Cattail (*Typha latifolia*) had the greatest potential for phytoremediation in wetland areas of Virginia, with Rippy *et al.* (2022) estimating that adult cattail plants can potentially sequester .4%-2.4% of sodium and 2.6%-14.3% of chloride that enter VDOT detention basins over the winter season. Additional candidate halophytes for phytoremediation of saline soils boarding highways should be selected based on salt tolerance, height, biomass production, and hardiness (Rippy *et al.*, 2022). Moreover, pyrolysis of harvested halophytes used in phytoremediation along roadsides could serve as the source of biochar in future ECS applications (Figure 9). ECS could be collected, salts removed, and recharged ECS redistributed by a 3rd party service so salts in commercial parking lots could be recycled. NaCl could potentially be reclaimed from used biochar through reverse osmosis or other industrial scale processes, allowing road salt to be recycled back to highway agencies, reducing salt acquisition expenditures.

It is reasonable to estimate a parking lot with 25 spaces for vehicles to be 7,260 square feet (Holland, 2014). Utilizing a rainfall intensity calculator through the USGS, 1" of rainfall in a parking lot with an area of 7,260 square feet would generate 4,525 gallons of runoff. If a 5' long ECS containing 10 kg biochar were placed in front of parking lots drains experiencing NaCl runoff of 1,000 mg/l, 53 would be needed to fully sequester the entire amount of runoff. If the

saline concentrations were 5,000 mg/l, 10,000 mg/l, 20,000 mg/l, the amount of 10 kg biochar ECS needed to completely filter the runoff would be 265, 526, 1,131, respectively. Therefore, it is unlikely to assume ECS could prevent 100% of dissociated Na^+ and Cl^- ions in runoff from entering parking lot drains if concentrations are high enough, which is likely to occur in parking lot settings where volume is low and salt application amounts are high. This does not mean their application would not provide benefits to the environment. Improvements are certainly possible as future experiments could yield a biofilter and/or mix of biofilters that have greater binding capacity than demonstrated in this experiment.

VDOT maintains 57,000 miles of roadway, the majority of which experiences some form of deicing treatment over the winter season (Fitch *et al.*, 2005). Phytoremediation in soils bordering these roadways offers an affordable solution that also offers supplementary benefits like increasing soil organic matter and improving soil structure (Rippy *et al.*, 2022). In the sites that grew *Typha latifolia* for phytoremediation in the Rippy *et al.* (2022) study, a range of 54 to 123 kg Na^+ and 178 kg to 407 kg Cl^- was assimilated over the winter season. If applications were expanded to more sites, the compounding of sequestration amounts over time is hard to underestimate.



Future Work

The ECS technology has great potential for protecting paved and environmentally sensitive areas near roads and parking lots; however, future research needs to be conducted to better understand road salt runoff as well as the potential solutions to this issue. In regards to the ECS technology, practical applications in parking lots are needed to gather field data about performance, such as placing them in front of parking lot drains such as in the Rippy *et al.*

(2022) study (Figure 9). Certain parameters like the flow rates of water through the ECS and the amount of NaCl sequestered are needed to quantify the volume of runoff that could be purified. From this determinations could be made on how many ECS would be required to filter runoff from a given area and how much biofilter would be needed. Multiple trials comparing different types of biofilters replicated several times each are needed. The size and shape of the sock itself could potentially be optimized.

Socks in Fig. 9 were small in diameter for ease of

handling and knotted to compare different binding materials in the same environment. The weight of larger saturated ECS from field applications could impede their implementation

effectiveness. Trials with smaller and larger diameter and length ECS are needed to develop the technology. ECS placed in front of drains will likely never be fully submerged as they were in this experiment, so smaller ECS could have greater binding potential relative to larger ones.



*Figure 10. An ECS positioned in front of parking lot drain at Virginia Tech CRC. Further experiments that place ECS in strategic areas as seen in this photo are needed to better comprehend this technology's ability to filter saline runoff. Adapted from Rippy *et al.*, 2022.*

Deploying several ECS in depth would be predicted to increase salt binding capacity. Samples should be taken from multiple locations of each ECS as the inner and/or upper areas of a biofilter may not get as saturated as outside areas.

In regards to the biochar, trials with alternative biomass sources should be tested since research has shown that adsorption capabilities of biochar vary depending on the biomass used (Zhao *et al.*, 2019; Tan *et al.*, 2017; Kamarudin *et al.*, 2021). Studies have suggested that hemp and its residual biomass post-processing could be used for other purposes like the production of biochar (Vávrová *et al.*, 2022; Kumar *et al.*, 2017). Further trials that test the Na⁺ and Cl⁻ binding capabilities of biochar produced from hemp fibers would be interesting to see if a superior form of biochar is possible. Out of four different biochars made from wheat straw, lodgepole pine, kentucky bluegrass, and hemp, it was hemp biochar that was shown to adsorb the highest amount of sodium (923 mg/kg) (Awan *et al.*, 2020).

