Journal of Urban Ecology

Road salt impact on soil electrical conductivity across an urban landscape

Thomas P. Shannon,[1] Sam J. Ahler,[1] Alex Mathers,[1] Carly D. Ziter[2] and Hilary A. Dugan [1,*]

[1]Center for Limnology, University of Wisconsin-Madison, 680 N. Park St, Madison, WI 53706, USA and [2]Biology Department, Concordia University, 7141 Sherbrooke St W, Montreal, QC H4B 2A7, Canada

Present address of Thomas Shannon: Department of Biological Sciences, Florida International University, 11200 SW 8th Street, Miami, FL 33199, USA
*Corresponding author. E-mail: hdugan@wisc.edu

Abstract

Road salt application is a necessary component of winter road maintenance but comes with an environmental cost. Salts are transported via stormwater drainage or overland and soil throughflow to surface waterbodies, where excess ions create unfavorable or even uninhabitable conditions for freshwater organisms. Soils may retain salts during the process of overland and subsurface flow, thus acting as reservoirs that slow the transport of salt into freshwaters. Understanding the capacity and consistency of anthropogenic salt storage in urban soils may allow us to discover when and where deicing salt applications are most harmful. This article investigates the degree to which soils across a heterogeneous urban landscape retain salts. We measured the electrical conductivity (EC) of soils in an urban setting. Land covers included forests, grasslands, open spaces, low- and medium-density developments and along roadsides. We found that across land-cover types, soil carbon and porosity were correlated to EC in late summer, which suggests that pore space is an important and long-lasting reservoir for salt. In addition, more developed areas, had higher mean soil EC and greater EC variability within and between sites, with 75% of overall variance occurring within individual sites. We hypothesize that this within-site heterogeneity is driven by anthropogenic modifications to salt inputs and soil characteristics. The high EC variance in highly developed urban soils is a previously undiscussed phenomenon and highlights the fine-scale complexity of heterogeneous urban landscapes and the need for high-resolution sampling to accurately characterize urban ecosystems.

Key words: urban ecology, road salt, electrical conductivity, soil

Introduction

In the Midwest and Northeast North America, deicing salts are applied to roads in order to remove snow and prevent ice buildup by decreasing the freezing point of water on paved surfaces. The most commonly used deicing agent in the USA is rock salt (sodium chloride, NaCl), because it is effective in the temperature range at which ice-formation is common (0 to -10°C) and is relatively inexpensive (Fischel 2001). In colder temperatures, magnesium chloride ($MgCl_2$) and calcium chloride ($CaCl_2$) are more effective than NaCl due to their divalent cations, but their price-point deters heavy use. The desire to reduce and optimize salt use is widely held (Strifling 2018). In addition to the negative environmental and drinking water impacts of excessive salt application (Stets et al. 2018), the purchase and distribution of salts is an economic burden to local governments, and road salts accelerate rusting of personal vehicles and infrastructure, such as bridges and steel-

reinforced concrete (Shi et al. 2013). As a result, understanding the movement of salts from paved surfaces and their retention in terrestrial and aquatic urban ecosystems provides critical information for maximizing both environmental and economic value (Strifling 2018).

Many of the environmental consequences of road salt application have focused on aquatic environments, as road salt, once dissolved, will end up in shallow groundwater or surface waters (Kaushal et al. 2005; Novotny et al. 2008; Kelly et al. 2010; Dugan et al. 2017). While a large proportion of road salts are flushed directly to aquatic systems via storm sewers, a portion can end up in terrestrial soils due to misapplication, salt bounce, aerosolization from traffic, flooding and application in areas without drainage infrastructure. Like aquatic environments, road salts can have detrimental effects on terrestrial urban environments (Cunningham et al. 2008). Saline soils near roadways negatively affect the health and growth of urban trees through osmotic stress and ion toxicity (Bryson and Barker 2002; Czerniawska-Kusza, Kusza, and Duzyński 2004; Equiza et al. 2017). Airborne salts can also have a direct impact on plant tissues and especially leaves, when salts come in contact with plants through misapplication or aerosolization from traffic (Paludan-Muller et al. 2002). In soils, high concentrations of salts can mobilize potentially toxic heavy metals including lead, cadmium and mercury, which are known to bioaccumulate in terrestrial and aquatic food webs (Norrström and Jacks 1998; Bäckström et al. 2004; Schuler and Relyea 2018). In addition, excess salt concentrations promote nitrate leaching (Green, Machin, and Cresser 2008; Haq, Kaushal, and Duan 2018), and sodium ions defloculate (disperse) clay particles within the soil, which block pore spaces and limit water infiltration (Frenkel, Goertzen, and Rhoades 1978; Qadir and Schubert 2002). The resultant decreased hydraulic conductivity may lead to increased surface runoff, erosion and anaerobic soils.

The length of time road salt will stay in surface soils before being leached into waterways depends on soil water residence time, soil characteristics and hydrologic flowpaths (Kincaid and Findlay 2009; Snodgrass et al. 2017). Robinson, Hasenmueller, and Chambers (2017) distinguish between two modes of ion retention: 1, conservative retention, where ions are held in pore water, and transport is governed by the degree of pore-water retention and 2, nonconservative retention, where sorption to organic and mineral particles, or uptake by plants (Lovett et al. 2005) and microbes (Bastviken et al. 2007), slows ion transport relative to pore-water residence time. While chloride ions are generally considered conservative, with retention being regulated by water residence time (Svensson et al. 2007), this does not equate to rapid flushing of chloride from soils. One study examining ion retention in urban and rural soils found that sodium and chloride retention lasted at least 2–5 months following salt application and that ions exhibited longer retention times in rural soils. The authors suggested that ion retention was expected to lengthen under nonlaboratory conditions due to less frequent flushing events and more complex movement pathways (Robinson, Hasenmueller, and Chambers 2017).

