

Received Date: 28/05/2015

Revised Date: 09/11/2015

Accepted Date: 07/12/2015

Article Type: **Original Article**

Running Head: Urban trees reduce nutrient leaching

## **Urban trees reduce nutrient leaching to groundwater**

Daniel A. Nidzgorski<sup>†</sup> and Sarah E. Hobbie

Department of Ecology, Evolution, and Behavior, University of Minnesota, Saint Paul, MN,  
55108

<sup>†</sup> e-mail address: [dnidz@civiceco.org](mailto:dnidz@civiceco.org); [orcid.org/0000-0003-4454-3042](https://orcid.org/0000-0003-4454-3042)

### **Abstract**

Many urban waterways suffer from excess nitrogen (N) and phosphorus (P) feeding algal blooms, which cause lower water clarity and oxygen levels, bad odor and taste, and the loss of desirable species. Nutrient movement from land to water is likely to be influenced by urban vegetation, but there are few empirical studies addressing this. In this study, we examined

This article has been accepted for publication and undergone full peer review but has not been through the copyediting, typesetting, pagination and proofreading process, which may lead to differences between this version and the Version of Record. Please cite this article as doi: 10.1002/15-0976

This article is protected by copyright. All rights reserved.

whether or not urban trees can reduce nutrient leaching to groundwater, an important nutrient export pathway that has received less attention than stormwater. We characterized leaching beneath thirty-three trees of fourteen species, and seven open turfgrass areas, across three city parks in Saint Paul, Minnesota. We installed lysimeters at 60 cm depth to collect soil water approximately biweekly from July 2011 through October 2013, except during winter and drought periods, measured dissolved organic carbon (C), N, and P in soil water, and modeled water fluxes using the BROOK90 hydrologic model. We also measured soil nutrient pools (bulk C and N, KCl-extractable inorganic N, Brays-P), tree tissue nutrient concentrations (C, N, and P of green leaves, leaf litter, and roots), and canopy size parameters (leaf biomass, leaf area index) to explore correlations with nutrient leaching. Trees had similar or lower N leaching than turfgrass in 2012 but higher N leaching in 2013; trees reduced P leaching compared with turfgrass in both 2012 and 2013, with lower leaching under deciduous than evergreen trees. Scaling up our measurements to an urban subwatershed of the Mississippi River (~17,400 ha, containing roughly 1.5 million trees), we estimated that trees reduced P leaching to groundwater by 533 kg in 2012 (0.031 kg/ha or 3.1 kg/km<sup>2</sup>) and 1201 kg in 2013 (0.069 kg/ha or 6.9 kg/km<sup>2</sup>). Removing these same amounts of P using stormwater infrastructure would cost \$2.2 million and \$5.0 million per year (2012 and 2013 removal amounts, respectively).

*Keywords: groundwater; nitrogen; nutrient leaching; nutrient pollution; phosphorus; plant traits; urban ecosystems; urban trees*

## Introduction

Urban trees enhance human well-being in many ways, from improving air quality (Nowak et al. 2006b) to reducing crime rates (Kuo and Sullivan 2001), but less is understood about how urban trees affect the water quality of local lakes and streams. Because trees' physiological traits contrast sharply with those of the turfgrasses that dominate residential landscapes, they are likely to alter nutrient cycling and the movement of nitrogen (N) and phosphorus (P) from land to water. Yet there has been little research on how trees or other urban vegetation affect nutrient transport to stormwater and groundwater, which in turn flow to local lakes and streams.

Although most urban water-quality efforts focus on stormwater, urban groundwater has been shown to contain concentrations of N and P high enough to contribute to water-quality problems (Trojan et al. 2003, Janke et al. 2013). Excess nutrients drive algal blooms that cause lower water clarity and oxygen levels, bad odor and taste, and the loss of desirable species. Urban trees and other "green infrastructure" may be able to reduce nutrient pollution and play an important role in improving the water quality and ecosystem service provisioning of local lakes and streams.

Many parts of American cities and towns outside dense downtown cores, such as residential neighborhoods, could be described as "urban savannas" with isolated, open-grown trees and a grassy understory. As in natural savannas, trees in urban savannas create altered microclimates with cooler soil and ground-surface temperatures (Peters and McFadden 2010) and lower wind velocities (Dewalle and Heisler 2012); their effects on air temperature, relative humidity, and soil moisture are less clear (Miller 1997). Trees differ from turfgrasses in ways that are also likely to affect nutrient cycling; for example, they have deeper roots with greater lateral spread, strong seasonal patterns of litterfall, and woodier tissues. Differences in nutrient cycling between trees and grasses are well documented in tropical savannas, where trees create "fertile islands"

with higher soil carbon (C), N, and P in the sub-canopy soil than in open grassy areas, even in tropical savannas where the trees are not symbiotic N-fixers (see reviews in Vetaas 1992, Rhoades 1997).

In temperate savannas and grasslands, however, this pattern is less clear. Nutrient cycling differs under trees and grasses, but the magnitude and even the direction of trees' influence vary. In some sites, for example, trees increased N mineralization (Dijkstra et al. 2006) and soil organic matter (Gill and Burke 1999), but elsewhere trees decreased them (Jackson et al. 2002, McCarron et al. 2003). Thus, while results from natural systems support the idea that urban trees are likely to be important drivers of nutrient cycling, they offer no clear consensus on the magnitude or direction of these effects.

A few studies have investigated tree effects on nutrient leaching in urban systems, also with contradictory results. For example, in newly planted experimental plots, a mix of woody and herbaceous ornamental species had higher N and P leaching fluxes than turfgrass plots, even though turfgrass plots were more heavily fertilized (Loper et al. 2013). The authors caution that these results might change once the ornamental species are fully established, however, since their data only span the first year after planting. Another set of experimental plots, where sampling began 1.5 years after planting, found opposite results (Qin et al. 2013): plots with 10% tree cover had higher N leaching than those with 25% or 40% tree + shrub cover, and P leaching was similar across all plots, even though trees and shrubs were fertilized more heavily than turfgrass. A study that compared nitrate leaching below several different categories of established landscaping at a horticultural garden in Rhode Island (Amador et al. 2007) did not find differences in total  $\text{NO}_3\text{-N}$  fluxes among turfgrass, evergreen trees, and deciduous trees, even though turfgrass areas were fertilized and trees were not. Sample sizes were limited, however,

and the trees were mulched annually with pine bark, which may have obscured species effects on N cycling and leaching. Other studies comparing nutrient leaching under trees (and/or shrubs) vs. turfgrass found higher nutrient leaching under vegetation types that received more fertilizer (e.g. Erickson et al. 2005, 2008, Groffman et al. 2009), so vegetation effects could not be separated from fertilization effects in these studies.

