Quantifying road salt impacts on forested wetland structure and function in eastern Connecticut

Basic Information

Title:	Quantifying road salt impacts on forested wetland structure and function in eastern Connecticut
Project Number:	2018CT314B
Start date:	3/1/2018
End date:	2/28/2019
Funding source:	104B
Congressional District:	2nd
Research Category:	Water Quality, Biological Sciences
Focus Category:	Wetlands, Groundwater, Water Quality
Descriptors:	Forested wetland, groundwater, road salt, vegetation
Principal Investigators:	Beth Lawrence, Gary Robbins

Publications

Walker, S., Robbins, G., Helton, A. Lawrence, B. Quantifying road salt impacts on forested wetland structure and function. *In prep.*

Quantifying road salt impacts on forested wetland structure and function in eastern Connecticut

FINAL REPORT

to

Connecticut Institute of Water Resources

United States Geological Survey

FY 2018-19 104 B Program

May 23, 2019

Beth Lawrence, UConn Dept. of Natural Resources and the Environment, Center for Environmental Science and Engineering

Gary Robbins, UConn Dept. of Natural Resources and the Environment, Dept. of GeoSciences

Introduction/Research Objectives

Salinization of fresh water ecosystems in the temperate north is a growing environmental and human health problem. A growing body of literature suggests that runoff from road salt has long-term implications for the chemical and biological dynamics of groundwater (Cassanelli and Robbins 2013; Mullaney et al. 2009; Vitale et al. 2017), streams (Kaushal et al. 2005), lakes (Dugan et al. 2017a), and wetlands (Dugan et al. 2017b; Richburg et al. 2001; Rhodes and Guswa 2016). Compounded with increased urban expansion and impervious surface cover, the quality of our surface and groundwater resources and the ecological communities and functions they support are increasingly at risk due to elevated salinity (Trombulak and Frissell 2000).

The 2017 Connecticut State Water Plan identified "areas affected by road salt" as needing data to address state water priorities. Specifically, impacts on streamside vegetation and aquatic habitats were noted as areas of interest (CT State Water Plan 2017). Connecticut provides an ideal study system to investigate road salt-wetland issues due to the abundance of wetlands, a dense road network, and the prolific use of road deicing salts in the region. Forested wetlands typically lie along rivers and streams with perennial flowthrough, or in undrained depressions (Tiner et al. 2013) and comprise two thirds of Connecticut's freshwater wetlands; these ecosystems have been identified as a high risk ecosystem due to environmental alterations associated with climate change and road fragmentation (CT DEEP 2015; Tiner et al. 2013). Our cumulative understanding of road salt impacts suggests that it is an emerging driver of environmental change in north temperate regions, though little is known about the impacts of road deicing salts on forested wetland structure and function.

The overarching objective of our proposed work was to examine the impact of road salt pollution on the structure (i.e., vegetation composition and abundance) and function (i.e., water quality, carbon cycling) of eastern Connecticut forested wetlands. To achieve this aim, we coupled field observations of wetland hydrology with controlled greenhouse experiments, and laboratory assays to address the following objectives:

1) *Characterize forested wetland hydrology and water quality in eastern Connecticut.*
    a. Quantify the temporal variability of salt stress
    b. Characterize the relative influence of ground vs. surface water and its quality in red maple forested wetlands communities
2) *Test how variable hydroperiods and salinity levels alter forested wetland seedling emergence*
3) *Examine how elevated salinity in roadside RM swamps alters the stability of stored soil carbon*

Materials/Procedures/Progress

Obj 1: In order to investigate the relationship between seasons and water salinity levels, we installed surface- and ground- water wells at the near-road wetland edge of a red maple forested wetland site in Coventry, CT. The groundwater well was installed 91.44 cm into the ground and the surface well was installed 30.48 cm into the ground. In February 2018, an In-Situ® Aqua TROLL® 200 (In-Situ, Fort Collins, CO) was put in each of the wells to monitor hourly water depth, EC, and salinity (ppt) of the surface- and ground- water. The surface water well probe was removed for two months (July to September) to support another project. Additionally, an In-

Situ® Rugged TROLL® 100 was installed nearby to measure barometric pressure for water depth calculations. Data were downloaded using Win-Situ® 5 (version 5.6.32.1) software (Win-Situ® 5, 2018).

To characterize spatial and temporal patterns of surface- and ground- water influence and salinity, EC and pH levels, we installed three pairs of surface- and ground- water wells each in near- (i.e., wetland edge) and far-road (~90 meters from the wetland edge) environments in May 2018 at three red maple forested wetland sites (Fig. 1) (3 well pairs x 2 zones x 3 sites = 18 well pairs). Depth to water data from each well was collected every two weeks from May to November. Water samples were collected from each well twice per season and then only after snowstorm thaw events during winter 2018-2019. Water samples were centrifuged then filtered using a 110-mm diameter Whatman paper filter, and we used an Orion Conductivity Cell and an Orion Star A215 pH Conductivity Meter with Ross Ultra pH/ATC Triode at room temperature (25°C) to quantify EC and pH. We used a hydraulic gradient threshold of ± 0.02 to determine if there was horizontal flow; anything higher was considered vertical flow. Daily precipitation data was downloaded from the National Oceanic and Atmospheric Administration's Hartford Bradley International Airport, CT US station (NOAA, 2019).