With road salt application, the concentration of ions retained in soils will be driven by soil type, precipitation and time (Oswald et al. 2019). Time is influential in that a given soil may exhaust its nonconservative ion retention capacity over the course of a season or years of exposure (Robinson, Hasenmueller, and Chambers 2017). When soils act as salt reservoirs, the flow of salt to aquatic environments is slowed (Kelly et al. 2008; Kincaid and Findlay 2009). In the short term, this may be beneficial and dampen heavy ion loads during precipitation and melt events, but overall, soil storage masks the enormity of the environmental contamination, as often we gauge chloride contamination via water concentrations. In the long term, salt storage on the landscape may undermine management efforts and delay remediation, as load reductions in the form of lower application rates may be masked by slow leaching of legacy salt.

Little is known about the role that complex urban soil environments play in influencing ion movement and retention, despite urban areas being the major recipient of total road salt application. In order to gain a better understanding of road salt retention in urban environments, we examined whether road salt impacted soils can be identified through electrical conductivity (EC) measurements as a proxy for salinity. We hypothesized that soils in close proximity to impermeable surfaces subject to road salt application (roads, sidewalks, parking lots and driveways) would have higher EC than soils in less developed settings (forests and grasslands). Our study measured soil EC across urban land-cover types that span a gradient of development intensity to identify areas impacted by anthropogenic road salt loading through comparative analyses. This builds on previous studies of ion retention rates in soils under laboratory conditions (Kincaid and Findlay 2009; Robinson, Hasenmueller, and Chambers 2017) but adds the complexity of comparing the effects of soil and land use differences in a heterogeneous urban environment.

Methods

Sample collection

Soil samples from a range of land-cover classes were collected during the summer of 2015 within the city limits of Madison, WI, USA (Ziter and Turner 2018). Twenty sites were selected from each of five urban land-cover classes: forest, grassland, open space, low-density development and medium-density development (Fig. 1). These land-cover types encompass areas such as city parks (open space), residential backyards (low- and medium-density) and restored prairies. Site selection and soil characteristics are described in Ziter and Turner (2018). In Dane County, soils are mainly sandy-loam glacial till, and in Madison, soils are predominantly Alfisols, with some Mollisols being found in forest and oak–savanna vegetation and wet sedge meadows (Bockheim and Hartemink 2017). In 2018, 20 additional roadside sites were sampled along vegetated strips (depending on local nomenclature, colloquially known as the terrace, parkway/parking strip, verge, public access, easement, tree lawn, devil/hell strip, berm, apron or curb strip). Sites were distributed throughout the city and chosen to represent the diversity of major roads, and therefore, varied in terms of slope and stormwater management. The only hard criteria were the availability of a 45-m transect of soil and assurance of traffic-related safety for the field researchers.

At each non-roadside site, four soil samples were collected within a 30 m × 30 m quadrant ($n = 400$, 100 sites × 4 plots). At roadside sites, four soil samples were taken at 15-m inervals along a 45-m transect ($n = 80$, 20 sites × 4 plots). Transects ran parallel to the road or at the center of the road median. Soil from one forest plot was not measured for EC, and therefore total measured samples equaled 479. In both sampling years, soil samples were taken as a 10-cm deep soil core. Two additional samples were taken from a University of Wisconsin-Madison gravel parking lot that receives all snow removed from the campus. This site was chosen as a potential end-member for high
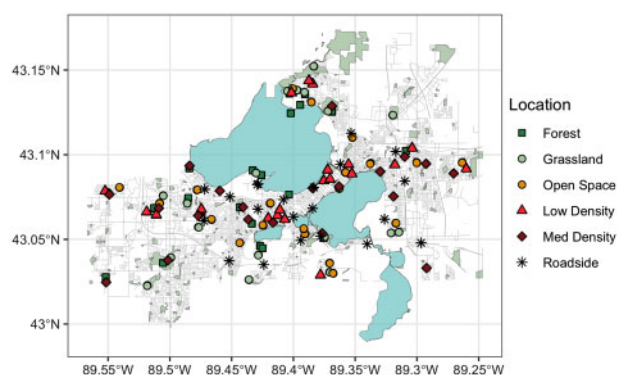
Figure 1: One hundred and twenty sampling sites across six land-cover classes in the city of Madison, WI, USA. The base map depicts lakes (blue polygons), roads (gray lines) and city parks (light green polygons)

road salt contamination. Samples were stored in a cool, dry area until processed.

Sample processing

Samples were air dried for 24–72 h, then put in a drying oven for 24 h at 60°C. Once dry, samples were sieved with a 200-μm mesh filter to remove gravel and other large nonsoil particles. If large solidified clumps had formed during the drying process, they were broken up and resieved. A homogenized 5-g subsample was then mixed with 10 ml of deionized water (<1 μS/cm) in a 30-ml high-density polyethylene plastic vial. The mixture was allowed to saturate for at least 15 min, with frequent resuspension of the settled solids. EC of the mixture was then measured with a Thermo Scientific Orion Star A222 portable conductivity meter (range, 1 μS/cm to 200 mS/cm). The meter was initially calibrated with a three-point calibration (100, 250 and 500 μS/cm), followed by a one-point calibration at the beginning of each processing period using a 500 μS/cm standard. Soil carbon percentage, bulk density and saturated hydraulic conductivities (K_s) for non-roadside sites were obtained from Ziter and Turner (2018) but were not measured for roadside sites.