In non-urban forests, nutrient leaching differs among tree species and is related to plant physiological traits. Nitrate leaching was inversely related to forest floor (Oe + Oa layers) C:N ratio in European coniferous forests (Gundersen et al. 1998) and Northeastern USA hardwood forests (Lovett et al. 2002). If similar differences occur among urban tree species, changes in species composition could also reduce nutrient leaching and improve water quality.

Our specific objectives were to: 1) compare nutrient leaching between trees and turfgrass, 2) compare nutrient leaching among tree species, and relate those differences to plant traits, and 3) scale up our leaching estimates to estimate the watershed-scale effects of trees on nutrient leaching. For our first objective, we hypothesized that trees would have lower leaching of N and P to groundwater than turfgrass. We expected that trees take up more N and P than turfgrass, and also provide soil microbes with more organic carbon than turfgrass, thereby increasing nutrient immobilization rates. Unlike prairie grasses that have very deep root systems, turfgrasses are more shallow-rooted and do not build deep organic-rich soils (Jackson et al. 1996).

For our second objective, we hypothesized that tree root and leaf-litter nutrient concentrations would be positively correlated with nutrient leaching. We expected that tree species that produce litter with lower nutrient concentrations would promote N and P immobilization rates, thus decreasing the amount of nutrients in readily leachable forms.

For our third objective, we scaled up our results to the Capitol Region Watershed, an area of approximately 17,400 ha draining into the Mississippi River and encompassing portions of the cities of Falcon Heights, Lauderdale, Maplewood, Roseville and Saint Paul, Minnesota. We expected that even small differences among vegetation types in N or P leaching per m<sup>2</sup> could result in watershed-scale effects large enough to substantially alter water quality.

## Methods

We measured nutrient leaching beneath urban trees and open turfgrass areas in city parks in Saint Paul, Minnesota, USA. We used tension lysimeters to obtain repeated samples of soil water to measure nutrient concentrations, and combined these data with modeled estimates of vertical soil water flux to calculate nutrient leaching fluxes. We also measured soil nutrient pools (bulk C and N, KCl-extractable inorganic N, Brays-P), tree tissue nutrient concentrations (C, N, and P of green leaves, leaf litter, and roots), and tree size parameters (leaf biomass, leaf area index) to explore relationships with nutrient leaching.

### *Study sites and sampling locations*

Our study sites were three city parks (Carty, College, and Horton; ~1.6 ha each) in residential neighborhoods of western Saint Paul, Minnesota, USA, roughly in a NW/SE line spanning 6 km. Saint Paul has a humid continental climate, with average annual precipitation of 780 mm (140 mm as snow) and average daily high and low temperatures ranging from -4.6 and -13.6°C in January to 28.6 and 17.8°C in July (1981-2010 averages for the Minneapolis-Saint Paul

International Airport weather station, compiled by NOAA). Groundwater is fairly shallow, generally <10 m below the surface (US Geologic Survey).

All three parks receive identical minimal management: they are not fertilized nor irrigated, and leaf litter and grass clippings are mulched into the grass with a mulching mower. Soils, derived from glacial till, are sandy to silty loams overlying coarse sand (USDA Web Soil Survey); the sandy subsoils are high in apatite, a mineral P source (Larson 1985). During lysimeter installation, we observed that the depth to the coarse sand varied considerably within parks, generally between 30 and 75 cm.

Across these parks, we selected forty sampling sites, 33 under trees (23 deciduous, 10 evergreen) and 7 in open turfgrass areas (at least one tree-height away from any tree). Focal trees were healthy, mature trees in relatively flat areas, without impervious surfaces or other management activities (e.g. mulch, non-turfgrass plantings) under the canopy. Fourteen focal tree species were selected to represent common deciduous and evergreen urban tree species with a range of leaf N and P concentrations: *Abies concolor* (Gord. & Glend.) Lindl. Ex Hildebr. (white/concolor fir, n=2), *Acer platanoides* L. (Norway maple, n=4), *A. saccharinum* L. (silver maple, n=2), *Celtis occidentalis* L. (hackberry, n=5), *Fraxinus pennsylvanica* Marshall (green ash, n=5), *Ginkgo biloba* L. (ginkgo, a deciduous broad-leaved gymnosperm, n=1), *Gleditsia triacanthos* L. (honeylocust, a symbiotic N-fixer, n=1), *Pinus resinosa* Alton (red pine, n=1), *P. strobus* L. (Eastern white pine, n=2), *P. sylvestris* L. (Scots pine, n=3), *P. ponderosa* Lawson & C. Lawson (Ponderosa pine, n=2), *Quercus macrocarpa* Michx. (bur oak, n=3), *Tilia cordata* Mill. (little-leaf linden, n=2). Two of our focal *F. pennsylvanica* trees were cut down by Saint Paul Parks & Recreation over the 2011-2012 winter; we continued to sample the lysimeters from these sites but excluded the data from deciduous averages, trait regressions, and statistical tests.

## *Lysimeters*

At each focal tree or open turfgrass area, we installed one tension lysimeter at 60 cm depth, below 80-90% of tree roots (Crow 2005), during early summer 2011 (with permission from the City of Saint Paul). For trees, we located our lysimeters along a random compass bearing from the focal tree's trunk, halfway between the trunk and the edge of the tree canopy (dripline). We excluded locations that were within one tree-height of a neighboring heterospecific tree unless it was on the opposite side of the focal tree. For turfgrass areas, we started at a location near the center of the open area and then placed the lysimeter 2 m away along a random compass bearing.

