

Contrasting soil nitrogen dynamics across a montane meadow and urban lawn in a semi-arid watershed

Steven J. Hall^{1,2} · Michelle A. Baker³ · Scott B. Jones⁴ ·
John M. Stark³ · David R. Bowling^{1,5}

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Abstract Urbanization substantially increases nitrogen (N) inputs and hydrologic losses relative to wildland ecosystems, although the fate of N additions to lawns and remnant grasslands remains contested. In montane semi-arid ecosystems, N cycling is often closely coupled to snowmelt (the dominant period of infiltration) and snow cover, which impact soil temperature and moisture. Here, we compared soil N dynamics between a fertilized and irrigated urban lawn and nearby riparian meadow in Salt Lake City, Utah during a snow manipulation experiment. Snow removal increased freeze/thaw events but did not affect N pools, microbial biomass, denitrification potential, or soil oxygen (O₂). Mineral N was similar between sites despite lawn fertilization, but dissolved organic N (DON) was four-fold greater ($2.1 \pm 0.1 \text{ mg N l}^{-1}$) in lawn soil water. Infiltration was lower in the lawn subsoil, and leaching losses (modeled with Hydrus) were small at both sites ($< 2 \text{ kg N ha}^{-1} \text{ y}^{-1}$) despite substantial lawn fertilization. Lawn soil O₂ fluctuated between 20.9 and 1.6 % following snowmelt and irrigation, but remained near 20 % in the meadow; the lawn had more reducing microsites as indicated by iron speciation. Post-snowmelt potential denitrification was six-fold greater in the lawn than the meadow. Lawns can potentially provide hotspots of denitrification in a semi-arid landscape that exceed some natural riparian ecosystems, whereas DON may represent an increasingly important form of N loss from lawns.

Electronic supplementary material The online version of this article (doi:10.1007/s11252-016-0538-0) contains supplementary material, which is available to authorized users.

✉ Steven J. Hall
stevenjh@iastate.edu

¹ Global Change and Sustainability Center, University of Utah, Salt Lake City, UT, USA

² Department of Ecology, Evolution, and Organismal Biology, Iowa State University, 251 Bessey Hall, Iowa State University, Ames, IA 50011, USA

³ Department of Biology and the Ecology Center, Utah State University, Logan, UT, USA

⁴ Department of Plants, Soils, and Climate, Utah State University, Logan, UT, USA

⁵ Department of Biology, University of Utah, Salt Lake City, UT, USA

Keywords Denitrification · Dissolved organic nitrogen · Lawn · Nitrogen leaching · Snowmelt · Soil oxygen

Introduction

Urban development and landscape management often dramatically alter the biophysical structure and nutrient dynamics of the ecosystems they replace. Elevated inputs of nitrogen (N) in urban ecosystems contribute to water quality impairment, eutrophication, and shifts in species composition, making the impacts of urbanization on ecosystem nitrogen (N) dynamics particularly relevant to environmental policy (Bernhardt et al. 2008; Pickett et al. 2011). Lawns represent a dominant land cover in urban and suburban areas of the United States and receive substantial yet highly variable N inputs, ranging from zero to hundreds of kg of N ha⁻¹ y⁻¹ (Barton and Colmer 2006). Yet, total N inputs from fertilizer and atmospheric deposition are often of similar magnitude at the scale of urban watersheds (Baker et al. 2001; Groffman et al. 2004), so understanding the fate of anthropogenic N inputs across both lawns and remnant natural ecosystems is important from the perspective of N mass balance. Relatively few studies have assessed the biogeochemical cycling and fate of N in lawns in comparison with the ecosystems they replace (Kaye et al. 2005; Zhu et al. 2006; Raciti et al. 2008; Groffman et al. 2009b; Raciti et al. 2011b; Raciti et al. 2011c). This is particularly true for urban ecosystems in the arid western United States (Kaye et al. 2005; Zhu et al. 2006; Wang et al. 2014). This region is undergoing rapid urbanization and concomitant increases in water demand, coupled with increased N deposition and changing precipitation dynamics (Fenn et al. 2003; MacDonald 2010) that are likely to impact ecosystem N cycling.

Urban lawns have the capacity to leach large quantities of fertilizer N as nitrate (NO₃⁻), yet the importance of N leaching losses remains contested and variable among urban ecosystems (Barton and Colmer 2006; Groffman et al. 2009b). Lawn fertilizer can potentially represent a dominant (Hale et al. 2014) or minor (Kaushal et al. 2011) source of N to urban waterways, likely due to variability in hydrologic flowpaths and soil biogeochemical N dynamics. Lawns frequently increase plant and soil microbial productivity relative to the ecosystems they replace, especially when water-limited systems become irrigated, and substantial quantities of added mineral N can accumulate in microbial biomass and soil organic matter (Kaye et al. 2005; Frank et al. 2006; Raciti et al. 2008; Raciti et al. 2011a; Wang et al. 2014). Microbial N assimilation has long been known to contribute to N retention during spring runoff in seasonally snow-covered ecosystems (Zak et al. 1990). However, dissolved organic N (DON) leaching is another potentially important form of N loss that has not been quantified in most previous studies of lawn N dynamics (Barton and Colmer 2006). Importantly, DON can represent a substantial fraction of hydrologic N exports in both urban and natural watersheds (Neff et al. 2003; Janke et al. 2014). Lawns could potentially contribute substantially to these fluxes in soils where microbial and plant N demand is sufficient to assimilate mineral N inputs, and sorption capacity for the retention of DON-rich decomposition products ultimately becomes saturated (e.g. Six et al. 2002).

While snowmelt and rainfall can promote N leaching, denitrification also has the potential to ameliorate aqueous N losses from lawns under wet conditions. Baltimore lawn soils showed relatively low background rates of denitrification under typical field conditions, but rates spiked under brief periods of saturating soil moisture and could account for approximately 13 % of N inputs (Raciti et al. 2011a). Experiments in southern California suggested a similar

fraction of N was lost from denitrification (Wang et al. 2014). However, the relative importance of denitrification in lawns vs. remnant natural ecosystems has received little attention. Natural riparian ecosystems have long been known to be important sites for N retention and denitrification (McClain et al. 2003). Yet, in semi-arid landscapes where many surface riparian soils remain dry for much of the year (Dawson and Ehleringer 1991), irrigated lawns could potentially represent hotspots of denitrification.

In addition to the large ecological differences between urban lawns and remnant natural ecosystems, climate variability could influence the fate of anthropogenic N in urbanizing areas. In semi-arid montane regions of the western United States and similar regions worldwide, winter snow is the primary source of precipitation, and snowmelt is the dominant hydrologic event that transports nutrients from surface soils to the ground/surface water system (Brooks et al. 1999). The characteristics of winter snowpacks not only influence water supply, but can also impact plant and microbial nutrient cycling. A rich literature has demonstrated the importance of winter snow depth and duration in regulating soil frost, wintertime microbial activity, and root survival, which are often closely coupled to subsequent patterns of N retention and loss (Brooks et al. 2011). For example, persistent snowpacks increase winter soil temperature and often promote subnivean microbial N assimilation and denitrification, whereas sparse or intermittent snow-cover often decreases microbial activity and increases root mortality and subsequent N losses (Brooks and Williams 1999; Fitzhugh et al. 2001; Durán et al. 2013; Campbell et al. 2014). Few studies have examined the impact of snow variability on biogeochemical dynamics in urban ecosystems, although Durán et al. (2013) found strong effects of snow removal on subsequent N pools and fluxes in an urban lawn. It remains unclear how the response of N biogeochemical cycling to variation in snowpack cover and soil frost might differ across land use gradients.

Here, we compared soil N dynamics between two ecosystems that represent end-members of a watershed land-use continuum spanning an undeveloped montane canyon to a major metropolitan area (Salt Lake City, UT) over a distance of several km. These sites included a riparian meadow (hereafter “meadow”) in the montane canyon, and a 13 year-old upland lawn. We used a plot-scale snow manipulation experiment at each site to evaluate potential impacts of winter climate variability on N transformations and losses during and following the period of snowmelt, the period of maximal potential N leaching losses in this biome. We tested three hypotheses: (1) Microbial biomass N accumulation and denitrification decline with snow removal at both natural and urban sites as a consequence of increased freeze/thaw events and drier, oxic conditions; (2) natural and urban sites display similarly low N leaching under ambient snow conditions, despite differences in fertilization, due to the maintenance of low mineral N pools by assimilation and denitrification; (3) DON represents an important form of N loss from the urban lawn.