Statistical and spatial analyses

A marginal analysis of variance (ANOVA Type III) and estimated marginal means were used to compare the difference between land-cover types (Lenth 2019). To assess within-site vs. among-site variance, the EC data were log-transformed, subset by land-cover class and fit with a linear mixed-effects intercept-only model with site ID as a random effect. Estimated variance between random-effect terms and within-group error variance was calculated with VarCorr in the lme4 package in R (Bates et al. 2015).

High-resolution impervious surface layers were not available for the city of Madison, WI. To calculate the likelihood of a soil being impacted by road salts, we manually digitized impervious surfaces (roads, sidewalks, driveways and parking lots) near sampling sites using a high-resolution Google Earth image overlain with city GIS data to ensure correct georectification. We then calculated the distance from each sampling plot to the nearest impervious surface using the sf package in R (Pebesma 2018). Given the error associated with GPS locations, any distance <1 m was set equal to 1 m, and distance values were log-transformed to conform to assumptions of normality. All statistical analyses and plotting were performed in R (R Core Team 2018).

Results

EC of soils from all sampling locations was generally between 154 and 435 μS/cm (5th and 95th percentile), although extremely high (1067 μS/cm) and low (3.4 μS/cm) conductivities were measured (Fig. 2). The range, variance and mean soil EC at forest and grassland sites were similar, with 90% of sites falling between 141 and 350 μS/cm (5th and 95th percentile, Fig. 2). Overall, grassland sites had the lowest mean EC (234 μS/cm) and were significantly lower than all other land-cover types ($P < 0.05$), with the exception of forest ($P = 0.98$) and road ($P = 0.14$) sites. The nonsignificant difference from road sites was due to the wide-ranging variability of roadside samples. Mean conductivities of soils in open space, low-density, medium-density, and roadside sites were not significantly different from one another ($P > 0.05$), although roadside sites did have the highest overall mean EC (330 μS/cm) in addition to the highest overall variance. The EC of samples taken from the snow storage lot were 1159 and 479 μS/cm. This upper value was the highest EC measured at any plot. The sample was a mixture of fine and coarse gravel.

Variance partitioning of EC at three levels (among land-cover classes, among sites and within individual sites) reveals that overall, 75% of the variance accounted for in EC is among plots within individual sites (Fig. 3). Partitioning of variance into two levels (among sites and within individual sites) independently for each land-cover class shows that at grassland, forest and open space sites, the majority of variance in EC was among sites. This relationship flipped for low-density, medium-density and roadside sites, where the variance was mostly within individual sites.

Soil EC was weakly correlated to soil properties of bulk density ($r^2 = 0.06$, $P < 0.001$) and percent carbon ($r^2 = 0.18$, $P < 0.001$) (Fig. 4a and b). There was no significant relationship between K_s and EC ($P = 0.22$). The strongest correlations with EC were seen at the scale of individual land-cover classes, with the exception of medium-density sites (Fig. 4b). Soil EC was also weakly negatively correlated to distance to the nearest impervious surface ($r^2 = 0.12$, $P < 0.001$) when all sites were pooled (Fig. 4c). Overall, forest sites were furthest from impervious surfaces (mean = 120 m), followed by grassland sites (75 m), open space (38 m), low-density sites (8.2 m), medium-density sites (6.3 m) and road sites (2.1 m) (Fig. 4c). Low-density and medium-density sites were all <30 m from impervious surface and showed no correlation to soil EC.

Ziter and Turner (2018) found that soils with higher percent carbon tended to have lower bulk densities and saturated hydraulic conductivity (K_s) (Fig. 5a and b). The mean bulk density at forest sites (1.21 g/cm³) was significantly lower ($P < 0.05$) than all other sites (1.30–1.36 g/cm³), and the upper limit of 1.75 g/cm³ was similar across all sites. Although there was a high variance among sites, K_s was highest in forest sites, followed by grassland sites. Open space, low density and medium density all had lower K_s. As the estimates of K_s were partially derived from bulk density measurements, the two metrics are highly correlated (Fig. 5c). To investigate the interaction of percent carbon and distance to impervious surface on EC, sites were split into three groups based on percent carbon. Among groups, sites with higher percent carbon had higher EC. Within each group, there is an influence of distance to impervious surface on EC, with sites further away from an impervious surface having lower EC (Fig. 5d). The magnitude of this effect is strongest at low percent carbon (2–3%).
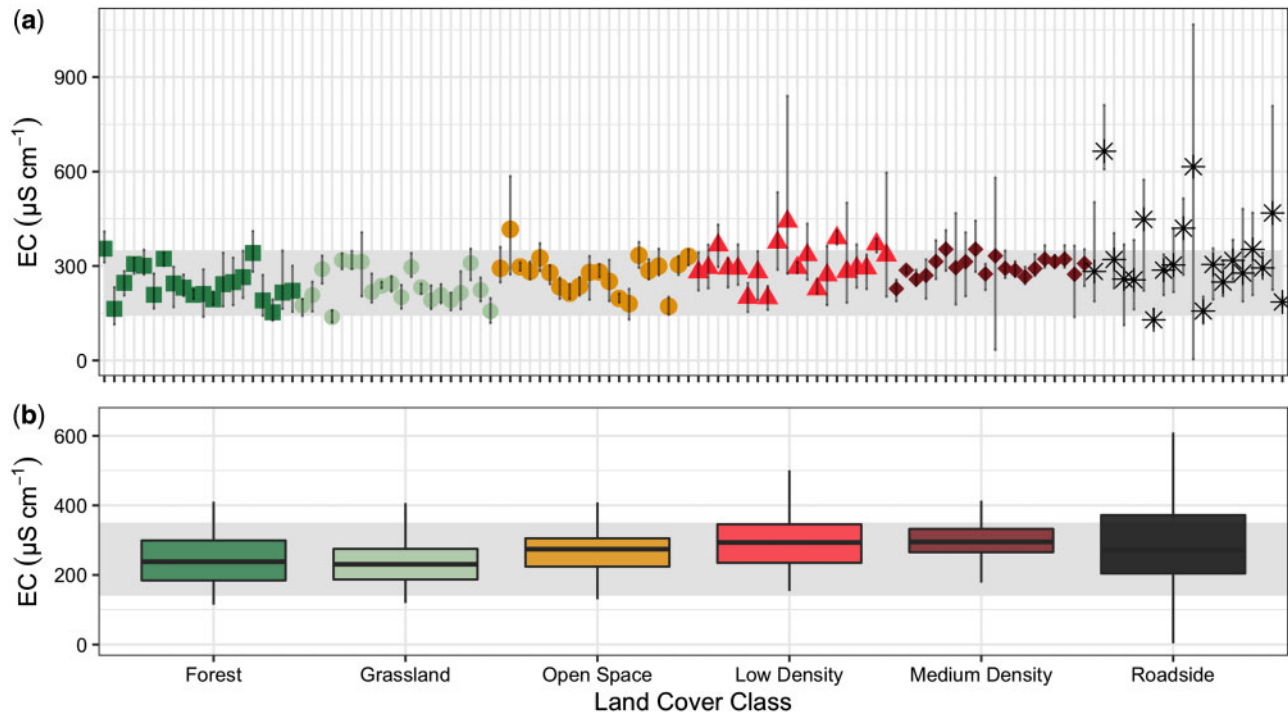
Figure 2: (a) Soil EC, as measured from a saturated soil-paste solution, at 479 plots (120 sites) across six land-cover classes. Gray polygon represents inferred natural background concentrations from 90% of forest and grassland sites (141–350 μS/cm). (b) Boxplots highlighting the median and 25th and 75th percentile of EC across the six land-cover classes