Lysimeters were constructed of a porous ceramic cup (SoilMoisture Equipment Corp, Santa Barbara, CA) attached to a 30 cm segment of PVC pipe, and capped with a rubber stopper. Flexible, inert plastic tubing led to the surface for sampling. Lysimeters were flushed with 10% HCl and several rinses of nanopure water after assembly, and after installation we collected and discarded 2-3 samples of soil water to further flush and equilibrate them. To install lysimeters, we augured an 8 cm-diameter vertical borehole, seated the lysimeter in silica flour, backfilled the borehole with clean sand, and sealed the top of the borehole with a layer of bentonite clay to prevent preferential water flow down the borehole.

We sampled each lysimeter approximately biweekly during the growing season (thaw to snow cover) unless the soil was too dry to yield a sample, from July 2011 through October 2013. To sample a lysimeter, we pumped a -50 kPa vacuum, sealed the lysimeter, and returned 1-2 d later to pump out the water it had collected. Samples were transported back to lab and frozen until nutrient analyses. Additional filtration was not required because the soil water had already passed through the porous ceramic cup of the lysimeter. We analyzed lysimeter samples for total dissolved N using a Shimadzu TOC Vc<sub>pn</sub> analyzer (Shimadzu Scientific Instruments, Columbia,



MD), total dissolved P using a persulfate digest followed by molybdate-blue colorimetry (DeMott et al. 1998),  $\text{NO}_3^-$ -N using vanadium-oxidation colorimetry (Doane and Horwath 2003), and soluble reactive P (SRP) by molybdate-blue chemistry.  $\text{NH}_4^+$ -N concentrations in soil water, measured using salicylate colorimetry (Willis et al. 1993), were too low to detect ( $< 0.1$  mg/L).

### *Soil and tree measurements*

During lysimeter installation, we collected soil samples from 0-10 cm, 10-20 cm, 20-40 cm, and 40-60 cm depth horizons, and also measured the borehole volume (to calculate bulk density) at each depth by lining the borehole with a thin plastic bag and measuring the amount of water needed to fill it. We measured total soil C and N using a Variomax CN combustion analyzer (Elementar Analysensysteme, GmbH, Haunau, Germany). For measuring extractable nutrient pools, we collected additional 0-10 cm soil samples (2.5 cm diameter) in July 2011, one halfway between each tree's trunk and dripline along each cardinal direction (four per tree), or at four points 2 m from each turfgrass lysimeter. We composited these soil samples to yield one sample per focal tree or turfgrass area (40 samples total); we extracted a 10 g subsample with 2M KCl to measure  $\text{NO}_3^-$  and  $\text{NH}_4^+$ , and incubated a second 10 g subsample in the dark at room temperature for 10 d. We then extracted the incubated subsample in the same manner, and calculated potential net N mineralization and nitrification as the difference between final and initial extractable inorganic N or  $\text{NO}_3^-$ -N, respectively. We also extracted a third 5 g subsample for Brays-P (Bray and Kurtz 1945).

We measured each tree's height, trunk diameter, canopy width, and canopy fullness in summer 2011 and used the urban-specific software iTree Eco v5.0.8 ([www.itreetools.org](http://www.itreetools.org)) to estimate growth rate (annual increase in kg C stored in tree wood), leaf biomass, canopy area,

and leaf area index (LAI). We measured C, N, and P contents of fully expanded sun leaves (collected July 2011), fresh leaf litter (fall 2011), and fine, live roots (<2 mm diameter, collected from 0-20 cm depth during lysimeter installation). One sample of each tissue type was analyzed for each focal tree. We measured leaf %C and %N as for soil C and N, and litter and root %C and %N using a Costech ECS4010 element analyzer (Costech Analytical, Valencia, California, USA). Phosphorus concentrations of all tissue samples were measured by ashing the sample (300°C for 30 min followed by 550°C for 2 h) and extracting with 10N sulfuric acid, followed by molybdate-blue colorimetry (DeMott et al. 1998).

### *Hydrologic modeling*

We modeled daily vertical water fluxes at 60 cm depth using the BROOK90 hydrologic model (Federer et al. 2003). For model inputs, we obtained daily precipitation and temperature data from the University of Minnesota climate station (located within 6 km of all study sites), as well as wind speed, dewpoint, and solar radiation data from the Saint Paul downtown airport (located within 12 km of all study sites). We modeled 2007-2010 as an initialization and calibration period for the model prior to our output period of 2011-2013. Using a turfgrass LAI of 1.5 (after Milesi et al. 2005) and default model parameters yielded good predictions for turfgrass evapotranspiration (ET) in 2008 when compared with eddy-flux measurements from a nearby study (Peters et al. 2011). For trees, we reduced the maximum leaf conductance parameter so modeled tree transpiration in 2008 matched values calculated from 2008 sapflow measurements on similar open-grown urban trees in parklike conditions, within 10 km of our study sites (Peters et al. 2010).

Because LAI exerts strong controls on ET, and hence soil water fluxes, we modeled each focal tree separately using that tree's LAI and height, with BROOK90 set to calculate ET from a sparse canopy. We modeled all evergreen trees without a turfgrass understory, even though some of our focal evergreens had some grass beneath the canopy. All of our focal deciduous trees had a turfgrass understory. To model this in BROOK90, which cannot explicitly model two separate canopy layers, we increased root density to the sum of the model's default deciduous and turfgrass root densities, added the turfgrass LAI (1.5, constant across the whole year) to the tree's LAI (which increased from zero to maximum in April, and decreased from maximum to zero in November), and reduced the light-extinction parameter to simulate partially shaded turfgrass.

### *Flux calculations*

We created a daily timeseries of estimated soil water N and P concentrations by interpolating linearly between soil water samples for each lysimeter. We interpolated across any missing values within a calendar year's sampling period, but we did not extrapolate beyond the first and last sample collected for that tree or turfgrass area. We multiplied each day's concentration estimates (in mg/L or  $\mu\text{g/L}$ ) by modeled soil water fluxes (in mm/d, equivalent to  $\text{L m}^{-2} \text{d}^{-1}$ ) to yield vertical soil water fluxes ( $\text{g m}^{-2} \text{d}^{-1}$ ) for all nutrients.