Methods

Site description

Study sites were located in the Red Butte Creek watershed, spanning urban Salt Lake City, Utah, and Red Butte Canyon in the adjacent Wasatch Mountains. This region experiences a dry-summer continental climate where precipitation is dominated by winter snow and spring rain, with occasional summer monsoonal storms. Red Butte Canyon represents an unusually

undisturbed landscape in close proximity to a major city, where public access has been restricted for over 100 years (Ehleringer et al. 1992). The University of Utah campus abuts the natural area and represents a stark transition to a landscape dominated by pavement, buildings, and managed lawns. We selected sites that represent two dominant near-riparian ecosystems in this watershed: a natural riparian meadow in Red Butte Canyon (elevation 1760 m), and a managed lawn (elevation 1500 m) located 4 km downstream. Both sites received at least $4 \text{ kg N ha}^{-1} \text{ y}^{-1}$ in wet + dry deposition from 2013 to 2014 (S. Hall, in review). The meadow often experiences greater snowfall due to orographic precipitation (e.g. Fig. 1).

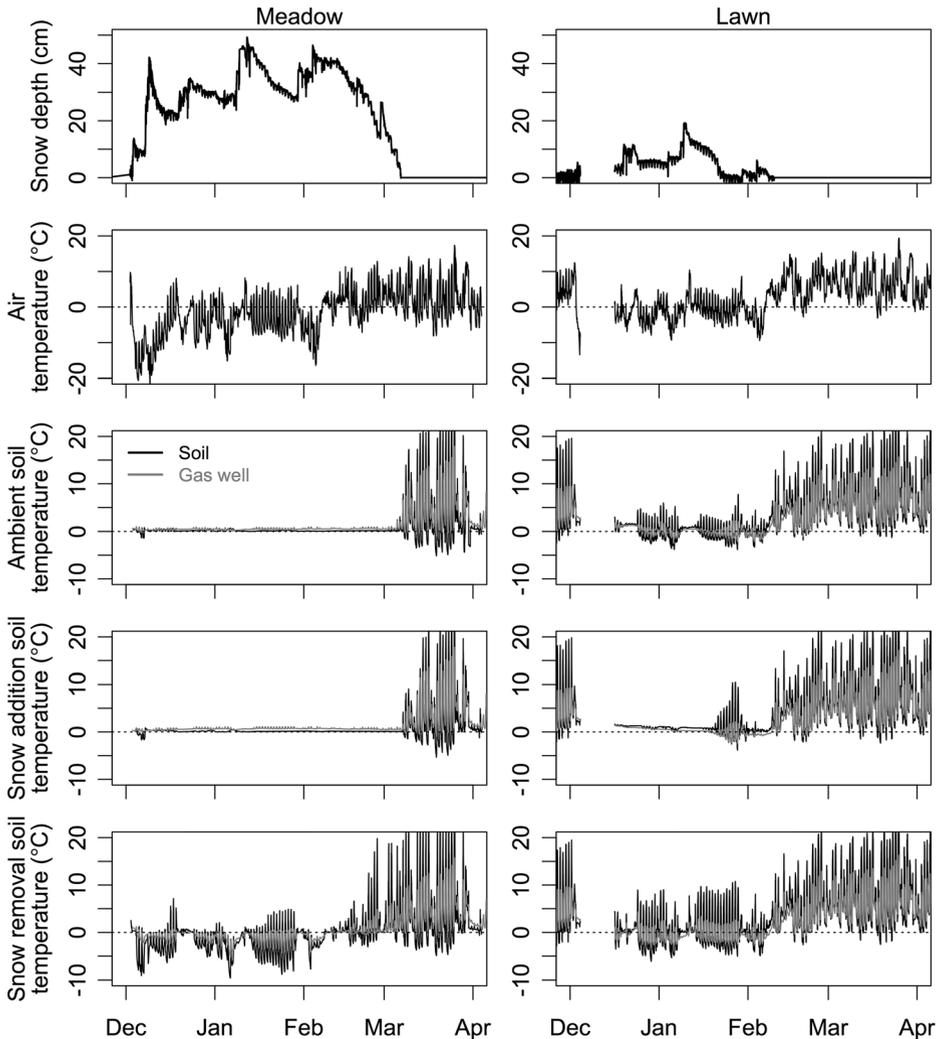


Fig. 1 Temporal trends in snow depth, air temperature, and soil temperature by snow manipulation treatment for the meadow and lawn sites (left and right columns, respectively). Dotted horizontal lines denote 0 °C. Soil temperature panels for “Ambient,” “Snow addition,” and “Snow removal” represent plots where snow was respectively unaltered, supplemented, or removed. Soil temperature was measured using vertical probes that spanned 0–8 cm soil depth (*black lines*), and using thermistors attached to the buried O₂ sensors (*grey lines*)

The meadow was adjacent to a second-order stream and had level topography with deep underlying alluvial soils (> 3 m depth). Meadow vegetation was dominated by grasses and forbs including *Helimeris multiflora*, *Poa pratensis*, *Bromus carinatus*, *Dactylis glomerata*, *Elymus glaucus*, *Achillea millefolium*, and *Melilotus officinalis*, with scattered *Acer negundo* (box elder) trees. The meadow soil was a loamy Mollisol formed from limestone and sandstone alluvium, and contained large (> 5 cm) carbonate concretions. Carbonate clasts persist over pedogenic timescales in these surface soils despite the presence of periodic moisture saturation; in fact, there are large carbonate features in the channel of Red Butte Creek. The majority of herbaceous plant roots were located above 30 cm depth, although the scattered riparian trees have much deeper roots (Dawson and Ehleringer 1991). Year-round bioturbation by rodents was indicated by soil disturbance and damage to buried instrumentation.

The lawn site was established in 2001 as a layer of sod and local soil overlying several meters of rocks and construction debris, according to typical local construction practices. The lawn surface soil was a sandy loam from a nearby sod farm, with a depth of approximately 15 cm to the underlying soil. Both the meadow and lawn soils had alkaline pH and similar bulk density (Table 1). Dominant plants at the lawn site were *Poa pratensis*, *Lolium perenne*, and *Trifolium campestre*. The lawn received N fertilizer inputs of approximately $100 \text{ kg N ha}^{-1} \text{ y}^{-1}$, half applied as a slow-release polymer-coated urea (25–3–10 ratio of N–P–K + 3 % iron) in July, and the other half applied as ammonium sulfate (21–0–0) in October. The lawn was irrigated from May through October to replenish 80 % of potential evapotranspiration (ET) by sprinklers three days per week. Mowing was conducted every two to three weeks with clippings left on the lawn surface. The lawn was mechanically aerated twice a year.

Experimental design and sensor measurements

A $14 \times 10\text{-m}$ grid was established at each site and divided into 12 plots measuring $2 \times 2 \text{ m}^2$, separated by equivalent buffers. We assigned plots randomly to one of three treatments: snow removal, snow addition, or ambient snow. Snow was moved from the removal to the addition plots within 24 h of each snowstorm from Dec 2013 to March 2014. To avoid disturbance of surface soil by shoveling, ~ 1 cm of snow remained on the surface of snow removal plots. At each site, three plots from each treatment were instrumented with SO-110 O_2 /temperature sensors (Apogee Instruments, Logan UT) and ACC-SEN-SDIP soil moisture/temperature sensors (Acclima Inc., Meridian ID). The O_2 sensors were installed in 10 cm polyvinylchloride gas wells (3.8 cm diameter) sealed with a cap on the top and a polytetrafluoroethylene membrane (200–07, International Polymer Engineering, Tempe AZ) on the bottom. Gas wells were installed flush with undisturbed soil at 10 cm depth after removing an equivalent soil volume. Gas well temperature was corrected by 1°C to account for a heating element on the O_2 sensor. Soil moisture sensors were installed vertically and reflect depth-integrated values from 0 to 8 cm. Climate stations measuring air temperature, snow depth, precipitation, wind speed, and net radiation were co-located with each site as part of the iUTAH ecohydrological observatory (http://data.iutahepscor.org/mdf/Data/Gamut_Network/). At the meadow site, a soil pit was instrumented with additional soil moisture sensors at depths of 5, 10, 20, 50, and 100 cm. All sensor data were recorded at 15-min intervals.