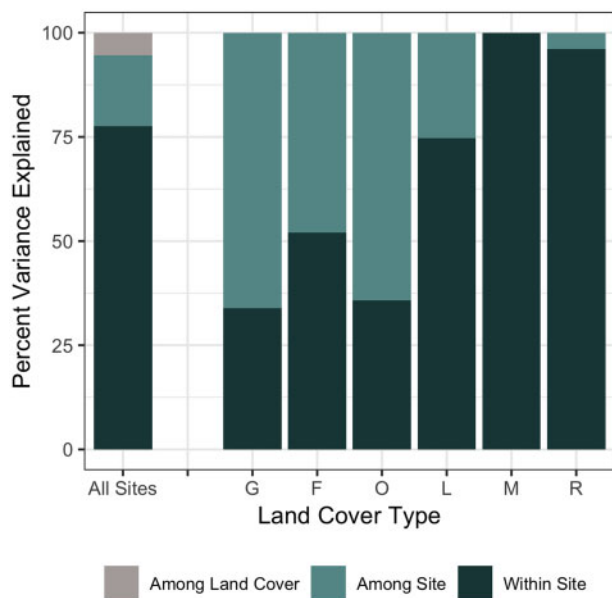**Figure 3:** Percentage of total variance in EC accounted for at all sites, as well as for each land-cover class. When all sites are pooled, variance is separated into among land cover, among site and within site. Only among site and within site are possible when land-cover classes are assessed independently. Land-cover classes include forest (F), grassland (G), open space (O), low-density sides (L), medium-density sides (M), and roadside sides (R)

Discussion

Natural variation in soil EC is due to soil characteristics such as porosity, permeability and underlying mineralogy, as well as the input of natural weathering products (Mawer, Knight, and Kitanidis 2015). To approximate the range of background EC, we consider our grassland and forest sites to be less anthropogenically manipulated than the open space, low- and medium-density and roadside sites, and although all urban soils may have elevated salt concentrations, we consider these sites representative of background EC across the city of Madison. From this premise, natural EC varies between ~141 and 350 μS/cm, and 11% of open space, 25% of low-density samples, 19% of medium-density samples and 35% of roadside samples exceed the upper threshold of 350 μS/cm (Fig. 2). Based on our hypothesis, we expected roadside sites to have uniformly high EC. Unexpectedly, 9% of roadside plots had EC below 145 μS/cm, the minimum EC of any natural soils sampled.