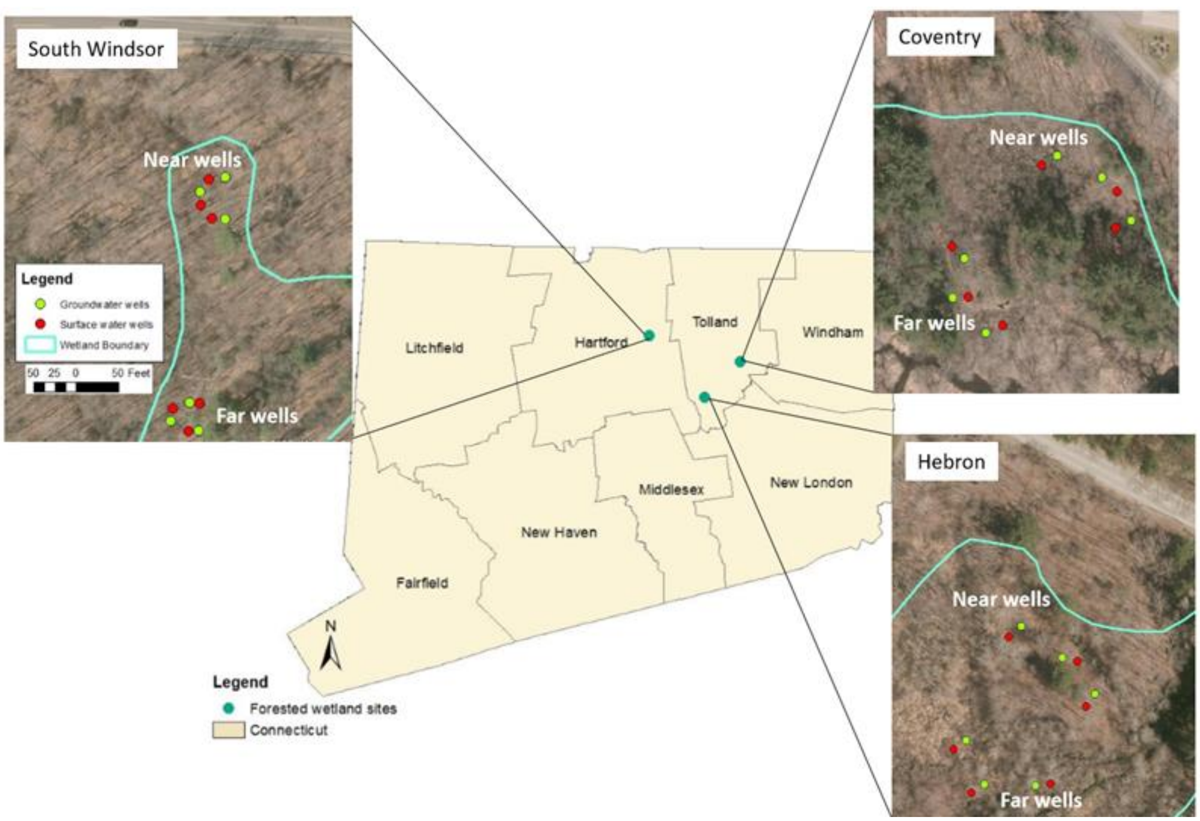

Figure 1. Pairs of ground and surface water wells were installed in three red maple forested wetlands in eastern CT in spring 2018. At each site, three pairs were placed in the near-road environment (wetland edge) and three were placed in far-road locations (≥90m).

Obj 2: To examine a range of experimental NaCl salinities on seed bank responses, we initiated a 4-month greenhouse seed bank experiment by collecting surface soils (10-cm depth) during February 2018 from the less road-affected portion (~90 m from road; relatively low soil Na^+) at a red maple-dominated forested wetland in Coventry, CT. We sieved soils through a 5-mm sieve to remove rhizomes and rocks, and then filled 1000 mL pots (11.5-cm diameter, 14.1-cm tall) with 9 cm (725 mL) of autoclaved-sterilized play sand and spread a 2-cm layer of sieved soil on top. Half of the pots had 0.95-cm holes drilled in them for periodic drainage for the salinity pulsing treatment; we used rubber stoppers to plug them and maintain water levels. We prepared a 50 ppt NaCl stock solution and diluted it as necessary to apply designated salinity treatments.