Table 1 Soil biogeochemical parameters measured at the lawn and meadow sites. Values represent means with standard errors in parentheses. Concentrations of Fe(II) were measured immediately following snowmelt in 0.5 M HCl extractions. The ratio Fe(II)/Fe_{HCl} represents the fraction of HCl-soluble Fe that was reduced (an index of reducing conditions in soil microsites)

Site	Depth	Litter N (%)	Soil N (%)	Fe(II) ($\mu\text{g g}^{-1}$)	Fe(II)/Fe _{HCl}	pH	Litter (g m^{-2})	Bulk density (g cm^{-3})	Litter N (g m^{-2})	Soil N (g m^{-2})
Lawn	0–5	1.41 (0.07)	0.36 (0.02)	53 (7)	0.46 (0.08)	7.53 (0.02)	380 (120)	0.86 (0.02)	5.3 (0.5)	150 (20)
	5–15	–	0.12 (0.01)	23 (1)	0.32 (0.08)	7.83 (0.03)	–	–	–	180 (30)
Meadow	0–5	0.82 (0.04)	0.41 (0.02)	7 (1)	0.04 (0.01)	7.73 (0.01)	1280 (40)	0.87 (0.02)	9.9 (0.9)	180 (20)
	5–15	–	0.36 (0.01)	8 (1)	0.03 (0)	7.69 (0.02)	–	–	–	310 (30)
	15–35	–	0.23 (0.01)	7 (1)	0.03 (0)	7.78 (0.02)	–	–	–	400 (40)

Soil sampling and analysis

Super Quartz tension lysimeters (Prenart, Denmark) were installed in each plot to sample the soil solution. We used tension as opposed to zero-tension lysimeters given that they can be installed with minimal disturbance, necessary given our plot lay-out, and can be sampled under vacuum representative of field moisture capacity to capture largely mobile water pools. Lysimeters were installed in holes generated by pushing a metal rod (1.8 cm diameter) into the soil at a 45 ° angle to depths of 35 cm (meadow) and 15 cm (lawn). A silica/water suspension was added via a plastic pipe, the lysimeter body was pushed to the bottom of the hole, and soil was backfilled to the surface. Lysimeter depths represented the lower limit of the majority of herbaceous root biomass at these sites. Glass collection bottles (500 ml) were buried outside of treatment plots and covered with foam insulation to prevent water samples from freezing, and lysimeters were sampled by applying a vacuum of -28 kPa relative to ambient pressure (86 kPa). We chose -28 kPa as a sampling pressure because it is similar to a common definition of field capacity (-30 kPa; Schaetzl and Anderson 2005), yielding a largely mobile pool of soil water. We applied vacuum to lysimeters on approximately weekly intervals during the period of snowmelt and spring rains. Several lysimeters collected water inconsistently due to spatial variation in soil moisture or rodent damage to tubing, so sample sizes varied among dates. We sampled lawn lysimeters from January – April 2014, with several additional samples in summer/fall (14 sampling dates, total $n = 123$). Meadow lysimeters were sampled from March through May (13 sampling dates, total $n = 78$); water samples could not be collected outside of this period due to low soil water potential. Samples were filtered through pre-combusted Whatman GF/F filters (pore size ~ 0.7 μm), refrigerated at 4 °C, and NH_4^+ , NO_3^- , and NO_2^- concentrations measured by ion chromatography (Metrohm Compact IC, Riverview, FL). A representative subset of lysimeter samples with sufficient volume was analyzed for total dissolved nitrogen (Shimadzu TOC-V, Columbia, MD), and DON was calculated by difference with inorganic N.

Soils were collected for chemical analysis in November 2013 prior to snowfall, and in February or March 2014 following snowmelt for the lawn and meadow sites, respectively. In each plot, we sampled 0–5 and 5–15 cm depths using a 6-cm diameter stainless-steel auger. Soil from 15 to 35 cm was collected in the meadow site but could not be sampled at this depth in the lawn due to buried rocks. In each plot, samples were collected from three randomly selected points within 0.5 m of the plot center and composited by depth. Soil samples were extracted in 1:5 (mass basis) slurries with 2 M potassium chloride (KCl) within 24 h. Additional samples were extracted in 0.5 M potassium sulfate (K_2SO_4), with duplicates incubated in K_2SO_4 plus liquid chloroform (40 ml and 0.5 ml, respectively) for microbial biomass analysis following Fierer and Schimel (2003). Concentrations of N as NO_3^- and NH_4^+ in the KCl extracts, and NO_3^- in the K_2SO_4 extracts (following a persulfate digestion), were measured colorimetrically using a Lachat (Milwaukee, WI) autoanalyzer. Microbial biomass N was calculated as the difference between extractable N in K_2SO_4 extracts which received chloroform and those that did not, after applying a correction factor of 0.54 to chloroformed extracts to account for incomplete recovery (Brookes et al. 1985).

We measured potential denitrification rates in shaken slurries of spring soil samples using the acetylene inhibition method (Groffman et al. 1999). Incubations were conducted at 4 °C (representative of field daytime soil temperature following snowmelt) and at 25 °C (to facilitate comparisons with other studies). Headspace gas was sampled after 45 and 90 min and

analyzed for nitrous oxide (N_2O) concentrations on a gas chromatograph (Shimadzu, MD) using an electron capture detector. Rates of N_2O production calculated from the 45 and 90-min samples agreed closely (typically within 5 %), so we report the 45-min data here.

Additional subsamples of the February/March soils were extracted in the field in 0.5 M HCl for analysis of reduced and oxidized iron (Fe) as a redox indicator. Briefly, some microbes can use Fe oxides directly (or indirectly via humic substances) as electron acceptors during anaerobic metabolism, converting oxidized Fe (Fe(III)) to reduced Fe (Fe(II)). Concentrations of Fe(II), and the ratio of Fe(II) to total Fe soluble in 0.5 M HCl (F_{eHCl}), represent indices of reducing conditions at the scale of soil microsites (Hall et al. 2013). Differences in energy yield among redox couples imply that if Fe reduction is occurring in anaerobic microsites, then denitrification in these microsites should have already occurred. Immediately after sampling, soil samples were homogenized and immersed in a 1:20 ratio with 0.5 M HCl. Samples were extracted for 2 h on a rotary shaker, filtered through 0.45 μ m membranes, and analyzed colorimetrically for Fe(II) and Fe(III) concentrations using a ferrozine method (Hall et al. 2013).

To compare soil N stocks among sites, the November 2013 soil samples were air dried and sieved to 2 mm. Samples were ground and analyzed for N concentrations on an elemental analyzer. Because carbonate clasts were extremely abundant in many meadow samples, we did not attempt to remove carbonates from soils prior to analysis as this would greatly inflate the apparent mass concentrations of N.

Modeling soil moisture fluxes

We used the Hydrus 1-D (Simunek et al. 2013) numerical model in combination with lysimeter data to simulate soil water infiltration and N leaching past the depths of soil sampling. We employed meteorological data measured at each site as model boundary conditions, and informed the numerical model with measured soil moisture data to fit soil hydrological parameters using inverse estimation. The combination of measured precipitation inputs and estimated hydraulic parameters was then used to simulate infiltration. Initial hydrologic parameters were estimated for each horizon using measured texture data and the Rosetta pedotransfer function (Schaap et al. 2001). For the meadow, we used horizons defined in three replicate pits dug to 1 m depth; previous work at this site documented the presence of unconsolidated soil/saprolite to a depth of >3 m, which we modeled using an additional soil layer with similar characteristics as the deepest measured horizon. For the lawn, in addition to the measured surface horizons we also fit hydraulic parameters for an arbitrary subsoil layer between 0.15 and 3 m depth, the presence of which was necessary to reproduce measured surface soil moisture dynamics. Because surface runoff was never observed at either site during heavy rainfall events, we used “surface ponding” as the atmospheric boundary condition. Free drainage was used for the lower boundary condition because water tables were at least several meters below the soil surface at each site. Lateral water inputs were assumed negligible at both sites due to the flat surface. Also, the deep meadow soils had no confining horizons and were located in a valley approximately 20 m from the adjacent hillslope, so shallow subsurface runoff likely infiltrated at the slope break as opposed to moving laterally across the meadow. This interpretation is supported by the soil moisture data, which did not show wetting from below. Modeled water fluxes were recorded at multiple depths (15, 35, and 100 cm in the meadow, and 5 and 15 cm in the lawn) and summed over time to yield cumulative net infiltration. Water fluxes at 15 and 35 cm in the lawn and meadow,

respectively, were multiplied by measured lysimeter N concentrations to calculate N leaching losses below the predominant rooting zones. We initiated model simulations immediately following snowmelt at each site due to difficulties in matching observed soil water dynamics with modeled snowmelt water inputs. We accounted for infiltration prior to complete snowmelt using mass balance, where any snowmelt that exceeded field capacity from the surface to a defined soil depth was assumed to have infiltrated below that depth.

Statistical analysis

Snow manipulation treatment effects on soil N and Fe pools and potential denitrification rates for the spring soil samples were assessed using linear models (ie, ANOVA) fit in R version 3.1.0 (www.R-project.org). In the absence of treatment effects in separate models fit for each site, we fit models testing for site differences irrespective of treatment. Trends in N over time were assessed using mixed-effects models fit using the `lme` function in R (Pinheiro et al. 2014) with plots as random effects to account for repeated sampling.