The right-tail skew of EC at low- and medium-density and roadside sites supports our hypothesis that sites with higher nearby impervious surfaces will be disproportionately impacted by anthropogenic salt loading. However, the relationship between impervious surface and EC is weak, and there is considerable within-site variance between plots at individual low- and medium-density and roadside sites. In fact, the lowest and second highest EC samples came from just one transect along a roadside within 45 m of one another (3.4 and 1067 μS/cm, respectively). This within-site fine-scale heterogeneity in EC does not appear in urban natural areas or less developed sites (grassland, forest and open space) with the exception of one open space site that is located adjacent to a community garden. The finding of high spatial heterogeneity in more developed land covers is consistent with other soil characteristics such as carbon content, soil available phosphorus and saturated hydraulic conductivity (Ziter and Turner 2018) and points to a consistent trend in urban soil characteristics. We hypothesize that within-site heterogeneity is driven by anthropogenic alterations to both salt inputs and soil characteristics. Examples of such fine-scale alterations include spot application of fertilizer or road
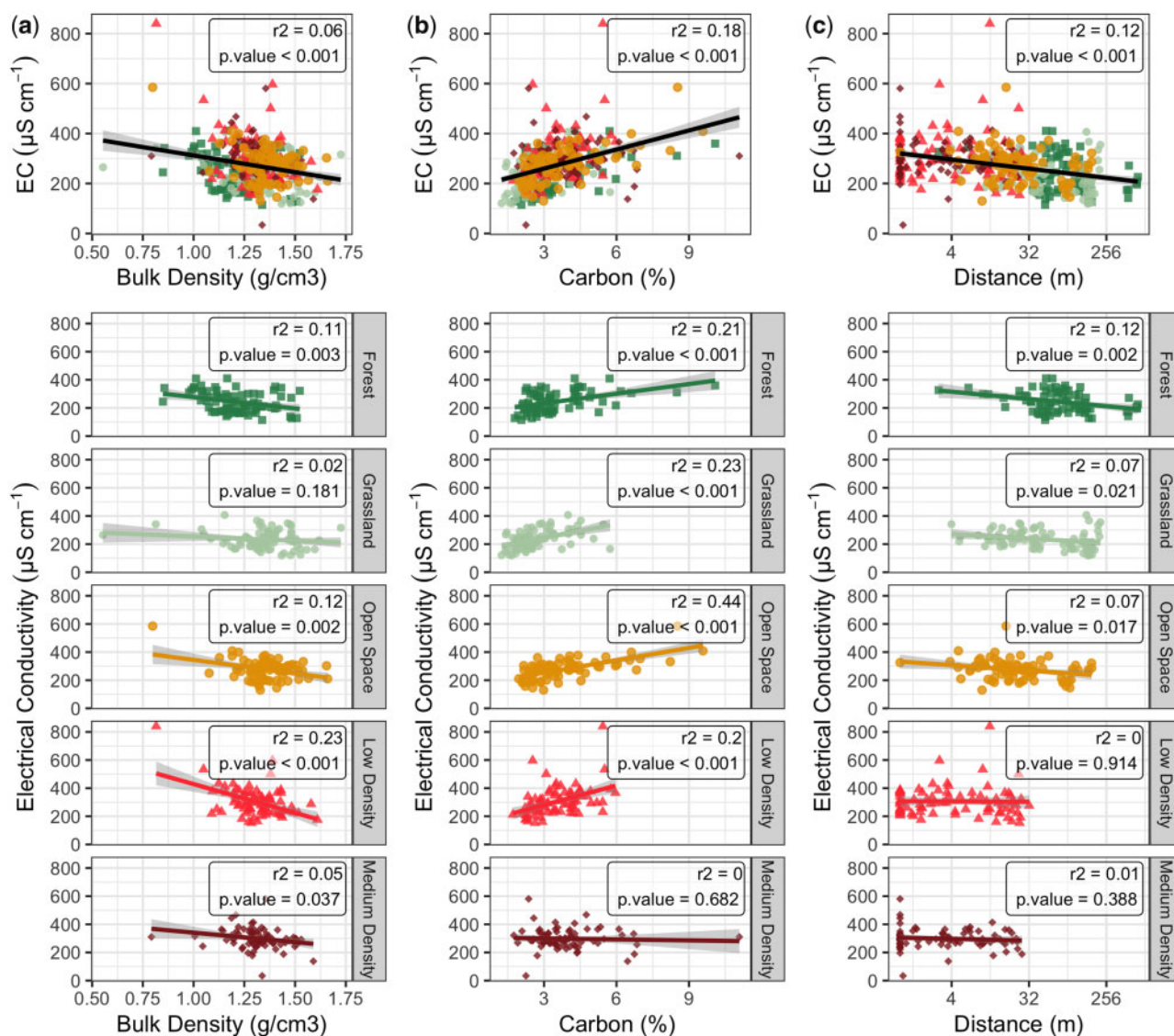
Figure 4: Soil EC vs. (a) bulk density, (b) percent carbon and (c) distance to nearest impervious surface. Linear regression r^2 and P-values given for all sampling points and by land-cover class

salt, imported soils and construction fill and engineered water retention/drainage features. It is important to note that in urban landscapes, soils are often imported as 'fill' in roadside or other construction projects, and it may be impossible to tell whether soils are native or imported. Imported soils may have more variable pore size, ion sorption capacity and natural ionic content relative to the native natural soils, exacerbating the patchiness of soil EC in heterogeneous urban environments (Pavao-Zuckerman 2008; Zhou, Pickett, and Cadenasso 2017) and variably moderating soil response to road salt intrusion. Overall, the heterogeneity in soil EC may explain the variation in detrimental effects on terrestrial environments, including salt damage to vegetation along roadsides or on private property.

Not all roadside sites had high variance. The site with the highest mean EC (range, 606–811 μS/cm) was located along a road near the University of Wisconsin-Madison snow storage lot. The transect was in a shallow grassy ditch along a road with no curbs. It may be that the low elevation of the site in relation to the road makes the site susceptible to road runoff. Additionally, the nearby snow storage pile may contribute to

salt loading. Conversely, the road site with the lowest mean EC (range, 107–163 μS/cm) is located on the inside curve of the confluence of two-major streets with a four-way stop. The transect is adjacent to a city cemetery, and road curbs are present. This site might have low EC due to curbs preventing road runoff, and the four-way stop limiting traffic speed and potential road salt spray. These observations highlight the potential importance of local planning and design features in driving ecological outcomes in urban landscapes (Zhou, Pickett, and Cadenasso 2017).