We implemented a full factorial seed bank experiment, manipulating three factors: salinity, water level, and timing. To explore seed bank response salinity thresholds, we tested six salinity treatments (0, 0.5, 1, 2, 4 and 8 ppt) that ranged from fresh to a level we expected would be lethal, as studies documented that many freshwater aquatic macrophytes cannot survive above 4 ppt of salinity (Hart et al., 1991; Nielsen et al., 2003). Red maple forested wetlands typically have undulating microtopography with distinct saturated hollows and raised, drier hummocks; therefore we tested two water levels ("surface": 0 cm, at soil surface; "low": ~ 4 cm below soil surface, moist sand) to investigate interactions between salinity and water level. Road salt inputs generally enter forested wetlands in pulses during freeze thaw events so we examined how timing of salinity exposure may impact seed bank responses differently by comparing constant (exposed to a constant level of salinity) and pulsing (pulsed with salinity for 2 weeks, drained, pulsed with freshwater for 2 weeks, drained, etc.) treatments. We replicated each treatment combination four-fold for a total of 96 experimental units (6 salinity treatments x 2 water levels x 2 timing treatments x 4 replicates).

At the beginning of March 2018, we implemented our seed bank experiment in a natural light, unheated greenhouse where we monitored hourly greenhouse temperature using I-buttons (Maxim Integrated, San Jose, CA), and water levels at least every other day. We quantified seed bank responses every two weeks (number of seedlings, species ID); seedlings were grouped based on distinct morphology until species could be identified. Several morphotypes were grown beyond the end of the experiment until floral development allowed us to identify them to species level. We collected soil EC and moisture data once a month using a WET-2 Sensor (Delta-T Devices, Cambridge, UK); data were averaged across the four months for data analysis. Pots were randomly placed on greenhouse benches and re-randomized each month. After four months (end of June 2018), pore water samples were taken using a Henry Sampler, seedling biomass was cut at the soil surface, dried at 65°C for 72 hours, and weighed to quantify aboveground biomass. Porewater pH was measured using an Orion Conductivity Cell and an Orion Star A215 pH Conductivity Meter with Ross Ultra pH/ATC Triode at room temperature (25°C). Over the course of the experiment, greenhouse average daily maximum temperature was 30.3 ± 0.85°C and minimum was 16.2 ± 0.23°C.

Obj 3: To assess the impact of elevated salinity in roadside forested wetlands on carbon lability, in August 2018 we collected three surface sediment cores (10-cm depth) in salt-affected (<10m) and low impact zones (>90m) at three red maple dominated sites (Fig 1). Sediments were sieved with 2-mm mesh and homogenized by zone. Using established Lawrence laboratory assay protocols (Johnson 2018), we quantified 24-hour carbon mineralization rates on field moist (50g) and soil slurries (50g soil + 50mL treatment solution) with different levels of salinity (3 NaCl salinities: 0ppt, 0.25ppt, 2.5 ppt) using a Picarro G2201i and 16-port manifolds (Fig. 2) which quantifies CO_2 and CH_4 concentrations real time. Three-fold replication resulted in a total of 72 soil incubation (field moist: 3 sites x 2 zones x 3 replicates = 18; slurries: 3 sites x 2 zones x 3 salinities x 3 replicates = 54).

Figure 2. A Picarro gas analyzer and 16-port manifolds were used to quantify CO_2 accumulation rates from forested wetland soil subjected to salinity treatments in incubation jars over a 24-hour period (carbon mineralization).

Results/Significance

Obj 1: For our two wells that were monitored hourly over a 14 month period (Fig 3), salinity was relatively low and ranged from 0.013 to 0.388 ppt, EC ranged from 31.289 to 795.41 uS/cm, and depth to water ranged from -0.485 (negative due to flooding above the surface water well) to 2.325 feet. Salinity and EC did not differ ($p > 0.05$) between our surface- and ground- water well, however our surface water well had a larger range of values with a coefficient of variation (CV) of 40.3% compared to the ground water well with a CV of 28.7%.