Results

Effects of snow manipulation on soil temperature and moisture

Snow depth and duration of snow cover in the meadow exceeded the lawn due to orographic precipitation and subtle differences in air temperature (Fig. 1) and energy balance. Meadow soil temperature in the ambient and snow addition treatments slightly exceeded 0 °C during most of the snow-covered period (Fig. 1), with mean (sd) values of 0.2 ± 0.2 and 0.1 ± 0.2 °C, respectively. In contrast, snow removal and consequent radiative energy exchange led to large diurnal oscillations in meadow soil temperature during the same period. During late January there were 14 consecutive days in the snow removal treatment where soil temperature varied from < -5 to > 0 °C, indicative of soil freeze/thaw cycles, and mean soil temperature was -1.3 ± 3.2 °C. In the lawn site, soil temperature in the control and snow removal plots fluctuated around zero for most of December – February, measuring 0.2 ± 1.7 and -2.0 ± 2.5 °C, respectively, (Fig. 1). In contrast, soil temperature in the snow addition plots remained above zero (0.9 ± 1.3 °C) for most of the snow-covered period, aside from a brief period of fluctuation during a melt event in late January (Fig. 1).

Temporal variation in moisture and O₂

Surface soil moisture peaked following snowmelt periods in both sites, and was elevated following sporadic summer rain events (Fig. 2). Temporal patterns in snow-cover and precipitation are shown in Figs 1 and 2, respectively. Meadow surface soil volumetric moisture was substantially lower than in the lawn and varied between 10 and 30 % during the period of snowmelt and spring rain from February through May. Snow removal plots tended to have the lowest surface soil moisture by several percent relative to the other treatments. By June, meadow soil moisture levels became similar among plots and remained below 10 % for much of the summer, aside from brief periods following storms. In contrast, volumetric soil moisture in the lawn exceeded 30 % from January through most of April. Snow manipulation treatments in the lawn had similar soil moisture. Following a relatively dry period in April and May,

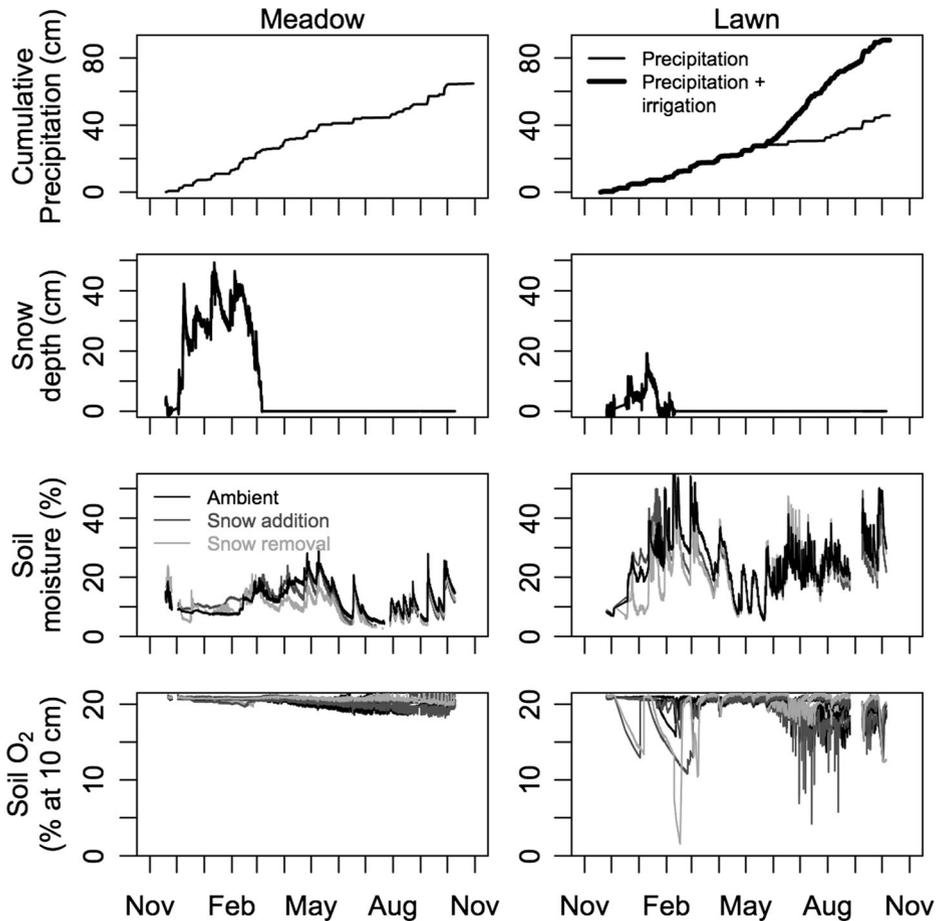


Fig. 2 Temporal trends in precipitation, snow depth, volumetric soil moisture, and soil O_2 at the meadow and lawn sites. Each soil moisture line represents the mean of three independent measurements. Lines on the soil O_2 plots represent individual sensors shown separately to illustrate spatial heterogeneity ($n = 9$ for each site). Soil O_2 did not vary consistently among snow manipulation treatments at either site

irrigation maintained the lawn soil moisture above 20 % from June through October and generated periodic spikes during irrigation events.

In the meadow, soil O_2 mixing ratios did not respond to moisture variability, whereas in the lawn, moisture inputs from snowmelt and irrigation events were often associated with large O_2 decreases, despite the fact that moisture inputs from snowmelt were smaller than in the meadow (Fig. 2). Snow manipulation treatments were not associated with any consistent patterns in O_2 at either site. Lawn soil O_2 showed high spatial as well as temporal variability, where individual sensors varied between 1.6–20.9 % O_2 during periods of winter snowmelt and several sensors showed trends of slowly declining O_2 that persisted for several weeks (Fig. 2). Following snowmelt (February), lawn O_2 remained above 18 % at all sensors until irrigation commenced in June. Following individual irrigation events, several sensors displayed sharp O_2 declines and oscillations of approximately 2–10 % (Supplemental Fig. 1). Post-snowmelt Fe(II) concentrations were significantly greater ($p < 0.0001$) in the

lawn than in the meadow, as were $\text{Fe(II)/Fe}_{\text{HCl}}$ ratios (Table 1). These indicated a much greater prevalence of reducing soil microsites in the lawn, where almost half of HCl-extractable Fe was present in the reduced form in 0–5 cm soils.

Variation in extractable soil N pools within and among sites

None of the measured soil N pools (NH_4^+ , NO_3^- , extractable organic N, microbial biomass N) were significantly affected by snow manipulation treatments at either the meadow or lawn sites (treatment $p > 0.05$ for each N pool measured in spring at the meadow and lawn; Supplemental Fig. 2). Mineral N pools varied relatively little between the meadow and lawn despite the large annual fertilizer inputs to the latter site (Fig. 3). Ammonium concentrations in lawn 0–5 cm soil were three-fold and significantly greater ($p < 0.001$) than the meadow in both fall and spring samples, although concentrations in 5–15 cm soil and from 0 to 15 cm were similar between sites ($p > 0.05$). Ammonium concentrations increased from fall to spring in the lawn 0–5 cm samples ($p < 0.01$) and approximately doubled in the meadow in 5–15 and 15–35 cm samples over this period ($p < 0.001$), and showed no change in the other depths.

Nitrate concentrations differed among sites and over time (Fig. 3b). In the fall samples, NO_3^- was two-fold greater ($p < 0.001$) in the meadow 5–15 cm soil as compared with the lawn; 0–5 cm NO_3^- concentrations were similar between the two sites. In the spring samples,