Evaluating drivers of soil EC and assigning causality to anthropogenic salt loading is challenging due to variable soil characteristics. Soils with a high clay content and low porosity more readily sorb ions, but the low porosity will limit the pore-water content. In contrast, soils with high porosity and permeability are less effective at sorbing dissolved ions but can retain higher volumes of saline pore water (Robinson, Hasenmueller, and Chambers 2017). These soils may retain incoming dissolved ions in the short term but will flush salt more readily with subsequent wetting. Soil EC may also be impacted by the presence
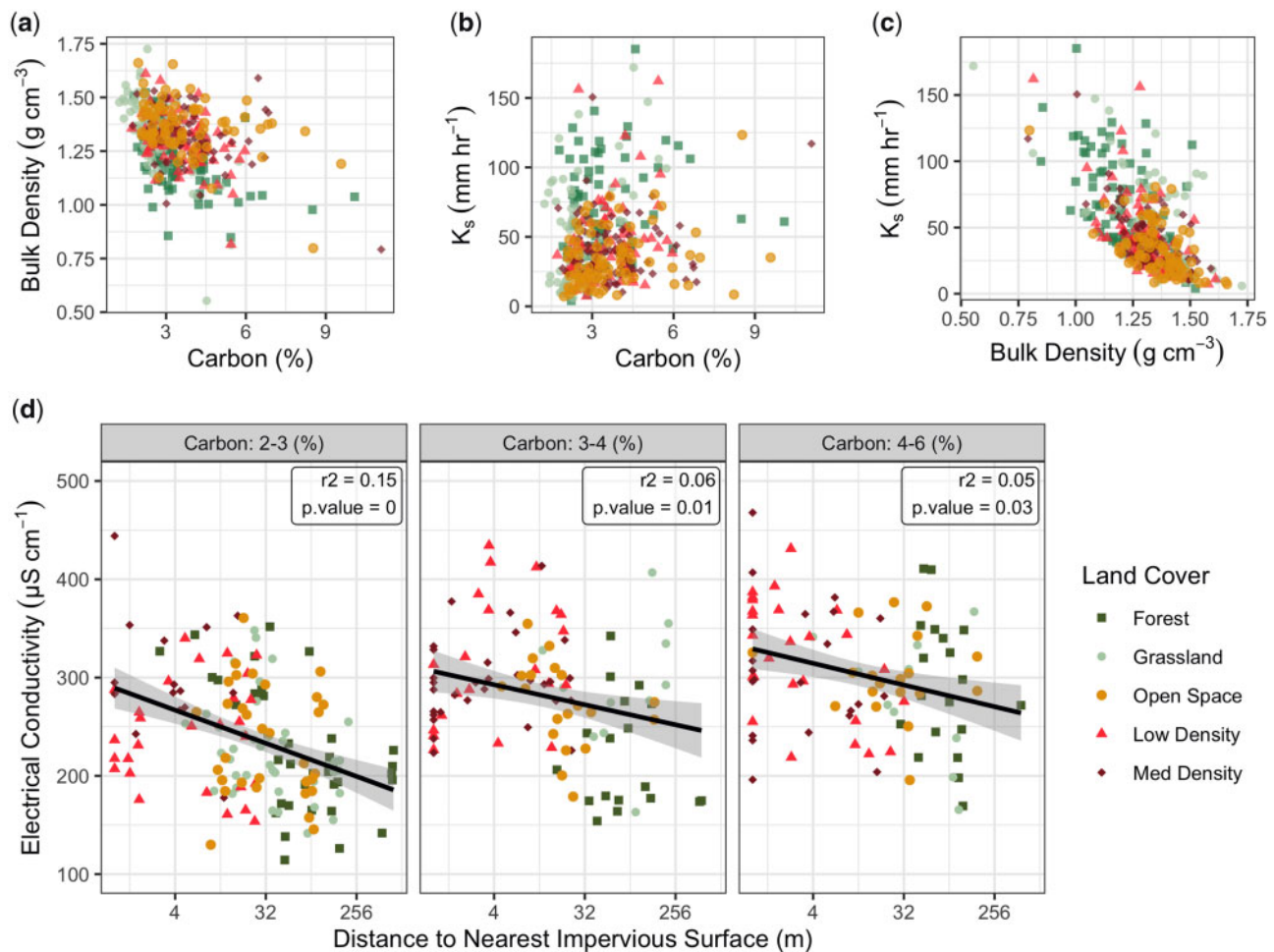
Figure 5: (a) Soil percent carbon vs. bulk density and (b) saturated hydraulic conductivity (K_s). (c) Soil bulk density vs. K_s. (d) Soil EC vs. distance to the nearest impervious surface at three groupings of percent carbon (2–3%, $n = 155$; 3–4%, $n = 103$; 4–6%, $n = 98$). Y-Axis limits were constrained to aid in visualization. Not shown are five data points with EC above 500 μS/cm and one data point with EC below 100 μS/cm

of organic matter, both through the influence of organic matter on soil bulk density and moisture retention (Omonode and Vyn 2006), and the capacity of organic matter to hold cations (cation-exchange capacity) (Sudduth et al. 2005).

Our data reveal percent carbon, and therefore organic matter, has the strongest relationship to EC of all our studied parameters (Fig. 4). While this relationship has been observed in other soil studies (Omonode and Vyn 2006), the positive correlation between organic matter and EC is not universal (Sudduth et al. 2005). Since we were interested in the influence of impervious surface on EC, we grouped soils into subgroups based on percent carbon. Accounting for percent carbon, we found that weak, but significant, correlation between distance to impervious surface and EC remains (Fig. 5d). We also hypothesize that porosity may play a larger role in controlling EC than permeability, although the two are highly correlated. We found that the high variance in K_s at forest and grassland sites did not manifest in high variance in EC. In contrast, the highest EC was often at sites with low bulk density. This suggests that high porosity promotes salt retainment, with permeability playing a secondary role. As our sites were sampled during the summer and were exposed to months of spring and summer precipitation, EC values may represent salts stored within the immobile water phase, where solutes move via diffusion rather than advection,

or may represent salts sorbed to the soils. Additionally, if soils have very poor drainage, salts could accumulate from evapo-concentration. Reduced infiltration has been found to be common in roadside soils as heavy traffic can compact soils, and a lack of vegetation will reduce permeability (Gregory et al. 2006; Zhou, Lin, and White 2008; Woltemade 2010).