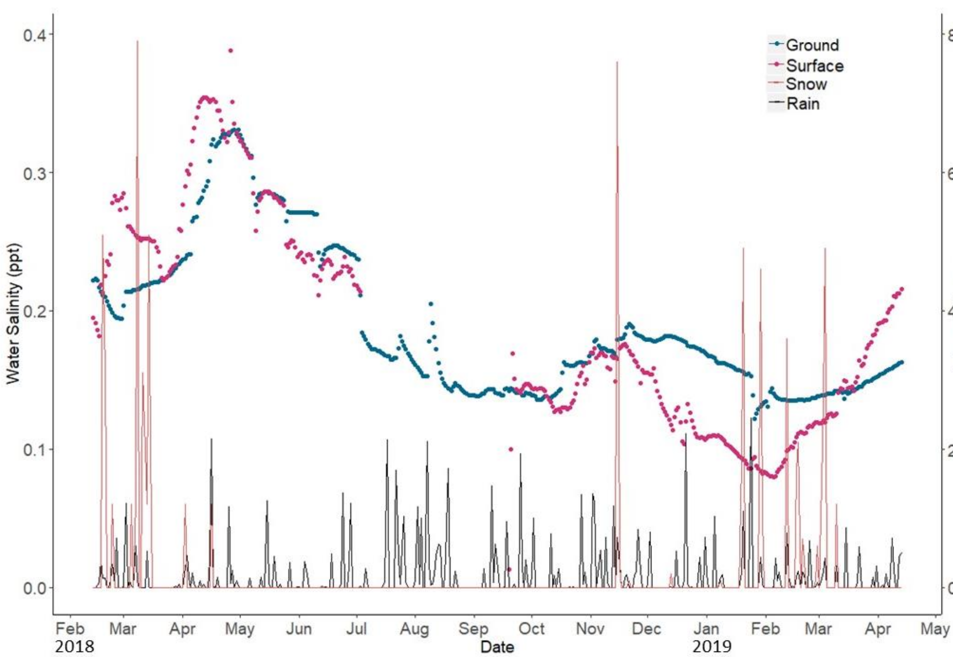

Figure 3. Water salinity (ppt) of surface- and ground- water over a 14-month period from a roadside red maple-dominated wetland in Coventry, CT. Hourly salinity concentrations were averaged by day. Precipitation- total daily snow (and ice) or rain from nearest weather station are shown for reference.

Water samples collected from the eighteen well pairs over the course of 2018 and 2019 suggest that conductivity was elevated near the road at two (Hebron, Coventry) of the three sites (Fig 4).

We did not detect spatial patterns in vertical flow when comparing near- and far- road environments, but variations in groundwater recharge/discharge were apparent with time of year (Fig. 5). Wells near roads had higher EC (near: 459.1 ± 22.3 µs/cm, far: 348.0 ± 16.6 µs/cm) ($F_{1,170} = 17.27$, $p < 0.001$) and lower pH (near: 6.1 ± 0.04, far: 6.2 ± 0.04) ($F_{1,170} = 3.91$, $p < 0.05$) when compared to far wells (Fig 4). Water pH was also affected by well type ($F_{1,170} = 4.79$, $p < 0.05$), with ground water wells having higher pH than surface water wells (groundwater: 6.3 ± 0.04, surface water: 6.1 ± 0.04).

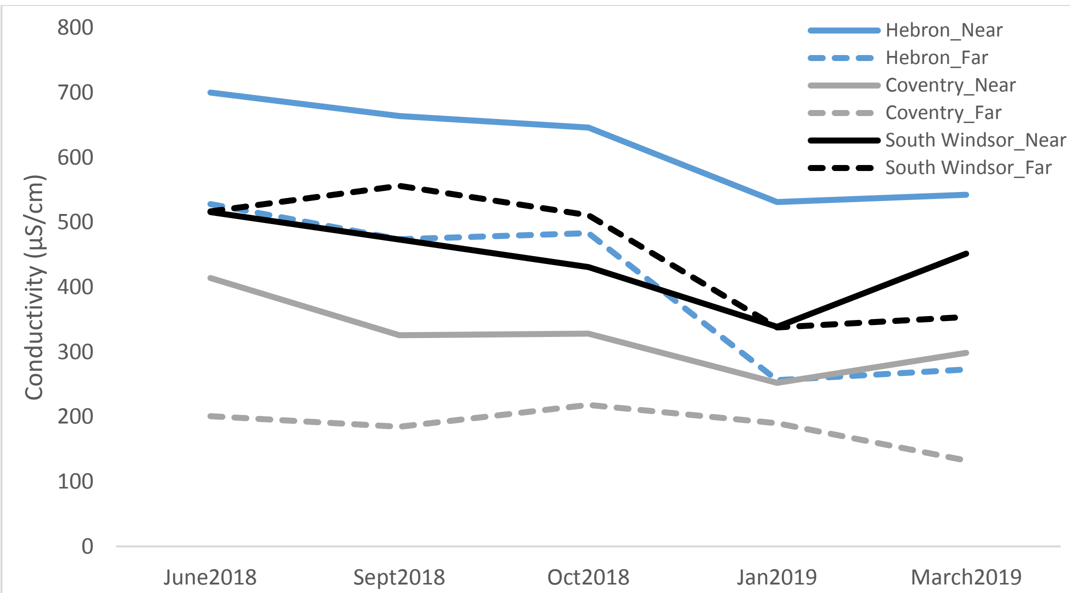

Figure 4. Water samples were collected from near and far-road environments from three sites in eastern CT periodically during 2018-2019; conductivity estimates of three surface and ground water well samples were averaged