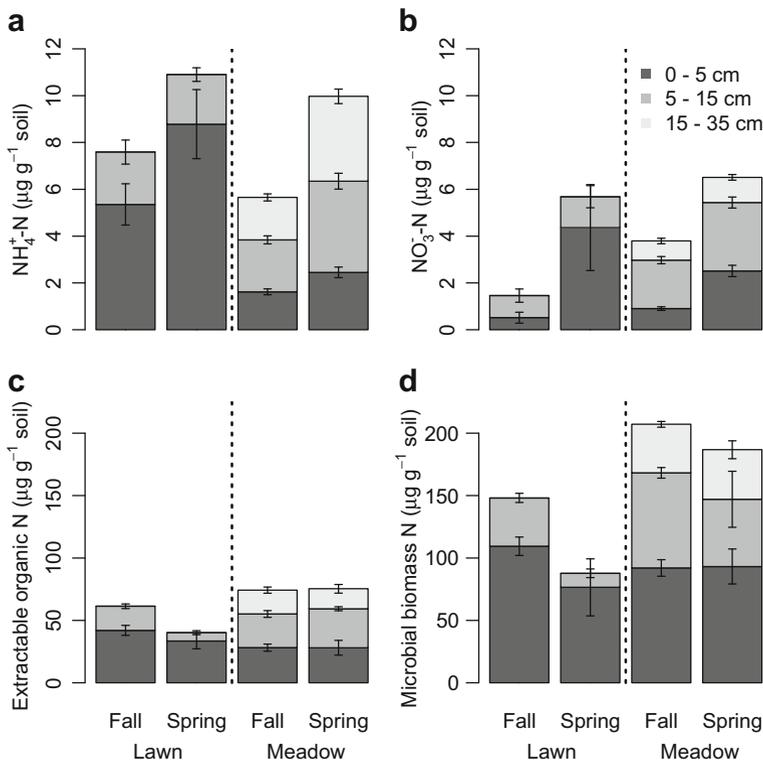


Fig. 3 Soil N pools (mean \pm SE) plotted by site, depth, and season (fall and spring). Fall samples were collected in November after plants had senesced, and spring samples were collected after snowmelt at each site. The lawn soil measured only 15 cm deep to underlying rocks

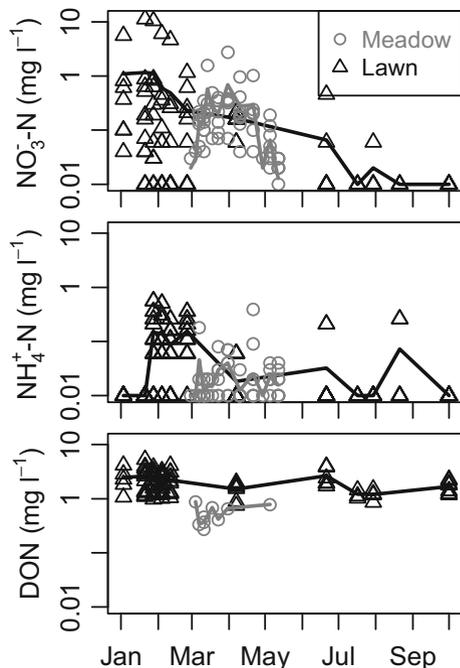
NO_3^- concentrations did not differ between the two sites at either depth. Nitrate concentrations significantly increased from fall to spring in both depths in the lawn ($p < 0.05$) and the meadow ($p < 0.001$). Lawn 0–5 cm soils showed the greatest change in NO_3^- , an approximately eight-fold increase, from fall to spring.

Potassium sulfate-extractable organic N and microbial biomass N showed less consistent variation as compared with mineral N (Fig. 3c, d). Extractable organic N significantly declined in both lawn depths from the fall to the spring ($p < 0.001$), but was similar over time in the meadow soils. Microbial biomass N did not differ between fall and spring samples at either site. In the fall, microbial biomass N was similar between lawn and meadow soils from 0 to 5 cm but was two-fold greater ($p < 0.001$) in meadow 5–15 cm soils as compared with the lawn. In the spring, microbial biomass N concentrations were highly variable and did not significantly differ among sites at any depth.

Spatial and temporal variation in soil solution N

Nitrate concentrations in lysimeters varied over three orders of magnitude among samples from the lawn and meadow but did not significantly differ between sites ($p = 0.53$), which had respective mean NO_3^- concentrations of 0.48 ± 0.16 and $0.24 \pm 0.05 \text{ mg N l}^{-1}$ (Fig. 4). Lawn lysimeter NO_3^- was greatest and most variable ($0.01\text{--}11.3 \text{ mg N l}^{-1}$) from January to March, and mean values were much lower ($\leq 0.1 \text{ mg N l}^{-1}$) from July to October. In the meadow, mean lysimeter NO_3^- concentrations were relatively consistent during the early period of snowmelt (March – April) and declined to $\leq 0.1 \text{ mg N l}^{-1}$ by early May (Fig. 4). Soil solution NH_4^+ concentrations were much lower than NO_3^- and did not significantly differ between sites ($p = 0.08$; Fig. 4). In contrast to the mineral N pools, DON measured in soil lysimeters strongly

Fig. 4 Temporal trends in meadow (circles) and lawn (triangles) soil solution NO_3^- , NH_4^+ , and DON measured in tension lysimeters installed below the predominant rooting depth at each site (35 and 15 cm at the meadow and lawn, respectively). Lines represent trends in mean values. Values below detection were assigned a value of 0.01 mg N l^{-1} for illustration on a logarithmic scale. Water potential in the meadow was high enough to permit lysimeter sample collection only during the period of snowmelt and shortly thereafter, whereas the lawn soils were wet enough to be sampled from January through October



differed between the lawn and the meadow ($p < 0.001$; Fig. 4). Mean DON concentrations were four-fold greater in the lawn ($2.07 \pm 0.09 \text{ mg N l}^{-1}$) than the meadow ($0.54 \pm 0.03 \text{ mg N l}^{-1}$), and varied little over time.

Potential denitrification rates following spring snowmelt

Snow manipulation had no significant impact on potential denitrification at either 4 or 25 °C ($p > 0.05$; Supplemental Fig. 3), but rates strongly differed between sites and depths (Fig. 5). Potential denitrification in 0–5 cm lawn soils measured 16 times the rates in meadow soils at 4 °C ($p < 0.0001$), and were six times greater than the meadow rates at 25 °C ($p < 0.001$; Fig. 5). Potential denitrification at 4 °C declined with depth in the lawn soils to only 9 % of the 0–5 cm rates ($p < 0.0001$); rates in the meadow soil were statistically equivalent among depth increments. At 25 °C, potential denitrification in the lawn soil was four-fold greater in 0–5 cm soil as compared with 5–15 cm soil ($p < 0.001$), and also significantly declined with depth in the meadow ($p < 0.05$; Fig. 5).

Modeled water infiltration and N leaching

Hydrus model simulations for the lawn and meadow adequately represented temporal dynamics in soil moisture. The lawn and meadow sites had root mean squared errors of 1.1 and 1.4 % soil moisture, respectively, while R^2 values were 0.90 and 0.81 at these sites. In the meadow, snowmelt and rain from February to early May represented the large majority of water inputs to soil below 35 cm (Supplemental Fig. 4). Similarly, in the lawn, infiltration during February and March dominated water inputs below 15 cm (Supplemental Fig. 4). Between February and October, net infiltration in the meadow at soil depths of 15, 35, and 100 cm, measured 27, 15, and 1 cm of water, respectively. In the lawn, net infiltration below 15 cm measured only 7 cm over this period. These infiltration rates implied small hydrologic fluxes of total N ($\text{NO}_3^- +$

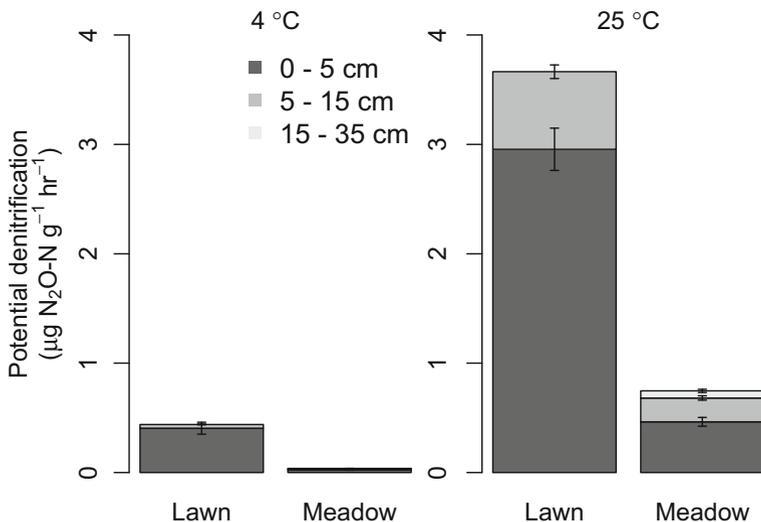


Fig. 5 Potential soil denitrification rates measured after snowmelt plotted by site and depth for incubation temperature of 4 and 25 °C, respectively. Differences among snow manipulation treatments were not significant

NH_4^+ + DON) below the majority of root biomass, measuring approximately $1.8 \text{ kg N ha}^{-1} \text{ y}^{-1}$ past 15 cm in the lawn, and $1.2 \text{ kg N ha}^{-1} \text{ y}^{-1}$ past 35 cm in the meadow.

Bulk soil and litter N concentrations

Total N was similar between the lawn and meadow 0–5 cm soils and declined with depth at both sites (Table 1). However, meadow soils had much greater surface soil N stocks to 15 cm than the lawn, measuring 490 ± 50 and $260 \pm 30 \text{ g N m}^{-2}$, respectively ($p < 0.001$). Litter N concentrations were almost two-fold greater in the lawn than the meadow, although litter N stocks were twice as high in the meadow ($p < 0.001$) due to greater litter mass (Table 1).