### *Sensitivity analysis*

We analyzed the sensitivity of our flux estimates to uncertainties in key parameters of the hydrologic model: LAI, transpiration, and soil texture. Unlike uncertainties in measuring soil water nutrient concentrations, which would affect flux estimates proportionally, threshold behaviors in the hydrologic model can potentially amplify or dampen changes in parameters. We first modeled soil water fluxes for an average deciduous tree (with grassy understory) using the

mean LAI and height of all deciduous trees in our study (LAI = 4.1 and height = 13.7 m). All other parameters were the same as in our primary modeling. From this baseline, we next modeled separate changes in each parameter: we changed LAI to 90% and 110% of baseline; we adjusted the maximum leaf conductance parameter to achieve 95%, 105%, and 110% of baseline total transpiration (we were unable to force transpiration as low as 90% of baseline); and we changed soil texture one class coarser or finer, from loamy sand to sand or sandy loam. Finally, we modeled two interaction scenarios to explore upper and lower bounds, simultaneously adjusting LAI, transpiration, and soil texture to first increase and then decrease predicted soil water fluxes.

We created a timeseries of soil water total N and P concentrations using the mean of all deciduous trees on each sampling date, employing the same procedure as for individual lysimeters. Using this timeseries, we calculated leaching fluxes for each model run in the sensitivity analysis, and the percent change in total N and P fluxes for 2012 and 2013 compared to the baseline model run.

### *Statistical analyses*

For lysimeter concentration and flux data, we tested differences among vegetation types using a repeated-measures ANOVA with Type II sums of squares. Our unit of observation was individual trees, rather than species means. We calculated pairwise contrasts among turfgrass, evergreens, and deciduous trees evaluated with the Holm procedure (overall  $\alpha=0.05$ ).

To test for relationships between soil water nutrient concentrations and tree traits, we used quantile regression (Koenker and Hallock 2001), which can handle non-constant variance and outliers better than ordinary least-squares regression, and also explores relationships at other

parts of the distribution besides the mean (Cade and Noon 2003). We calculated quantile regressions at quantiles ( $\tau$ ) from 0.01 to 0.99 in increments of 0.01. At each  $\tau$ , we also calculated 90% confidence intervals for slope and intercept estimates using an inverted rankscore test (Koenker 1994) to determine statistical significance. We present the results of quantile regression graphically (in Appendix A) as stair-step plots of a slope or intercept coefficient, plus an envelope of its confidence intervals, both at  $\tau$  increments of 0.01. Selected statistically significant regression lines were also plotted on the data scatterplot.

We focused on identifying trait relationships that explained not only a central tendency in the lysimeter data (quantiles around the median) but also in the upper quantiles (i.e. trees with the highest nutrient leaching to groundwater) which explain large differences among trees with the highest leaching. Relationships at low quantiles, by contrast, explain relatively small differences among trees with low leaching and are less useful for guiding management decisions.

We performed all statistical analyses in R 3.0.2 (R Core Team, Vienna, Austria), using base packages plus ANOVA functions from ez 4.2-2 and quantile-regression functions from quantreg 5.05.

### *Scaling up*

We scaled up our results to the Capitol Region Watershed (~17,400 ha), which has a land cover of 33% tree canopy cover and 22% low-statured vegetation (Kilberg and Bauer 2011). We treated all low-statured vegetation as open turfgrass, and estimated the proportion of deciduous and evergreen trees using data from a survey spanning public and private lands in Minneapolis (immediately west of the Capitol Region Watershed), which found 89.9% deciduous trees and 10.1% evergreen trees by canopy area (Nowak et al. 2006a). Because some tree canopy

overhangs impervious surfaces, which do not contribute to leaching, we used QuantumGIS 1.8 (QGIS Development Team, Open Source Geospatial Foundation) to clip the tree canopy maps to exclude areas overhanging buildings, streets, and alleyways mapped by the Capitol Region Watershed District (*unpublished data*). Sidewalks and other smaller impervious surfaces were not mapped, so we slightly overestimated the area of tree canopy over pervious surfaces. The land-cover classification by Kilberg and Bauer did not distinguish between deciduous and evergreen trees; we assumed that all tree canopy cover over streets, alleyways, or buildings was deciduous trees because evergreens are only planted as boulevard trees in wide boulevards and parkways where they typically do not overhang the street (Saint Paul Department of Parks and Recreation 2013), and they generally have pyramidal forms rather than high spreading canopies that overhang buildings.

In scaling up our results, we assumed that our lysimeters sampled an area representing average soil water nutrient concentrations under a tree's canopy. We averaged together all deciduous or all evergreen trees in our study; if future work is better able to resolve species differences, it will be possible to weight the averaging by each species' proportion of total canopy area in the watershed (Nowak et al. 2006a).

We estimated the ecosystem service value of trees reducing P leaching to groundwater by using the cost to remove the same amount of P with stormwater infrastructure, as measured in our study watershed. Annual costs per kg P removed were \$1,570 for a regional stormwater pond, \$4,200 for infiltration trenches, and \$6,140 for rain gardens (Capitol Region Watershed District 2010). We used the cost/kg for infiltration trenches because infiltration trenches were the

most cost-effective option that, like trees, could theoretically be used throughout the entire watershed. Regional-scale ponds, while more cost-effective, could only serve a small portion of the watershed and are less useful as a comparison.

## Results

### *Soil water nutrient concentrations*

Below-normal summer and fall precipitation (Appendix A; Figure A1) in all sampling years frequently left the soil too dry to obtain soil water samples, starting in mid- to late summer and persisting through the following early spring. Successful sampling dates spanned 7/28-9/1/2011, 4/26-7/13/2012, and 5/16-8/2/2013. We were not able to obtain water from every lysimeter on every sampling date, especially towards the beginning and end of sampling periods each year.

For both N and P, deciduous trees had lower soil water nutrient concentrations than open turfgrass areas (Table 1, Figure 1). By contrast, evergreen trees had lower soil water P concentrations than turfgrass, but similar N concentrations. N concentrations showed a seasonal pattern, with high concentrations in spring 2012 and 2013 declining rapidly, but P showed no clear seasonal pattern. There were significant differences among vegetation types: For N, deciduous trees < evergreens  $\approx$  turfgrass; and for P, deciduous trees < evergreens < turfgrass (Table 1). There was also noticeable interannual variability.