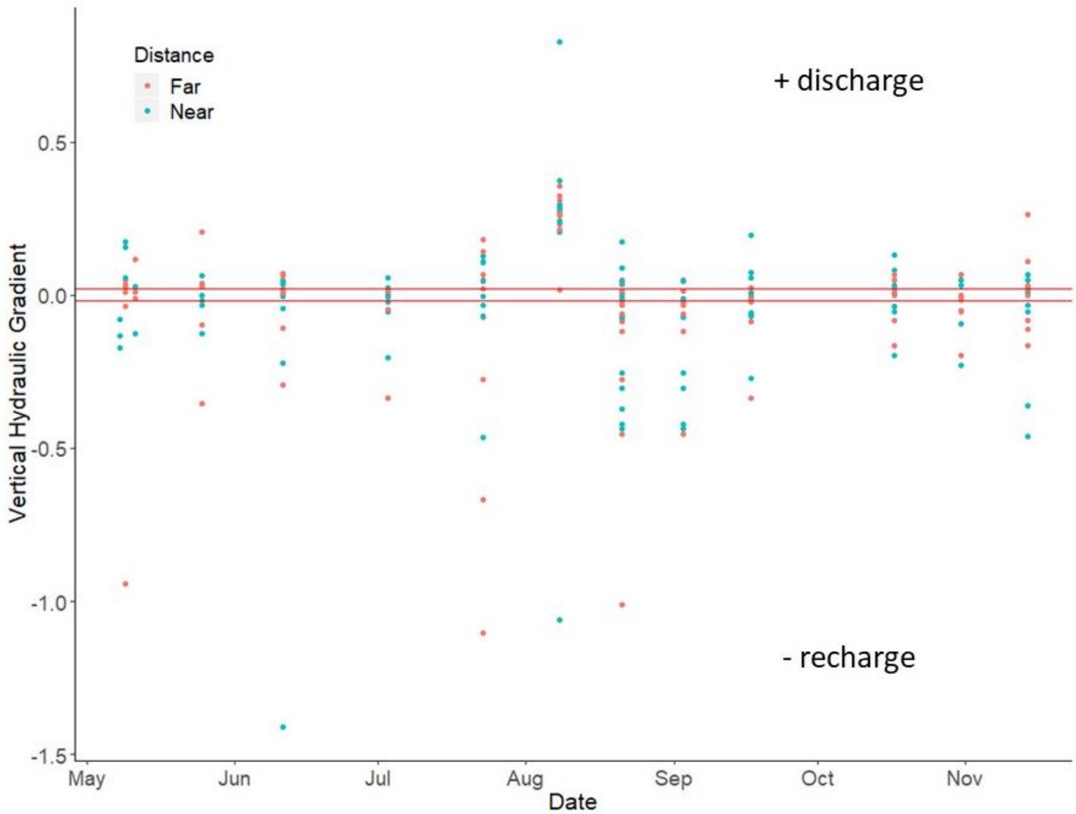

Figure 5. Vertical flow trends of our 18 wetland well pairs (3 sites x 6 well pairs/site) across 6-month period from May 2018 to November 2018. Well pairs are coded based on near- or far-road environments, but no distinct distance patterns are seen for vertical flow; however seasonal patterns are seen.

Obj 2: Our salinity treatments significantly affected several seedling responses, including species richness, maximum seedling density, and aboveground biomass (Table 1). Species richness (≥ 2 ppt: 2.5 ± 0.3, < 2 ppt: 6.7 ± 0.2) and aboveground biomass (≥ 2ppt: 0.1 ± 0.01 g, < 2 ppt: 0.3 ± 0.01 g) were reduced with salinity treatments ≥ 2 ppt compared to 0 ppt freshwater controls, whereas maximum seedling density was reduced at ≥ 4 ppt (≥ 4 ppt: 9.9 ± 1.2, < 4 ppt: 28.3 ± 1.3) (Fig. 6). Timing treatments also affected species richness and maximum seedling density (Table 1). Relative to constant salinity levels, pulsing salinity treatments increased species richness (pulsing: 4.9 ± 0.4, constant: 4.3 ± 0.4), maximum seedling density (pulsing: 23.6 ± 1.9, constant: 20.7 ± 1.7), and aboveground biomass (pulsing: 0.18 ± 0.02 g, constant: 0.16 ± 0.02 g). Interestingly, water level affected maximum seedling density, but no other seedling responses (Table 1); surface water levels elevated maximum seedling densities (surface: 24.3 ± 1.7, low: 19.9 ± 1.9) relative to low water. Timing treatments had distinct effects on species richness and aboveground biomass with increasing salinity due to an interactive effect (Table 1). Similarly, timing treatments had distinct effects on maximum seedling density and aboveground biomass with a change in water level due to an interactive effect (Table 1). Salinity and water level had an interactive effect only on aboveground biomass (Table 1).