The physical characteristics of urban soils and the alteration of natural hydrological processes in urban settings (Voter and Loheide 2018) play a pivotal role in delaying salt inputs to deeper groundwater. With higher mean EC, developed land covers retain more salts than more natural land-cover classes, and with 70+ years of road salt application, many urban soils are likely saturated in salts. If anthropogenic salt inputs cease, it is unclear to what magnitude this legacy salt will eventually leach into surface and groundwaters. Currently, low- and medium-density developed land covers represent 30 and 17% of the Madison landscape, respectively (Ziter and Turner 2018), and these percentages will likely grow as Madison and Dane County's populations continue to increase. With increasing development, anthropogenic salt inputs threaten to salinize urban soils and contribute to the storage of legacy salt on the landscape.

The deleterious effects of road salt use on urban ecosystems in the USA over the past 70 years are becoming well

documented, especially for urban waters (Corsi et al. 2015; Dugan et al. 2017; Stets et al. 2018). Here, we show a weak relationship between soil EC and impervious surface, with higher soil EC, and increased heterogeneity, on developed land. An important next step is to directly monitor soil EC over the course of a year, from winter salt application through spring melt, to understand seasonal variability in EC and its relationship to hydrology. These observations will inform our understanding of salt transport through soils to shallow groundwater. Additionally, we encourage future researchers in this area to examine the impacts of urban design and planning (e.g. curbs, traffic speed and storm drains) on salt accumulation and transport, to monitor the efficacy of improved salt management approaches such as brine application and to better link urban ecology research with the social actions and environmental perceptions that drive salt application and management in urban landscapes.

Acknowledgements

Funding

REFERENCES

Bäckström, M. et al. (2004) 'Mobilisation of Heavy Metals by Deicing Salts in a Roadside Environment', *Water Research*, **38**: 720–32.

Bastviken, D. et al. (2007) 'Chloride Retention in Forest Soil by Microbial Uptake and by Natural Chlorination of Organic Matter', *Geochimica et Cosmochimica Acta*, **71**: 3182–92.

Bates, D. et al. (2015) 'Fitting Linear Mixed-Effects Models Using {lme4}', *Journal of Statistical Software*, **67**: 48.

Bockheim, J. G., and Hartemink, A. E. (2017) *Current and Future Soil Research*, pp. 223–228. Cham: Springer.

Bryson, G. M., and Barker, A. V. (2002) 'Sodium Accumulation in Soils and Plants along Massachusetts Roadsides', *Communications in Soil Science and Plant Analysis*, **33**: 67–78.

Corsi, S. R. et al. (2015) 'River Chloride Trends in Snow-Affected Urban Watersheds: Increasing Concentrations Outpace Urban Growth Rate and Are Common Among All Seasons', *Science of the Total Environment*, **508**: 488–97.

Cunningham, M. A. et al. (2008) 'Accumulation of Deicing Salts in Soils in an Urban Environment', *Urban Ecosystems*, **11**: 17–31.

Czerniawska-Kusza, I., Kusza, G., and Duzyński, M. (2004) 'Effect of Deicing Salts on Urban Soils and Health Status of Roadside Trees in the Opole Region', *Environmental Toxicology*, **19**: 296–301.

Dugan, H. A. et al. (2017) 'Salting Our Freshwater Lakes', *Proceedings of the National Academy of Sciences of the United States of America*, **114**: 4453–8.

Equiza, M. A. et al. (2017) 'Long-Term Impact of Road Salt (NaCl) on Soil and Urban Trees in Edmonton, Canada', *Urban Forestry & Urban Greening*, **21**: 16–28.

Fischel, M. (2001) Evaluation of Selected Deicers Based On a Review of the Literature. Page Publication No. CDOT-DTD-R-2001-15. A Final Report prepared for the Colorado Department of Transportation. Denver, CO.

Frenkel, H., Goertzen, J. O., and Rhoades, J. D. (1978) 'Effects of Clay Type and Content, Exchangeable Sodium Percentage, and Electrolyte Concentration on Clay Dispersion and Soil Hydraulic Conductivity', *Soil Science Society of America Journal*, **42**: 32–9.

Green, S. M., Machin, R., and Cresser, M. S. (2008) 'Effect of Long-Term Changes in Soil Chemistry Induced by Road Salt Applications on N-Transformations in Roadside Soils', *Environmental Pollution*, **152**: 20–31.

Gregory, J. H. et al. (2006) 'Effect of Urban Soil Compaction on Infiltration Rate', *Journal of Soil and Water Conservation*, **61**: 117–24.

Haq, S., Kaushal, S. S., and Duan, S. (2018) 'Episodic Salinization and Freshwater Salinization Syndrome Mobilize Base Cations, Carbon, and Nutrients to Streams Across Urban Regions', *Biogeochemistry*, **141**: 463–86.

Kaushal, S. S. et al. (2005) 'Increased Salinization of Fresh Water in the Northeastern United States', *Proceedings of the National Academy of Sciences of Sciences of the United States of America*, **102**: 13517–20.