N and P concentrations in soil water samples were essentially uncorrelated and independent (Appendix A; Figure A2). On average, 60% of both total N and P were present in inorganic forms ( $\text{NO}_3^-$  and SRP). Samples with higher total N or P also had a statistically discernable

tendency to have a higher percent as inorganic N or P, respectively (Appendix A; Figure A3).

88% of soil water samples with N concentration over 2 mg/L and 86% of soil water samples with P concentrations over 100  $\mu\text{g/L}$  were more than 50% inorganic. The percent of N as  $\text{NO}_3^-$  showed a highly bimodal distribution; 21% of the lysimeter samples had less than 1% of the N as  $\text{NO}_3^-$ , and 19% had more than 99% of the N as  $\text{NO}_3^-$ . In contrast, the percent P as SRP showed a relatively even distribution.

The two sites where *F. pennsylvanica* trees had been cut down had high pulses of soil water N in spring 2012, following removal (data not shown). In spring 2013, their soil water N concentrations were higher than average for deciduous trees but comparable to a neighboring still-living *F. pennsylvanica* tree. These sites did not show a similar pulse of P after removal, and their soil water P concentrations throughout 2012 and 2013 were similar to their neighboring live conspecific.

Our analyses excluded N data from under one evergreen (*A. concolor*) that had suspiciously high N concentrations in its soil water samples, possibly indicating contamination of the lysimeter or the soil. P concentrations from under this tree were within the range of other samples, so we kept those data in our analyses.

### *Water and nutrient fluxes*

Modeled estimates of daily vertical water fluxes at 60 cm showed episodic pulses with dampened responses to rainstorms that tailed off more gradually than precipitation (Appendix A; Figure A4). During periods when BROOK90 predicted that the upper soil layers were drier than deeper soil layers to the extent that the upwards matric potential gradient was stronger than gravity, it predicted a small upwards (negative) soil water flux. We included these negative



values when calculating average or total nutrient fluxes, as we believe that they represent a real upwards wicking of soil water and nutrients

Overall, BROOK90 predicted lower vertical water fluxes under turfgrass than under most trees, driven by the trees having lower ET than turfgrass, with the exception of two high-LAI evergreens (both *A. concolor*). Because trees had lower soil water nutrient concentrations than turfgrass, while water fluxes showed an opposing pattern of higher water fluxes under trees, our estimated differences among functional types' nutrient fluxes were smaller than the differences among their soil water nutrient concentrations.

Total (2011-2013) N leaching (Table 2) was statistically similar between deciduous trees and turfgrass (though still suggestive of lower leaching under deciduous trees) despite statistically discernable differences in soil water N concentrations. Total N leaching was discernably higher for evergreens, driven by the predicted higher water flux under evergreens than turfgrass.

Seasonal patterns of N leaching, and patterns among vegetation types, were also substantially different among years (Table 2, Figure 2), largely driven by different patterns of soil water N concentration. Total P leaching was discernably different among the three vegetation types: deciduous trees < evergreens < turfgrass (Table 2, Figure 2).

### *Sensitivity analysis*

In our sensitivity analysis of key hydrologic parameters, the percent changes in total N and P leaching fluxes were substantially smaller than the shifts in model parameters (Appendix A, Table A1). Increasing LAI and transpiration decreased total N and P fluxes (and vice versa); changing LAI by 10% changed fluxes by 1-2%, and changing total transpiration by 10% changed fluxes by 2-3%. Changing soil texture one class coarser or finer both reduced fluxes by 1-4%,

with the sandy loam causing larger reductions. For the interaction scenarios, setting both LAI and transpiration = 110% and soil to sandy loam decreased total N and P fluxes by 5-10%.

Setting LAI = 90% and transpiration = 95%, with soil unchanged at loamy sand, increased total fluxes by 2-3%.

### *Tree trait relationships*

We tested relationships between soil water N and P concentrations and tree traits singly and in combination: leaf, litter, and root chemistry, growth rate, leaf biomass, projected canopy area, and LAI. Although we had also planned to test trait relationships with total annual nutrient fluxes, our estimates of individual trees' fluxes span different lengths of time because individual lysimeters often did not yield water at the beginning or end of each year's sampling period, making this impossible.

We divided our sampling dates into “spring” and “summer” seasons because of the strong seasonality in soil water N concentrations. We classified the first two sampling dates in both 2012 and 2013, with high and sharply declining N concentrations, as “spring” seasons and all others as “summer.” We first averaged together samples within a season (i.e. summer 2011, spring 2012, summer 2012, spring 2013, summer 2013) because concentrations were more similar within a season than among seasons. We then averaged together the two spring season means to create an overall mean spring value, the three summer means to create an overall mean summer value, and all five seasonal means to produce a grand mean. We tested for relationships with tree traits using both seasonal means as well as the grand mean for response variables, testing evergreen and deciduous trees separately. Using seasonal means rather than individual sampling data in our regressions helped limit the effects of missing data (cases where a lysimeter

did not yield water on that sampling date, or yielded too little water for all chemical analyses), and reduced the uncertainty associated with individual sampling events,

For N, we found different relationships during spring and summer. Springtime soil water total N concentrations among deciduous trees showed a negative relationship with root %N (Figure 3). It should be noted, however, that the five trees with high root %N (>1.5%) that drove this pattern were all *C. occidentalis*, and without this species the relationship no longer held. Among evergreens, springtime soil water N concentrations showed positive relationships with any of a well-correlated set of predictors: leaf biomass (iTree-estimated), canopy N mass (green leaf %N x leaf biomass), or litter N mass (litter %N x leaf biomass). Variation among trees' leaf or litter %N was much smaller than variation among their leaf biomass, so variation in leaf biomass accounted for most of the variation in all three predictors. Summertime soil water N concentrations did not show statistically discernable relationships with tree traits, though they were suggestive of similar relationships as for springtime concentrations. We excluded the *G. triacanthos* N-fixer from all regressions with soil water N concentrations (but did not exclude it when averaging deciduous trees' concentrations or leaching fluxes) because it was an outlier with higher soil water N concentrations than any other tree in our study (Figure 3a).