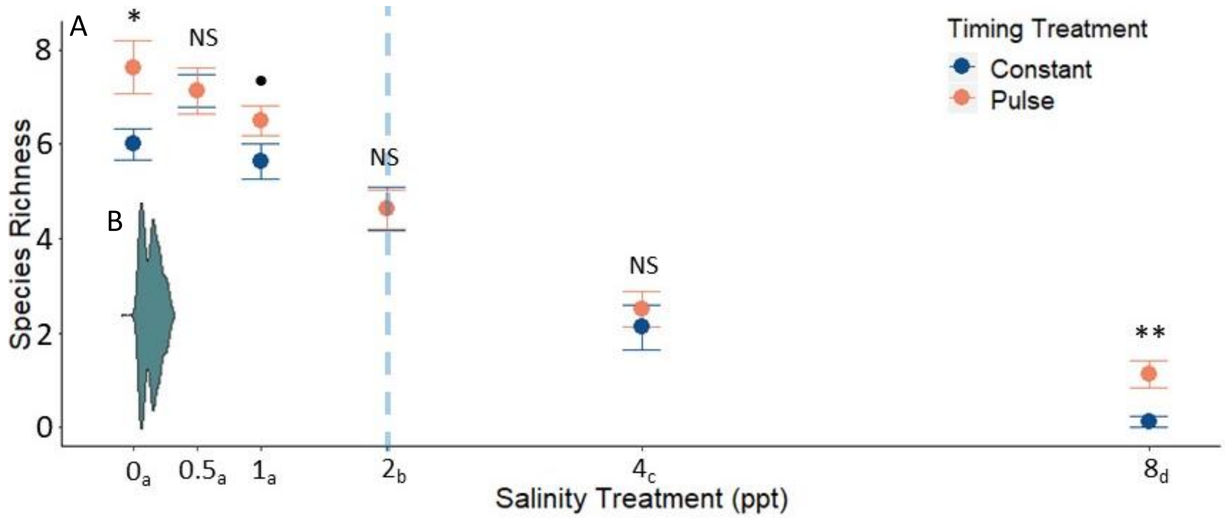

Figure 6. (A) **S**pecies richness (± 1 SE) at end of seed bank experiment for salinity and timing treatments. Symbols above the points show comparisons between timing treatments at each salinity level whereas letters on the x-axis show overall species richness differences compared to the control (0 ppt). NS $p > 0.10$, • $p < 0.10$, * $p < 0.05$, ** $p < 0.01$). (B) A violin plot (blue-green) overlay of the frequency and distribution of field salinity measurements over the course of 14 months from ground and surface wells from one of our most salt-effected red maple wetlands.

As expected, salinity treatments altered soil EC ($F_{5,72} = 723.7$, $p < 0.01$), with all treatments having significantly different EC values. Timing ($F_{1,72} = 19.3$, $p < 0.01$) and water level ($F_{1,72} = 17.4$, $p < 0.01$) treatments also influenced soil EC. Surface water levels had greater soil EC than low water treatments (surface: 2909.9 ± 311.4 µs/cm, low: 2441.0 ± 221.0 µs/cm). Soil moisture was altered by salinity ($F_{5,72} = 28.6$, $p < 0.01$), timing ($F_{1,72} = 14.7$, $p < 0.01$), and water level ($F_{1,72} = 121.2$, $p < 0.01$) treatments. Soil moisture increased with the concentration of salinity treatments, likely due to the apparent collapse of soil structure at higher concentrations (0 ppt: 46.2 ± 0.5 % vol, 8 ppt: 53.1 ± 1.0 % vol). Surface water and constant salinity treatments also had higher soil moisture than other treatments. Soil pH was only affected by the timing treatment ($F_{1,72} = 40.4$, $p < 0.01$), with higher soil pH with pulsing than constant treatments (pulsing: 7.9 ± 0.05, constant: 7.6 ± 0.04). Salinity and water level had an interactive effect on soil EC ($F_{5,72} = 11.2$, $p < 0.01$) and soil moisture ($F_{5,72} = 5.5$, $p < 0.01$), therefore the two water levels had distinct trends with increasing salinity for these seed bank responses. Timing treatments had different effects on soil EC ($F_{5,72} = 10.3$, $p < 0.05$), pH ($F_{5,72} = 4.3$, $p < 0.01$), and moisture ($F_{5,72} = 3.1$, $p < 0.05$) with increasing salinity due to an interactive effect. Water level and timing treatments had an interactive effect on soil pH ($F_{1,72} = 3.2$, $p < 0.1$) and soil moisture ($F_{1,72} = 13.4$, $p < 0.01$).

Table 1. 3-way ANOVA seedbank results. Compares significance levels of the effects of treatments and their interactions on plant responses (aboveground biomass, species richness, and maximum seedling density).