Barriers to Future Research and Applications

One of the issues that could impede research on identifying potential NaCl sequestering biofilters is finding a laboratory to test materials. For example, in the Rippy *et al.* (2022) study, no lab agreed to analyze the diatomaceous earth samples that were obtained. Also, proper description of a sample is critical to obtain accurate results, placing responsibility on both the researcher and the laboratory to properly label the biofilter material. The cost to send samples and have them analyzed by a laboratory can also represent a barrier in the absence of appropriate funding.

Time frame constraints to conduct the experiment also present a challenge, as the local weather conditions of the site influence when to deploy ECS technology in the field. Further unforeseen factors like interference of ECS by individuals or resistance to testing in parking lots by businesses also complicate matters.

Furthermore, adequate capital to expand the scope of ECS implementation to other areas with high saline runoff presents an issue. For example, a request to place ECS technology in strategic areas alongside highways in Oregon requires a permit application that costs \$500,000 (S. Rittay, Personal Communication, March 28, 2022; A. Roller, Personal Communication, March 31, 2022).

Conclusions

- 1) There is a great need to reduce salt pollution from deicing salt runoff; however, desalination of soil and freshwater is expensive and difficult. Further efforts are needed to find environmentally friendly and cost-effective solutions to prevent or mitigate NaCl runoff.
- 2) ECS have potential for sequestering Na⁺ and Cl⁻ ions from saline runoff depending on how they are deployed.
 - a) Utilization in areas with impervious surfaces like parking lots offers the most benefit for their usage.
 - b) ECS could be used in combination with phytoremediation efforts in a systematic approach to sequester road salt.
 - c) ECS should be placed in areas where salt is applied prior to precipitation/runoff events.
- 3) Identified potential salt biofilters materials to be used in future ECS applications.
 - a) Biochar showed the most potential in sequestration capacity of Na⁺ and Cl⁻ ions. A 1kg ECS with a biochar biofilter sequestered 12.7 g of sodium and 19.8 g of chloride.
 - b) Efficiency of biofilters is dependent on different factors like CEC, surface area, porosity.

- 4) ECS that are utilized in novel applications to prevent NaCl runoff into the environment should be flexible to bend around culvert drains, also light enough to handle once saturated.
- 5) Further research is required to optimize ECS technology for salt remediation by improving biofilters and finding ways to reclaim sequestered salt.

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Table 1. Weights of control and treatment sections.

Treatment Number	Biofilter	Weight
1A, 2A	Biochar	585 g
1B, 2B	Hemp Hurd	250 g
1C, 2C	Hemp Bast Fiber	152 g
1D, 2D	Biochar, Hemp Hurd, Hemp Bast Fiber	172 g 55 g 55 g

Table 2. Filtration capabilities of a 10kg ECS in various NaCl runoff concentrations.

Biofilter	NaCl 1,000 mg/l	NaCl 5,000 mg/l	NaCl 10,000 mg/l	NaCl 20,000 mg/l
Biochar Filtration	323.8	64.8	32.4	16.2
Hemp Hurd Filtration	174	34.8	17.4	8.8
Hemp Bast Filtration	207.4	41.5	20.8	10.4
Mix Filtration	224.5	44.9	22.5	11.2

Figure 1. Diagram of roadway deicing salt transportation and affects.

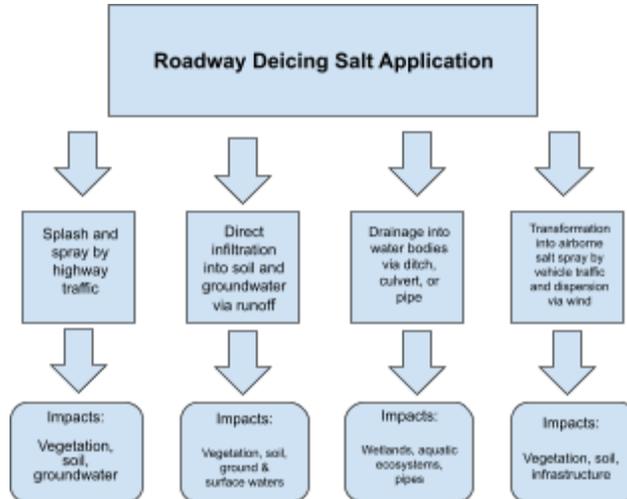
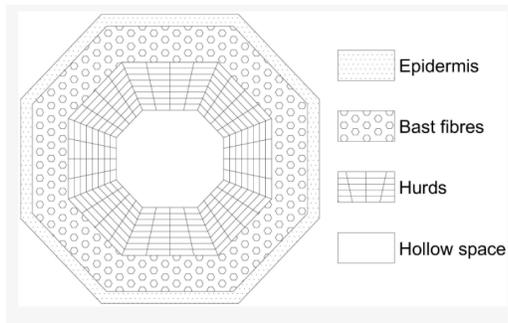


Figure 2. Photo detailing the negative impacts of NaCl on roadside trees.