For P, we found similar relationships in both spring and summer, so we present trait relationships using the grand mean. Both deciduous and evergreen trees showed a statistically discernable relationship between soil water total P concentrations and litter C:P ratio, though in opposite directions: positive for deciduous trees, and negative for evergreen trees (Figure 4). In both cases, the relationship was stronger (steeper slope) at higher quantiles. Litter %P and litter C:P ratio were tightly correlated for both evergreen and deciduous trees, and quantile regressions using either predictor gave very similar results. We chose to present litter C:P because it better

explains the two deciduous trees with highest soil water P concentrations.

Neither xylem anatomy (gymnosperm, ring-porous, diffuse-porous), mycorrhizal type (arbuscular mycorrhizae, ectomycorrhizae), nor LAI appeared to explain any of the residual variability in our trait relationships for N or P concentrations.

### *Soil nutrients*

Average soil bulk %C and %N declined with depth, from 3.8 %C and 0.30 %N in 0-10 cm to 1.1 %C and 0.06 %N in 40-60 cm. In the 0-10 cm samples for extractable nutrients, KCl-extractable inorganic N was 0.032 mg N/g soil, the average 10 d net mineralization rate was 0.024 mg N/g soil, and the average Brays-P was 0.0094 mg P/g soil. We found no discernable differences among vegetation types for soil nutrient pools or net mineralization or nitrification rates. (We had originally planned to repeat Brays-P, extractable-N, and net-mineralization and nitrification measurements during each spring, summer, and fall of our sampling period, but the drought prevented this.)

There was a positive relationship at central and upper quantiles between net mineralization rate and springtime soil water N concentration (Appendix A; Figure A7), and a weaker positive relationship with summertime soil water N concentration. Net nitrification showed a weaker positive relationship with springtime soil water N concentration at central and upper quantiles, and no discernable relationship with summertime soil water N concentration. For P, there was a positive relationship across nearly all quantiles between Brays-P and soil water P concentrations (Appendix A; Figure A8). Including tree traits and soil nutrients together in quantile regressions did not improve their predictive power.

## *Scaling up*

Of the 5,740 ha of tree canopy cover in the Capitol Region Watershed (Kilberg and Bauer 2011), we estimated that 4,870 ha (74.8%) is deciduous canopy over pervious surfaces, and 579 ha (10.1%) is evergreen canopy over pervious surfaces. Multiplying by our average fluxes for each vegetation type, we estimated that trees reduced P leaching to groundwater by 533 kg (\$2.24 million worth; 0.031 kg/ha) in 2012, and 1201 kg (\$5.04 million worth; 0.069 kg/ha) in 2013. We estimated that a typical mature deciduous tree (100 m<sup>2</sup> projected canopy area) reduced P leaching to groundwater by 1.1 g (\$5 worth) in 2012 and 2.3 g (\$10 worth) in 2013 relative to turfgrass, and a typical mature evergreen (50 m<sup>2</sup> projected canopy area) reduced P leaching to groundwater by 0.1 g (\$0.40 worth) in 2012 and 0.7 g (\$3 worth) in 2013.

## **Discussion**

### *Nutrient leaching to groundwater*

N and P concentrations in soil water were high enough to potentially degrade water quality: 40% of lysimeter samples exceeded 50 µg P/L, which is the average of deep- and shallow-lake eutrophication standards for the ecoregion (MN Administrative Rules, Ch. 7050), and 17% exceeded the 10 mg NO<sub>3</sub>-N/L standard for drinking water (MN Administrative Rules, Ch. 4717). Although there are not formal eutrophication standards for total N in Minnesota surface waters, soil water samples often had higher total N concentrations than typically would be found in agricultural streams (national median = 4 mg/L, Dubrovsky and Hamilton 2010).

In our study system, we do not expect substantial removal or retention of N and P below our

lysimeters at 60 cm. The combination of coarse sandy subsoils, P-rich mineralogy, and shallow water table (<10 m) suggest that the bulk of the soil water N and P we measured at 60 cm will reach groundwater. Once in groundwater, we expect that little removal or retention occurs before reaching surface waters, as shallow groundwater in our study area connects readily to local lakes and the Mississippi River. These expectations are supported by data showing elevated N and P concentrations in groundwater-dominated springs, curtain drains, and storm-drain baseflow: mean = 3.03 mg N/L and 22.13 µg P/L (J. C. Finlay and B. Janke, *unpublished data*). These values are on the low end of our lysimeter soil water samples, indicating that some removal or retention has likely occurred (and/or possibly dilution; groundwater movement in the watershed is not yet well understood), but that substantial amounts of leached N and P reach groundwater and surface waters.

The substantial percentage of soil water P that was in inorganic forms (SRP) was unexpected. Because SRP is fairly immobile in many soils, we had speculated that high concentrations of P in soil water might exist largely as dissolved organic P, in molecules that do not adsorb strongly onto soil mineral surfaces. While there may have been some mineralization of organic P in the lysimeters, our findings do not support a conceptual model where P leaches primarily in organic forms.

### *Watershed context*

Nutrient leaching to groundwater may be a substantial pathway of N and P transport in the Capitol Region Watershed. To generate a first-order estimate, we multiplied our leaching fluxes by the total watershed area of turfgrass and deciduous or evergreen tree canopy over pervious surfaces. We estimated that nutrient leaching contributed 5.1 kg/ha N and 0.063 kg/ha P in 2012, and 9.9 kg/ha N and 0.15 kg/ha P in 2013. For comparison, input rates for our study area are

estimated to be  $14.5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  and  $0.46 \text{ kg P ha}^{-1} \text{ yr}^{-1}$  from atmospheric deposition, and  $0.14 \text{ kg P ha}^{-1} \text{ yr}^{-1}$  from weathering (Hobbie et al., *in prep.*). We estimate that leaching was equivalent to 35% of N inputs and 10% of P inputs from deposition and weathering in 2012, and 68% of N inputs and 24% of P inputs in 2013. These comparisons do not reflect the proportion of total inputs that leach to groundwater, however, since our study sites likely received additional inputs from N fixation and pet waste.