Effect	Biomass			Sp. Richness			Seedling Density		
	df	F	p	*df*	F	p	*df*	F	p
Salinity	5	45.3	***	5	114	***	5	24.3	***
Water Level	1	0.2	NS	1	2.2	NS	1	6.7	*
Timing	1	2.5	NS	1	13.4	***	1	2.9	•
Salinity x Water Level	5	2.1	•	5	1.7	NS	5	0.3	NS
Salinity x Timing	5	2.4	*	5	3.5	**	5	0.8	NS
Water Level x Timing	1	6.5	*	1	0.9	NS	1	5.7	*
Salinity x Water Level x Timing	5	1.3	NS	5	1.1	NS	5	1.0	NS

NS $p > 0.10$
• $p < 0.10$
* $p < 0.05$
** $p < 0.01$
*** $p < 0.001$

Obj 3: We observed inconsistent patterns in carbon mineralization between near and far-road zones across our three sties (Fig 7), as the interaction between zones and sites was significant. Possibly, the sites differed in underlying edaphic factors (pH, soil organic matter; Walker 2019) which may be associated with functionally distinct microbial communities. There is evidence from other studies that salt stress is a strong selective factor, resulting in salt-tolerant microbial communities in near road environments (Craig and Zhu 2018; Lancaster et al. 2016).

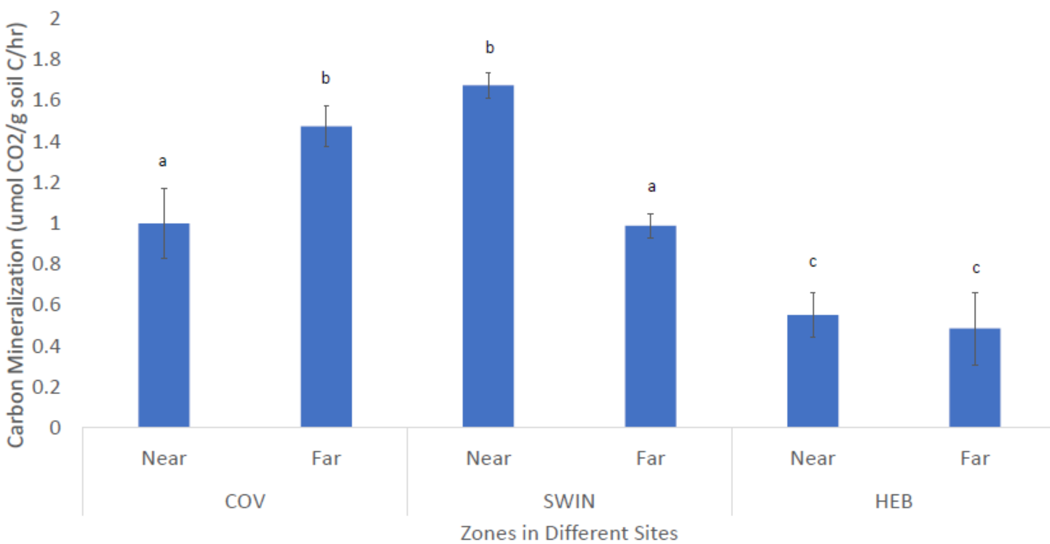

Figure 7. Carbon mineralization rates (mean ± 1 SE) of forested wetland soils were not consistent across sites and distance from road zones. Same letters indicate similar carbon mineralization rates.

Surprisingly, we did not detect differential responses to our salinity-slurry treatments, but the field moist treatment had higher carbon mineralization rates than the slurries (Fig 8). It is feasible that our treatments may not have been extreme enough (max salinity 2.5 ppt; short duration) to elicit a response.

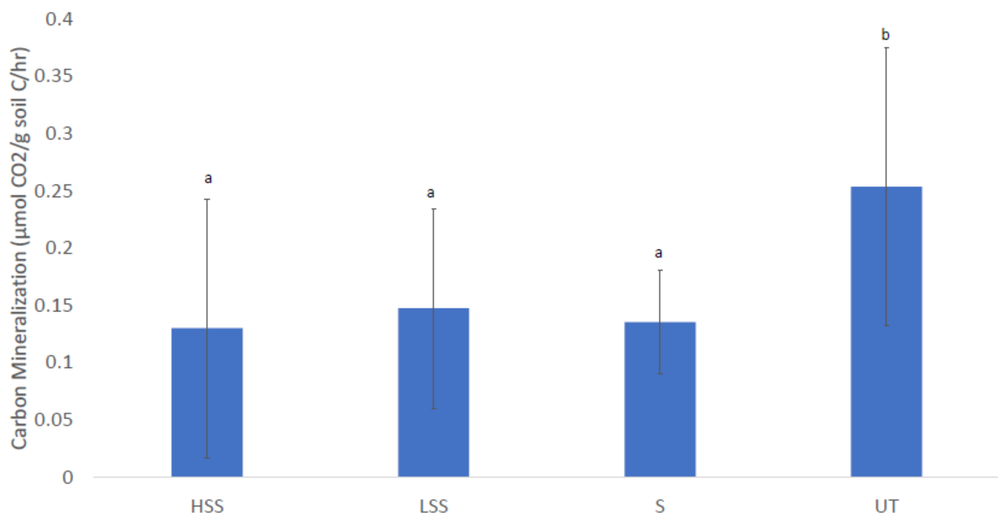

Figure 8. Forested wetland soil carbon mineralization rates differed between untreated field moist soils (UT) and soil slurries. Interestingly, NaCl treatment (HSS: high salt slurry, LSS: low salt slurry vs. S: slurry) did not affect short term carbon mineralization rates.