Source: USDA Forest Service

Figure 3. Cross-section of a hemp stalk.



Source: Stevulova *et al.*, 2014

Figure 4. Photos of experimental materials.



Figure 5. Photos of ECS control and treatment groups in bins.



Figure 6. Photos of ECS treatment group placed and submerged in a bin containing 100 mM NaCl solution.



Figure 7. Test results detailing sodium and chloride binding abilities of various biofilters.

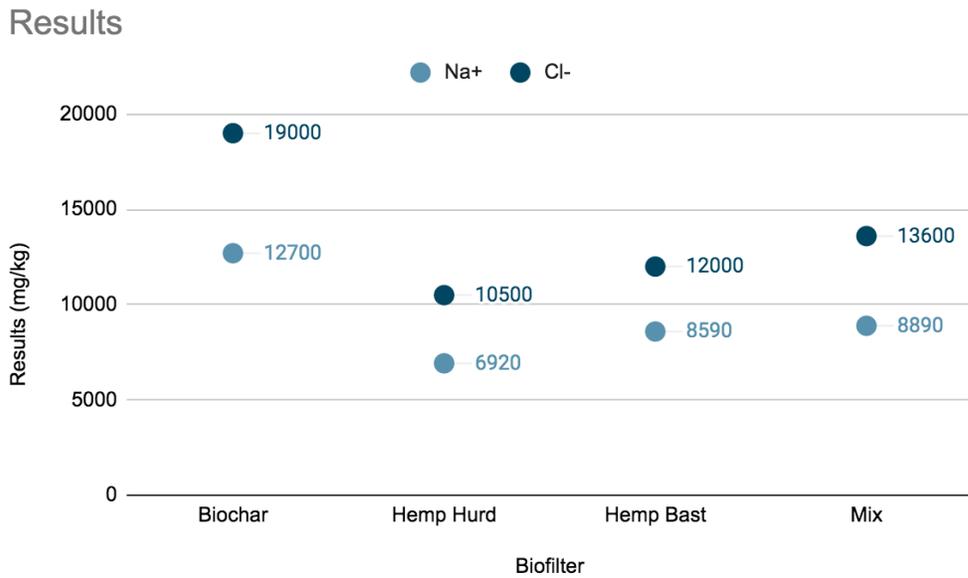
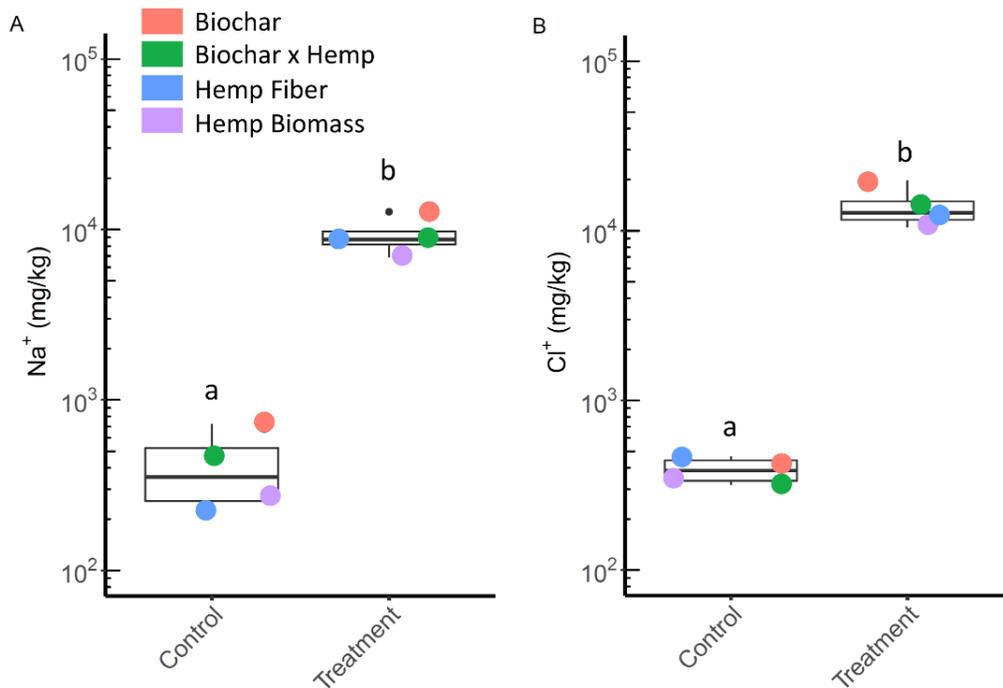


Figure 8. Graph comparing sodium and chloride binding abilities of various biofilters.



Source: Rippy *et al.*, 2022

Figure 9. Conceptual diagram of NaCl reclamation and reuse system.



Source: Rippey *et al.*, 2022

Figure 10. Photo of ECS in front of parking lot drain.