To compare leaching with stormwater nutrient loading, we used data from Janke and coauthors (2013): stormflow in different subwatersheds of the Capitol Region Watershed contributed 1.6-5.0 kg/ha total N (dissolved + particulate) and 0.20-0.80 kg/ha total P (dissolved + particulate) during the warm season (May 1 – Oct. 31; averages of data from 2006-2011). Our data suggest that leaching to groundwater from pervious surfaces was greater than N loading from stormwater in the watershed, and >10% as much as annual P loading from stormwater. Janke and coauthors also found that baseflow in storm drains, which comes largely from lake overflow and shallow groundwater leaking into the drains, had similar N and P fluxes as our estimates of leaching.

### *Comparisons with other sites*

Estimates of annual N leaching from sites in other regions with similar vegetation spanned a broad range of values; our estimates (2012 and 2013 fluxes; Table 2) are generally on the high end of this range. Our leaching estimates under unfertilized turfgrass ( $1.4\text{-}1.5 \text{ g N m}^{-2} \text{ yr}^{-1}$ ) are comparable to those from Baltimore, MD ( $1\text{-}2 \text{ g N m}^{-2} \text{ yr}^{-1}$ ; Groffman et al. 2009), but substantially higher than two studies in southern New England ( $0.1\text{-}0.2 \text{ g N m}^{-2} \text{ yr}^{-1}$ ; Gold et al. 1990, Guillard and Kopp 2004). Studies of fertilized turfgrass also spanned a similarly broad

range (0.1-2.6 g N m<sup>-2</sup> yr<sup>-1</sup>; Gold et al. 1990, Guillard and Kopp 2004, Amador et al. 2007, Erickson et al. 2008, Groffman et al. 2009), with no clear relationship between fertilization and leaching rates. Our leaching estimates under trees varied between years (evergreens: 1.5-2.6 g N m<sup>-2</sup> yr<sup>-1</sup>; deciduous: 0.5-2.1), and were on the high end of the range reported for European forests with similar N-deposition rates (0-2 g N m<sup>-2</sup> yr<sup>-1</sup>; Gundersen et al. 1998), forests in the United States (0.05-0.5 g N m<sup>-2</sup> yr<sup>-1</sup>; Iseman et al. 1999, Strahm et al. 2005) and landscaping deciduous and evergreen trees (0.7 g N m<sup>-2</sup> yr<sup>-1</sup>; Amador et al. 2007). No other studies, to our knowledge, have measured leaching under trees with turfgrass understories. Note that, except for the study by Strahm and coauthors (2005), all these studies measured only nitrate and likely underestimated total N leaching, as nitrate was 88% of the total N leached in our study.

We found only two estimates of P leaching under similar vegetation (both from Florida), both fertilized with 15-50 kg P ha<sup>-1</sup> yr<sup>-1</sup>. Plots of 10-40% tree + shrub cover, and the remainder turfgrass, had annual leaching comparable to our results (0.01-0.03 g P m<sup>-2</sup> yr<sup>-1</sup>; Qin et al. 2013) though only SRP was measured; in another study, plots of turfgrass or mixed ornamentals (trees, shrubs, and groundcovers) both had substantially higher P leaching (0.2-0.4 g P m<sup>-2</sup> yr<sup>-1</sup>; Erickson et al. 2005).

### *Uncertainties in hydrologic modeling*

Our predicted differences in water fluxes, where urban trees have lower ET and higher leaching than turfgrass, run counter to the paradigm developed from watershed studies, where forested watersheds have higher ET and lower streamflow than grassy watersheds (see reviews in Zhang et al. 2001, Brown et al. 2005). This discrepancy is due primarily to differences in estimating tree ET: Comparing our modeled ET with watershed ET predictions from Zhang and



coauthors (2001, Eq. 8), our estimates for urban trees are lower than those for forested watersheds, while our estimates for open turfgrass are similar to those for grassy watersheds. Our tree transpiration parameters were calibrated to local empirical sapflow data (Peters et al. 2010); the authors report that their findings are comparable to non-urban measurements of the same species in similar vapor-pressure deficit conditions, lending confidence in their data. Furthermore, the only other empirical study we could find that compared urban tree and turfgrass ET or leaching without confounding factors (e.g. lawn irrigation) was a study in Baltimore, MD, USA, where zero-tension lysimeters consistently collected larger volumes of water under remnant deciduous forest patches than under turfgrass areas (Groffman et al. 2009). These studies suggest that the dominant paradigm, that trees have higher ET than turfgrass, is not necessarily accurate for urban systems, and highlight the need for further empirical studies.

In addition, no studies (to our knowledge) have measured ET or leaching for open-grown trees with grassy understories, despite the prevalence of these conditions in urban areas. Peters and coauthors (2011) also modeled an urban combined tree-turfgrass system and estimated that its ET is lower than open turfgrass ET, but their model and ours still need to be validated against empirical data.

If our model were underestimating urban tree ET (and overestimating vertical water fluxes), increasing tree ET would reduce our estimates for N and P leaching under trees, reducing our overall estimates for nutrient leaching in the watershed and amplifying the differences between trees and turfgrass. The differences among vegetation types' P leaching would be greater than our current estimates, and the differences in N leaching could increase to the point where they become statistically discernable. As such, our finding that trees decrease P leaching is robust against the uncertainties in our hydrologic model, while the lack of discernable differences in N

leaching is more sensitive to the hydrologic model.

Our sensitivity analysis showed that our quantitative N and P flux estimates are not overly sensitive to changes in key hydrologic model parameters. If our estimates of LAI and transpiration were systematically biased by 10%, and soil texture by one class, that would affect our flux estimates by 10% or less. We therefore consider our results fairly robust against uncertainties in estimating these parameters.

### *Uncertainties in scaling up*

Our watershed-scale estimates of both total nutrient leaching and tree effects contain considerable uncertainty. We assumed that tree effects are proportional to the projected canopy area over pervious surfaces; however, tree rooting zones generally extend well beyond the canopy edge (Crow 2005) and affect a larger area. Our calculations therefore likely underestimated the amount by which trees reduced P leaching, and overestimated total leaching in the watershed. In addition, we did not investigate the spatial heterogeneity in soil water N and P concentrations under a tree to understand either random heterogeneity or systematic variation with distance from the trunk, so we could not test our assumption that concentrations measured halfway between the trunk and canopy edge represent an average value. In scaling up to the watershed, we further assumed that all ages and sizes of trees have similar leaching rates per m<sup>2</sup> of canopy area, although we only studied mature trees.