Conclusions

Our study suggests that even in exurban areas with low impervious surface cover, road deicing salt application affects water chemistry of forested wetlands months after active road salt application, highlighting road salt legacy effects. However, the conductivity measurements we made in the field during 2018 and 2019 were relatively low (max < 0.4 ppt), and well below the 2ppt threshold we identified during our seedbank experiment. Although chronic exposure to road salts could alter mineral nutrition of wetland vegetation through cation exchange, currently surface and ground water salinities do not appear salty enough to alter vegetation community composition (Walker 2019). Our vertical flow analyses indicate that forested wetlands in this region are often recharging groundwater (~36% of measurements during 2018); if chloride laden surface waters are recharging regional aquifers, this may exacerbate groundwater salinization. While constructed-roadside wetlands may mitigate road salt impacts throughout the watershed, relying only on naturally occurring wetlands may lead to landscape-level degradation of wetlands through salinization.

Literature Cited

Cassanelli, J. P., & Robbins, G. A. (2013). Effects of Road Salt on Connecticut's Groundwater: A Statewide Centennial Perspective. *Journal of Environment Quality*, 42(3), 737.

CT DEEP: Connecticut Department of Energy and Environmental Protection. 2015. Connecticut wildlife action plan. Available at: http://www.ct.gov/deep/cwp/view.asp?a=2723&q=329520&deepNav_GID=1719#Review

CT State Water Plan 2017. Final Draft Report Available at: http://www.ct.gov/water/lib/water/draft_state_water_plan_june_2017.pdf

Craig, S. and Zhu, W., 2018. Impacts of Deicing Salt and Nitrogen Addition on Soil Nitrogen and Carbon Cycling in a Roadside Ecosystem. *Water, Air, & Soil Pollution*, *229*(6), p.187

Dugan, H.A., Bartlett, S.L., Burke, S.M., Doubek, J.P., Krivak-Tetley, F.E., Skaff, N.K., Summers, J.C., Farrell, K.J., McCullough, I.M., Morales-Williams, A.M. and Roberts, D.C., 2017a. Salting our freshwater lakes. *Proceedings of the National Academy of Sciences*, *114*(17), pp.4453-4458.

Dugan, H.A., Helmueller, G. and Magnuson, J.J., 2017b. Ice formation and the risk of chloride toxicity in shallow wetlands and lakes. *Limnology and Oceanography Letters*.

Kaushal, S.S., Groffman, P.M., Likens, G.E., Belt, K.T., Stack, W.P., Kelly, V.R., Band, L.E. and Fisher, G.T., 2005. Increased salinization of fresh water in the northeastern United States. *Proceedings of the National Academy of Sciences of the United States of America*, *102*(38), pp.13517-13520.

Lancaster, N.A., Bushey, J.T., Tobias, C.R., Song, B. and Vadas, T.M., 2016. Impact of chloride on denitrification potential in roadside wetlands. *Environmental pollution*, *212*, pp.216-223.

Mullaney, J.F., Lorenz, D.L., and Arntson, A.D. 2009. Chloride in groundwater and surface water in areas underlain by the glacial aquifer system, northern United States, US Geological Survey, Scientific Investigations Report # 2009-5086, Reston, VA.

Rhodes, A.L. and Guswa, A.J., 2016. Storage and release of road-salt contamination from a calcareous lake-basin fen, western Massachusetts, USA. *Science of The Total Environment*, *545*, pp.525-545.

Richburg, J.A., Patterson III, W.A. and Lowenstein, F., 2001. Effects of road salt and *Phragmites australis* invasion on the vegetation of a western Massachusetts calcareous lake-basin fen. *Wetlands*, *21*(2), pp.247-255.

Tiner, R.W., Herman, J., and Roghair, L. 2013. Connecticut wetlands: characterization and landscape-level functional assessment. Prepared for the Connecticut Department of Environmental Protection, Hartford, CT. U.S. Fish and Wildlife Service, Northeast Region, Hadley, MA. 45 pp. plus appendices.

Trombulak, S.C., and Frissell, C.A. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology* 14 (1), pp. 18-30.

Vitale, S.A., G.A. Robbins, and L.A. McNaboe, L.A. 2017. Impacts of road salting on water quality in fractured crystalline bedrock. *Journal of Environmental Quality* 46: 288-294.

Walker, S. 2019 Effects of road salt on forested wetlands of southern New England. University of Connecticut, MS Thesis (defended 5/23/19; not yet submitted to Graduate School).