Discussion

Snow manipulation affected the soil biophysical environment but not N cycling

Increased soil frost during snow-free winter conditions often leads to increased fine root mortality, and decreased microbial biomass, denitrification, and NO_3^- uptake (Groffman et al. 2001; Groffman et al. 2009a; Campbell et al. 2014). In our study, snow manipulation altered surface soil temperature and freeze/thaw dynamics among treatments, yet contrary to our first hypothesis we found no evidence for any N-cycling impacts at either site. Snow manipulation did not significantly affect extractable N pools, microbial biomass, or potential denitrification. This finding may have been influenced by the relatively brief duration of continuous snow cover at these sites (~1.5 months at the lawn, ~3 months at the meadow) relative to many of the sub-alpine and alpine sites studied previously. Also, temperature often exceeded $-5 \text{ }^\circ\text{C}$, potentially allowing persistence of some liquid water (Brooks et al. 2011). Our finding of no N-cycling responses to snow removal at both sites suggests that the differences in snow depth and snow cover between these sites (Fig. 1) may not have been a critical factor in explaining differences in N pools and fluxes between these sites. While numerous studies have shown that shallow and intermittent snowpacks are often associated with increased N export, other work has also shown minimal impacts of soil frost on N pools or microbial biomass (Groffman et al. 2001; Groffman et al. 2010). Identifying the biophysical characteristics associated with resilience in soil N cycling under a range of snowpack and microclimatic conditions remains an important knowledge gap, given the potentially detrimental effects of frost-induced root mortality and N loss (Campbell et al. 2014). The observation of multiple years of consistently low snow accumulation might be necessary to discern biogeochemical impacts of soil frost in this system.

Interestingly, snow manipulation also had no discernable impact on surface soil O_2 concentrations. Previous work showed that deep snow cover can result in soil CO_2 accumulation and corresponding depletion of O_2 (Solomon and Cerling 1987), which in combination with warmer subnivean soil temperature can stimulate winter microbial N assimilation and denitrification (Brooks et al. 1997). Alternatively, variation in soil surface temperature and freeze/thaw dynamics can promote ice formation that inhibits soil-atmosphere gas transport and dramatically increases soil CO_2 (Moyes and Bowling 2013), leading to anaerobiosis. Here, spatial heterogeneity in soil biophysical properties outweighed any impacts of snow manipulation on bulk surface soil O_2 , which varied tremendously in the lawn and minimally at the meadow.

Convergence in mineral N pools between sites and the importance of DON

Extractable N pools were notably similar between the lawn and natural meadow sites despite substantial lawn fertilizer application, supporting our second hypothesis. Increased mineral N from fall to spring was consistent with C (as opposed to N) limitation of microbial growth in these sites during winter, a pattern observed elsewhere (Brooks et al., 2011). The pattern of greater fall NO_3^- in the meadow than the lawn may have been due to strong water limitation of plant growth, resulting in decreased plant N uptake relative to mineralization and nitrification. In contrast, the lawn was not likely water limited, but was possibly N limited despite the mid-summer application of fertilizer, as suggested by consistently low lysimeter NH_4^+ and NO_3^- concentrations during summer. Anecdotally, we observed an increased abundance of clover throughout the lawn (as well as the meadow) over the summer, pointing to N-limiting conditions where legumes could compete with the more productive grasses. High biological N demand combined with low water fluxes at the lysimeter sampling depth (15 cm) likely explained the fact that lysimeter N concentrations did not respond after the July fertilizer application. These patterns were confirmed by low mineral N concentrations in the 5–15 cm depth in the fall soil samples.

The similarities in mineral N between our sites contrasts with data from the Baltimore, Maryland metropolitan area, where Raciti et al. (2011c) found approximately five-fold greater mean NO_3^- mass to a depth of one meter in urban lawns than in forests, and where Groffman et al. (2009b) frequently observed greater NO_3^- leaching from lawns than nearby forested ecosystems. Similarly, in Phoenix, Arizona, Zhu et al. (2006) also found three-fold greater NO_3^- concentrations in urban vs. desert soils. Unfortunately, we cannot necessarily extrapolate our present findings to the broader landscape given a lack of site replication. However, our sites represented useful end-members for assessing differences in N cycling between representative wildland and urban ecosystems in the Red Butte Creek watershed. Our study provides an example of tight mineral N cycling in a fertilized lawn ecosystem, lending further support to the hypothesis that lawns can maintain similarly low standing pools of mineral N as natural ecosystems, in spite of substantial fertilizer inputs, due to rapid N assimilation and denitrification losses (Raciti et al. 2011a; Raciti et al. 2011b; Wang et al. 2014).

These findings are notable given that both natural grasslands and lawns can potentially leach significant amounts of N under conditions of ample precipitation or irrigation. For example, leaching rates in low-diversity German grasslands measured between 8 and 16 kg $\text{NO}_3^- \text{N ha}^{-1} \text{y}^{-1}$ (Scherer-Lorenzen et al. 2003), and up to 70 kg $\text{N ha}^{-1} \text{y}^{-1}$ from managed lawns (Barton and Colmer 2006). Yet, both systems can also exhibit minimal NO_3^- leaching (Scherer-Lorenzen et al. 2003; Barton and Colmer 2006) when biological demand is high and infiltration rates are low, as observed here. In contrast to the relatively low mineral N concentrations in lysimeter water from both sites, DON typically measured an order of magnitude greater than NO_3^- at the lawn, and was similar or greater than NO_3^- in the meadow. This finding supported our third hypothesis that DON represents a potentially important form of hydrologic N loss from lawns. A review of lawn N leaching studies indicated that measurements of DON have been broadly neglected (Barton and Colmer 2006). High DON concentrations could reflect initial stages of C saturation (e.g. Six et al. 2002) at this site, where fertilization and irrigation fuel high productivity and biomass decomposition, yet N-rich macromolecular decomposition products are apparently stabilized (ie, adsorbed, complexed, or occluded in aggregates) to a lesser degree. If the occurrence of high lawn DON concentrations is broadly generalizable across sites, especially sites with more permeable substrata, lawn

DON exports could potentially contribute to the substantial DON fluxes sometimes observed in urban aquatic ecosystems (Janke et al. 2014).

Differences in surface soil O₂ dynamics among sites

Riparian ecosystems often display low and/or dynamic bulk soil O₂ concentrations (Burgin et al. 2010; Morse et al. 2015) and high denitrification rates due to the confluence of increased soil moisture, organic matter, and NO₃⁻ supply. Here, the lawn showed much more dynamic and variable soil O₂ than the riparian meadow over the entire period of measurement, despite the fact that winter moisture inputs were approximately 30 % greater in the meadow as a consequence of orographic precipitation. These urban soils displayed frequent (days – weeks) fluctuations in O₂ with wide amplitude (potentially exceeding 10 %) analogous to upland tropical rainforest soils (Hall et al. 2013). Both sites had similar bulk density and organic matter contents in 0–5 cm soil (as reflected by total soil N content), yet lawn soil moisture was often twice as high as in the meadow despite the fact that they received similar overall surface moisture inputs (as a consequence of irrigation). It appears that the rocky lawn subsurface restricted water infiltration, as indicated by the relatively low 15-cm water fluxes modeled by Hydrus. This led to water accumulation near the soil surface following rainfall and irrigation events, contributing to decreased gaseous diffusion and O₂ depletion in microsites with suitably high respiration rates. Although the lawn surface was mechanically aerated on an annual basis, the relatively deep meadow soil was mixed by burrowing rodents to depths >35 cm. This mixing likely promoted infiltration of snowmelt and rain to meadow subsurface soil horizons, thus maintaining high bulk soil O₂ in surface horizons.

We cannot conclusively extend our inference from these sites to the entire watershed given the lack of site replication, but note that the patterns of deep and extensive bioturbation in the natural riparian areas and subsurface compaction in recently-constructed urban lawns appear broadly representative of this area. Many lawns in the urban reach of Red Butte Creek bordered by the University of Utah and related commercial land uses are constructed by applying sod and a thin layer of topsoil over a subsurface that has been leveled, compacted, and/or amended with rocks or gravel fill to achieve desired topographic features. These practices decrease hydraulic conductivity, potentially increasing variability in moisture and O₂ dynamics as we observed in our lawn site.