Kelly, V. R. et al. (2008) 'Long-Term Sodium Chloride Retention in a Rural Watershed: Legacy Effects of Road Salt on Streamwater Concentration', *Environmental Science & Technology*, **42**: 410–5.

Kelly, W. R. et al. (2010) 'Using Chloride and Other Ions to Trace Sewage and Road Salt in the Illinois Waterway', *Applied Geochemistry*, **25**: 661–73.

Kincaid, D. W., and Findlay, S. E. G. (2009) 'Sources of Elevated Chloride in Local Streams: Groundwater and Soils as Potential Reservoirs', *Water, Air, and Soil Pollution*, **203**: 335–42.

Lenth, R. 2019. emmeans: Estimated Marginal Means, aka Least-Squares Means. R package version 1.4.4. https://CRAN.R-project.org/package=emmeans.

Lovett, G. M. et al. (2005) 'The Biogeochemistry of Chlorine at Hubbard Brook, New Hampshire, USA', *Biogeochemistry*, **72**: 191–232.

Mawer, C., Knight, R., and Kitanidis, P. K. (2015) 'Relating Relative Hydraulic and Electrical Conductivity in the Unsaturated Zone', *Water Resources Research*, **51**: 599–618.

Norrström, A. C., and Jacks, G. (1998) 'Concentration and Fractionation of Heavy Metals in Roadside Soils Receiving De-Icing Salts', *Science of the Total Environment*, **218**: 161–74.

Novotny, E. et al. 2008. A Salt (Chloride) Balance for the Minneapolis/St. Paul Metropolitan Area Environment. St. Anthony Falls Project Report: 29.

Omonode, R. A., and Vyn, T. J. (2006) 'Spatial Dependence and Relationships of Electrical Conductivity to Soil Organic Matter, Phosphorus, and Potassium', *Soil Science*, **171**: 223–38.

Oswald, C. J. et al. (2019) 'Spatial Distribution and Extent of Urban Land Cover Control Watershed-Scale Chloride Retention', *Science of the Total Environment*, **652**: 278–88.

Paludan-Muller, G. et al. (2002) 'Differences in Salt Sensitivity of Four Deciduous Tree Species to Soil or Airborne Salt', *Physiologia Plantarum*, **114**: 223–30.

Pavao-Zuckerman, M. A. (2008) 'The Nature of Urban Soils and Their Role in Ecological Restoration in Cities', *Restoration Ecology*, **16**: 642–9.

Pebesma, E. (2018) 'Simple Features for R: Standardized Support for Spatial Vector Data', *The R Journal*, **10**: 439–46.

Qadir, M., and Schubert, S. (2002) 'Degradation Processes and Nutrient Constraints in Sodic Soils', *Land Degradation & Development*, **13**: 275–94.

R Core Team. (2018) *R: A Language and Environment for Statistical Computing*. Vienna, Austria: R Foundation for Statistical Computing.

Robinson, H. K., Hasenmueller, E. A., and Chambers, L. G. (2017) 'Soil as a Reservoir for Road Salt Retention Leading to Its Gradual Release to Groundwater', *Applied Geochemistry* , **83**: 72–85.

Schuler, M. S., and Relyea, R. A. (2018) 'A Review of the Combined Threats of Road Salts and Heavy Metals to Freshwater Systems', *BioScience*, **68**: 327–35.

Shi, X. et al. (2013) 'Use of Chloride-Based Ice Control Products for Sustainable Winter Maintenance: A Balanced Perspective', *Cold Regions Science and Technology*, **86**: 104–12.

Snodgrass, J. W. et al. (2017) 'Influence of Modern Stormwater Management Practices on Transport of Road Salt to Surface Waters', *Environmental Science & Technology*, **51**: 4165–72.

Stets, E. et al. (2018) 'Increasing Chloride in Rivers of the Conterminous U.S. and Linkages to Potential Corrosivity and Lead Action Level Exceedances in Drinking Water', *Science of the Total Environment*, **613–614**: 1498–509.

Strifling, D. A. (2018) 'Reducing Chloride Discharges to Surface Water and Groundwater: A Menu of Options for Policymakers', *Environmental Law*, **48**: 167–210.

Sudduth, K. A. et al. (2005) 'Relating Apparent Electrical Conductivity to Soil Properties across the North-Central USA', *Computers and Electronics in Agriculture*, **46**: 263–83.

Svensson, T. et al. (2007) 'Chlorine Transport in a Small Catchment in Southeast Sweden during Two Years', *Biogeochemistry*, **82**: 181–99.

Voter, C. B., and Loheide, S. P. (2018) 'Urban Residential Surface and Subsurface Hydrology: Synergistic Effects of Low-Impact Features at the Parcel Scale', *Water Resources Research*, **54**: 8216–33.

Woltemade, C. J. (2010) 'Impact of Residential Soil Disturbance on Infiltration Rate and Stormwater Runoff', *JAWRA Journal of the American Water Resources Association*, **46**: 700–11.

Zhou, W., Pickett, S. T. A., and Cadenasso, M. L. (2017) 'Shifting Concepts of Urban Spatial Heterogeneity and Their Implications for Sustainability', *Landscape Ecology*, **32**: 15–30.

Zhou, X., Lin, H. S., and White, E. A. (2008) 'Surface Soil Hydraulic Properties in Four Soil Series Under Different Land Uses and Their Temporal Changes', *Catena*, **73**: 180–8.

Ziter, C., and Turner, M. (2018) 'Current and Historical Land Use Influence Soil-Based Ecosystem Services in an Urban Landscape', *Ecological Applications*, **28**: 643–54.