In our riparian meadow site, moisture and O₂ dynamics indicated hydrologic isolation from the nearby stream, a finding supported by previous studies of plant water sources at this site (Dawson and Ehleringer 1991). This pattern may be relatively common in semi-arid catchments in this region, where streamflow is primarily generated by infiltration of snowmelt to groundwater at high elevations (Covino and McGlynn 2007; Hall et al. 2016). Solute exports from surface riparian soils do not necessarily flow directly to Red Butte Creek in either the montane or urban reach, as indicated by stream solute data from multiple sites across this catchment (Hall et al. 2016). Although our meadow represents only one site, its surface soil moisture dynamics appear broadly representative of nearby sites in the Red Butte Creek watershed (D. Eiriksson, unpublished data; http://data.iutahepsc.org/mdf/Data/Gamut_Network/).

High potential denitrification rates in the lawn soils

High soil moisture, variable bulk O₂ concentrations, microsite-scale reducing conditions implied by the Fe redox data, and net over-winter N mineralization and nitrification appeared

to create a highly suitable environment for denitrification in the lawn soils during the spring snowmelt period. Springtime potential denitrification rates in the lawn soils were comparable to or greater than many measurements in riparian, wetland, and aquatic ecosystems, assessed at equivalent temperature. Roach and Grimm (2011) found potential denitrification rates of approximately 1, 1.5, and 5 $\mu\text{g N g}^{-1} \text{h}^{-1}$ in stream, floodplain, and lake sediments from Phoenix, AZ, which spanned the rates measured in our lawn surface soils (3 $\mu\text{g N g}^{-1} \text{h}^{-1}$). Our lawn potential denitrification rates also exceeded previous measurements from surface riparian soils and stormwater control wetlands (Groffman et al. 2002). Potential denitrification in the meadow surface soils from our study fell at the bottom end of this range, consistent with the high bulk soil O_2 concentrations that we measured. Our meadow potential denitrification rates were very similar to the “dry” riparian zone site measured by Burgin et al. (2010) as well as the lawns measured by Durán et al. (2013). Our finding of greater denitrification potential in surface than subsurface horizons also echoes previous results (Groffman et al. 2002), likely reflecting the importance of plant C and N inputs near the soil surface. The presence of high potential denitrification rates combined with abundant anaerobic microsites, as indicated by the Fe(II) data, demonstrate that lawns can provide hotspots of denitrification in a semi-arid landscape that potentially exceed the potential activity of natural riparian ecosystems, which are traditionally thought to dominate denitrification at the watershed scale (McClain et al. 2003).

Conclusions

The riparian meadow and managed lawn sites in this semi-arid watershed displayed surprising similarities in most soil N pools examined and yielded low hydrological N losses below the herbaceous plant rooting zone. Snow manipulation affected winter soil temperature dynamics but had no discernable effect on soil N pools or fluxes, in contrast to previous studies. Potential denitrification in the lawn exceeded the meadow, consistent with differences in redox dynamics between the two sites as reflected by measurements of bulk soil O_2 and Fe(II) in soil extractions. These findings support the hypothesis that lawns can maintain low standing mineral N pools and hydrologic losses despite substantial N inputs due to a combination of high N assimilation and denitrification rates (Groffman et al. 2009b; Raciti et al. 2011a; Raciti et al. 2011c; Wang et al. 2014). In contrast to mineral N, however, DON concentrations measured in soil lysimeters were high throughout the period of water availability, especially at the lawn where they consistently measured $\sim 2 \text{ mg N l}^{-1}$. As urban lawns age and capacity for soil organic matter retention begins to saturate, the transformation of fertilizer N to DON could be an important source of N loss from urban lawns.

Acknowledgments We thank Sue Pope, Lorenzo Lopez, and Marty Huebner from the University of Utah landscape staff for supporting our work on campus. We gratefully acknowledge field and lab assistance from Jillian Turner, Simone Jackson, Kendalynn Morris, Dave Eiriksson, Suvankar Chakraborty, Harrison Quinn, and Caitlin Szymanski. This research was supported by NSF EPSCoR grant IIA 1208732 awarded to Utah State University as part of the State of Utah Research Infrastructure Improvement Award. Any opinions, findings, and conclusions or recommendations expressed are those of the author(s) and do not necessarily reflect the views of the National Science Foundation.

References

- Baker LA, Hope D, Xu Y, et al. (2001) Nitrogen balance for the central Arizona-Phoenix (CAP) ecosystem. *Ecosystems* 4:582–602. doi:[10.1007/s10021-001-0031-2](https://doi.org/10.1007/s10021-001-0031-2)
- Barton L, Colmer TD (2006) Irrigation and fertiliser strategies for minimising nitrogen leaching from turfgrass. *Agric Water Manag* 80:160–175. doi:[10.1016/j.agwat.2005.07.011](https://doi.org/10.1016/j.agwat.2005.07.011)
- Bernhardt ES, Band LE, Walsh CJ, Berke PE (2008) Understanding, managing, and minimizing urban impacts on surface water nitrogen loading. *Ann N Y Acad Sci* 1134:61–96. doi:[10.1196/annals.1439.014](https://doi.org/10.1196/annals.1439.014)
- Brookes PC, Landman A, Pruden G, Jenkinson DS (1985) Chloroform fumigation and the release of soil nitrogen: A rapid direct extraction method to measure microbial biomass nitrogen in soil. *Soil Biol Biochem* 17:837–842. doi:[10.1016/0038-0717\(85\)90144-0](https://doi.org/10.1016/0038-0717(85)90144-0)
- Brooks PD, Campbell DH, Tonnesen KA, Heuer K (1999) Natural variability in N export from headwater catchments: snow cover controls on ecosystem N retention. *Hydrol Process* 13:2191–2201. doi:[10.1002/\(SICI\)1099-1085\(199910\)13:14/15<2191::AID-HYP849>3.0.CO;2-L](https://doi.org/10.1002/(SICI)1099-1085(199910)13:14/15<2191::AID-HYP849>3.0.CO;2-L)
- Brooks PD, Grogan P, Templer PH, et al. (2011) Carbon and nitrogen cycling in snow-covered environments. *Geogr Compass* 5:682–699. doi:[10.1111/j.1749-8198.2011.00420.x](https://doi.org/10.1111/j.1749-8198.2011.00420.x)
- Brooks PD, Schmidt SK, Williams MW (1997) Winter production of CO₂ and N₂O from alpine tundra: Environmental controls and relationship to inter-system C and N fluxes. *Oecologia* 110:403–413. doi:[10.1007/PL00008814](https://doi.org/10.1007/PL00008814)
- Brooks PD, Williams MW (1999) Snowpack controls on nitrogen cycling and export in seasonally snow-covered catchments. *Hydrol Process* 13:2177–2190. doi:[10.1002/\(SICI\)1099-1085\(199910\)13:14/15<2177::AID-HYP850>3.0.CO;2-V](https://doi.org/10.1002/(SICI)1099-1085(199910)13:14/15<2177::AID-HYP850>3.0.CO;2-V)
- Burgin AJ, Groffman PM, Lewis DN (2010) Factors regulating denitrification in a riparian wetland. *Soil Sci Soc Am J* 74:1826. doi:[10.2136/sssaj2009.0463](https://doi.org/10.2136/sssaj2009.0463)
- Campbell JL, Soggi AM, Templer PH (2014) Increased nitrogen leaching following soil freezing is due to decreased root uptake in a northern hardwood forest. *Glob Chang Biol* 20:2663–2673. doi:[10.1111/gcb.12532](https://doi.org/10.1111/gcb.12532)
- Covino TP, McGlynn BL (2007) Stream gains and losses across a mountain-to-valley transition: Impacts on watershed hydrology and stream water chemistry. *Water Resour Res* 43:W10431. doi:[10.1029/2006WR005544](https://doi.org/10.1029/2006WR005544)
- Dawson TE, Ehleringer JR (1991) Streamside trees that do not use stream water. *Nature* 350:335–337. doi:[10.1038/350335a0](https://doi.org/10.1038/350335a0)
- Durán J, Rodríguez A, Morse JL, Groffman PM (2013) Winter climate change effects on soil C and N cycles in urban grasslands. *Glob Chang Biol* 19:2826–2837. doi:[10.1111/gcb.12238](https://doi.org/10.1111/gcb.12238)
- Ehleringer JR, Arnow LA, Arnow T, et al. (1992) Red Butte Canyon Research Natural Area: history, flora, geology, climate, and ecology. *Gt Basin Nat* 52:95–121
- Fenn ME, Haeuber R, Tonnesen GS, et al. (2003) Nitrogen emissions, deposition, and monitoring in the western United States. *Bioscience* 53:391–403. doi:[10.1641/0006-3568\(2003\)053\[0391:NEDAMI\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2003)053[0391:NEDAMI]2.0.CO;2)
- Fierer N, Schimel JP (2003) A proposed mechanism for the pulse in carbon dioxide production commonly observed following the rapid rewetting of a dry soil. *Soil Sci Soc Am J* 67:798–805. doi:[10.2136/sssaj2003.7980](https://doi.org/10.2136/sssaj2003.7980)
- Fitzhugh RD, Driscoll CT, Groffman PM, et al. (2001) Effects of soil freezing disturbance on soil solution nitrogen, phosphorus, and carbon chemistry in a northern hardwood ecosystem. *Biogeochemistry* 56:215–238. doi:[10.1023/A:1013076609950](https://doi.org/10.1023/A:1013076609950)
- Frank KW, O'Reilly KM, Crum JR, Calhoun RN (2006) The fate of nitrogen applied to a mature Kentucky bluegrass turf. *Crop Sci* 46:209. doi:[10.2135/cropsci2005.04-0039](https://doi.org/10.2135/cropsci2005.04-0039)
- Groffman PM, Boulware NJ, Zipperer WC, et al. (2002) Soil nitrogen cycle processes in urban riparian zones. *Environ Sci Technol* 36:4547–4552. doi:[10.1021/es020649z](https://doi.org/10.1021/es020649z)
- Groffman PM, Driscoll CT, Fahey TJ, et al. (2001) Effects of mild winter freezing on soil nitrogen and carbon dynamics in a northern hardwood forest. *Biogeochemistry* 56:191–213. doi:[10.1023/A:1013024603959](https://doi.org/10.1023/A:1013024603959)
- Groffman PM, Hardy JP, Fashu-Kanu S, et al. (2010) Snow depth, soil freezing and nitrogen cycling in a northern hardwood forest landscape. *Biogeochemistry* 102:223–238. doi:[10.1007/s10533-010-9436-3](https://doi.org/10.1007/s10533-010-9436-3)
- Groffman PM, Hardy JP, Fisk MC, et al. (2009a) Climate variation and soil carbon and nitrogen cycling processes in a northern hardwood forest. *Ecosystems* 12:927–943. doi:[10.1007/s10021-009-9268-y](https://doi.org/10.1007/s10021-009-9268-y)
- Groffman PM, Holland EA, Myrold DD, et al. (1999) Denitrification. In: Robertson GP, Bledsoe CS, Coleman DC, Sollins P (eds) *Standard Soil Methods for Long-Term Ecological Research*. Oxford University Press, New York, pp. 272–290
- Groffman PM, Law NL, Belt KT, et al. (2004) Nitrogen fluxes and retention in urban watershed ecosystems. *Ecosystems* 7:393–403. doi:[10.1007/s10021-003-0039-x](https://doi.org/10.1007/s10021-003-0039-x)

- Groffman PM, Williams CO, Pouyat RV, et al. (2009b) Nitrate leaching and nitrous oxide flux in urban forests and grasslands. *J Environ Qual* 38:1848. doi:10.2134/jeq2008.0521
- Hale RL, Turnbull L, Earl S, et al. (2014) Sources and transport of nitrogen in arid urban watersheds. *Environ Sci Technol* 48:6211–6219. doi:10.1021/es501039t
- Hall SJ, McDowell WH, Silver WL (2013) When wet gets wetter: Decoupling of moisture, redox biogeochemistry, and greenhouse gas fluxes in a humid tropical forest soil. *Ecosystems* 16:576–589. doi:10.1007/s10021-012-9631-2
- Hall SJ, Weintraub SR, Eiriksson D, et al. (2016) Stream nitrogen inputs reflect groundwater across a snowmelt-dominated montane to urban watershed. *Environmental Science & Technology* In press. doi:10.1021/acs.est.5b04805
- Janke BD, Finlay JC, Hobbie SE, et al. (2014) Contrasting influences of stormflow and baseflow pathways on nitrogen and phosphorus export from an urban watershed. *Biogeochemistry* 121:209–228. doi:10.1007/s10533-013-9926-1
- Kaushal SS, Groffman PM, Band LE, et al. (2011) Tracking nonpoint source nitrogen pollution in human-impacted watersheds. *Environ Sci Technol* 45:8225–8232. doi:10.1021/es200779e
- Kaye JP, McCulley RL, Burke IC (2005) Carbon fluxes, nitrogen cycling, and soil microbial communities in adjacent urban, native and agricultural ecosystems. *Glob Chang Biol* 11:575–587. doi:10.1111/j.1365-2486.2005.00921.x
- MacDonald GM (2010) Water, climate change, and sustainability in the southwest. *Proc Natl Acad Sci* 107:21256–21262. doi:10.1073/pnas.0909651107
- McClain M, Boyer E, D CL, et al. (2003) Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. *Ecosystems* 6:301–312. doi:10.1007/s10021-003-0161-9
- Morse JL, Durán J, Groffman PM (2015) Soil denitrification fluxes in a northern hardwood forest: the importance of snowmelt and implications for ecosystem N budgets. *Ecosystems* 18:520–532. doi:10.1007/s10021-015-9844-2
- Moyes AB, Bowling DR (2013) Interannual variation in seasonal drivers of soil respiration in a semi-arid Rocky Mountain meadow. *Biogeochemistry* 113:683–697. doi:10.1007/s10533-012-9797-x
- Neff J, Chapin FS, Vitousek PM (2003) Breaks in the cycle: dissolved organic nitrogen in terrestrial ecosystems. *Front Ecol Environ* 1:205–211. doi:10.1890/1540-9295(2003)001[0205:BITCDO]2.0.CO;2
- Pickett STA, Cadenasso ML, Grove JM, et al. (2011) Urban ecological systems: Scientific foundations and a decade of progress. *J Environ Manag* 92:331–362. doi:10.1016/j.jenvman.2010.08.022
- Pinheiro J, Bates D, DebRoy S, et al (2014) nlme: linear and nonlinear mixed effects models.
- Raciti SM, Burgin AJ, Groffman PM, et al. (2011b) Denitrification in suburban lawn soils. *J Environ Qual* 40:1932. doi:10.2134/jeq2011.0107
- Raciti SM, Groffman PM, Fahey TJ (2008) Nitrogen retention in urban lawns and forests. *Ecol Appl* 18:1615–1626. doi:10.1890/07-1062.1
- Raciti SM, Groffman PM, Jenkins JC, et al. (2011a) Accumulation of carbon and nitrogen in residential soils with different land-use histories. *Ecosystems* 14:287–297. doi:10.1007/s10021-010-9409-3
- Raciti SM, Groffman PM, Jenkins JC, et al. (2011c) Nitrate production and availability in residential soils. *Ecol Appl* 21:2357–2366. doi:10.1890/10-2009.1
- Roach WJ, Grimm NB (2011) Denitrification mitigates N flux through the stream–floodplain complex of a desert city. *Ecol Appl* 21:2618–2636. doi:10.1890/10-1613.1
- Schaap MG, Leij FJ, van Genuchten MT (2001) Rosetta: a computer program for estimating soil hydraulic parameters with hierarchical pedotransfer functions. *J Hydrol* 251:163–176. doi:10.1016/S0022-1694(01)00466-8
- Schaetzl RJ, Anderson S (2005) *Soils: Genesis and Geomorphology*. Cambridge University Press
- Scherer-Lorenzen M, Palmberg C, Prinz A, Schulze E-D (2003) The role of plant diversity and composition for nitrate leaching in grasslands. *Ecology* 84:1539–1552. doi:10.1890/0012-9658(2003)084[1539:TROPDA]2.0.CO;2
- Simunek J, Sejna M, Saito H, et al. (2013) The HYDRUS-1D software package for simulating the movement of water, heat, and multiple solutes in variably saturated media. University of California Riverside, Riverside, CA, USA, Department of Environmental Sciences
- Six J, Conant RT, Paul EA, Paustian K (2002) Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. *Plant Soil* 241:155–176. doi:10.1023/A:1016125726789
- Solomon DK, Cerling TE (1987) The annual carbon dioxide cycle in a montane soil: observations, modeling, and implications for weathering. *Water Resour Res* 23:2257–2265. doi:10.1029/WR023i012p02257
- Wang W, Haver D, Pataki DE (2014) Nitrogen budgets of urban lawns under three different management regimes in southern California. *Biogeochemistry* 121:127–148. doi:10.1007/s10533-013-9942-1
- Zak DR, Groffman PM, Pregitzer KS, et al. (1990) The vernal dam: plant-microbe competition for nitrogen in northern hardwood forests. *Ecology* 71:651–656. doi:10.2307/1940319
- Zhu W-X, Hope D, Gries C, Grimm NB (2006) Soil characteristics and the accumulation of inorganic nitrogen in an arid urban ecosystem. *Ecosystems* 9:711–724. doi:10.1007/s10021-006-0078-1