These factors highlight the uncertainty associated with using our data on open-grown trees in parks to estimate nutrient leaching under trees in other situations such as boulevard trees, which comprise a substantial portion of the urban forest. Trees near streets or building foundations have constrained and distorted rooting zones compared to open-grown trees, likely leading to different

ratios of tree root mass (and hence processes such as root litter inputs) per unit of canopy area over pervious surfaces. Similarly, any litter- or wood-based processes reducing leaching (litter export, nutrient storage in wood) would not be decreased by having impervious surface under the canopy and may have been underestimated by excluding tree canopy over impervious surfaces.

Management practices in our study parks are also not representative of the entire watershed. Leaf litter in the parks was mown into the grass and retained on-site, while considerable amounts of leaf litter in other sites are removed from the property, exporting N and P and likely decreasing leaching. For example, 66% of watershed residents reported taking leaf litter off-site to county composting facilities (Wein et al. 2010). We also likely underestimated N leaching in fertilized areas compared to our unfertilized parks; 52% of residential lawns in the watershed are fertilized (Wein et al. 2010), as are most city parks in suburbs outside of Saint Paul (E. Peters, *personal communication*) and all turfgrass areas in golf courses, cemeteries, and college/university campuses in the Capitol Region Watershed (Hobbie et al., *in prep.*). Note that P is restricted from lawn fertilizer in Minnesota, so fertilization does not directly add P but could affect P cycling.

Finally, our watershed-scale estimates assume that 100% of nutrient fluxes at 60 cm reach groundwater. As discussed above (*Discussion – Nutrient leaching to groundwater*), while some removal or retention likely takes place below 60 cm, we expect that the bulk of N and P fluxes we measured do reach groundwater and contribute to nutrient loading in local lakes and the Mississippi River.

### *Tree effects*

We found that trees in grassy areas reduced P leaching to groundwater, with lower leaching under deciduous than evergreen trees. In contrast, our data do not give a clear answer whether trees increase or decrease N leaching, since trees had lower leaching than turfgrass in 2012 but higher leaching than turfgrass in 2013. Much of the N leaching took place during high springtime pulses, the beginnings of which we likely did not sample completely because both the 2012 and 2013 growing seasons began with a period of rewetting after severe drought that made it difficult to obtain lysimeter samples. In addition, the magnitude of these pulses and the differences among vegetation types may also be different during normal hydrologic years than during the post-drought rewetting we observed, as drying/rewetting cycles have been shown to increase N leaching (Gordon et al. 2008). We recommend sampling at least one spring in non-drought conditions to better understand tree effects on N leaching in more normal hydrologic conditions.

Trees can reduce P leaching either by reducing P inputs and/or increasing P storage (in soil and/or biomass); we consider it more likely that trees increased P storage. P inputs to our unfertilized study sites were primarily from atmospheric deposition, soil mineral weathering, and pet waste. Although tree leaves intercept P-containing fine dust, this would not have reduced inputs since the P either washed off leaves during rain events or became part of leaf-fall. We consider it unlikely that trees substantially reduced mineral weathering rates relative to turfgrass, since soil water fluxes were slightly higher under trees, and because the increased root mass and rooting depth of trees is likely to have led to more root exudates and a slight increase in weathering. We also consider it unlikely that pet waste inputs were substantially lower under trees than in open grassy areas, given the propensity of dogs to urinate on tree trunks. (This is in contrast with N inputs: we observed less clover growing in the shade of trees, suggesting that

trees could reduce N inputs by decreasing N fixation.)

Trees can increase P storage in soil or wood. Although we did not directly measure soil P, due to the difficulty in separating biologically meaningful fractions, we examined soil nutrient storage through relationships with litter chemistry. If trees were increasing soil storage of P by accumulating P in soil organic matter, we hypothesized that trees with lower litter nutrient concentrations would have slower-decomposing litter, higher storage of nutrients in soil organic matter, and lower nutrient leaching. Although we found significant relationships between soil water P concentrations and litter C:P ratios, this was positive among deciduous trees and negative among evergreens. Because the relationship among deciduous trees is opposite to our hypothesis, our data do not offer clear support for trees increasing soil storage of P.

To estimate P storage in tree wood, we used iTree estimates of annual tree growth, reported in kg C/yr as “gross C sequestration.” We assumed a constant C:P ratio of 1533.3 (Fissore et al. 2011), since species-specific estimates are not available, to estimate that average annual P storage in tree wood was 9.9 g for deciduous trees, and 3.5 g for evergreens. This is substantially larger than our estimates of the amount by which an individual typical-sized tree reduced annual P leaching fluxes relative to turfgrass (deciduous: 1.1 g in 2012, 2.3 g in 2013; evergreen: 0.1 g in 2012, 0.7 g in 2013), suggesting that P storage in wood is large enough to account for the observed reductions in P leaching. However, we did not find a statistically discernable relationship between growth rate and soil water P concentrations (data not shown), so there is no evidence that differences in P storage drive differences among trees within vegetation types.

### *Alternative mechanisms*

Differences in P concentrations and fluxes between trees and turfgrass do not necessarily demonstrate that trees reduced overall watershed-scale P leaching to groundwater. We examined three alternative mechanisms that could cause differences in soil water P concentration between trees and turfgrass: 1) Trees establish preferentially in microsites with different abiotic conditions; 2) Trees spatially redistribute nutrients to turfgrass areas, creating spatial heterogeneity but no net reduction in leaching; and 3) Trees store nutrients during the growing season, temporarily reducing leaching, but release those nutrients to be leached during the fall. If one or more of these alternative mechanisms were the primary driver(s) of the observed differences in soil water P concentration, planting or removing trees would not change the amount of P leaching to groundwater.

The first alternative hypothesizes that pre-existing abiotic differences favor tree establishment in microsites with favorable soil properties that might also have lower P leaching. However, our focal trees were planted, rather than establishing naturally, and their locations and species were selected for reasons that do not relate to microsite patterns of nutrient cycling (e.g. aesthetics, salt tolerance, overhead wires). As a result, we do not believe that pre-existing abiotic differences are responsible for the patterns we observed.

The second and third alternatives hypothesize that trees do not affect the total P leaching in a watershed, but only redistribute the P leaching spatially and/or temporally. We observed that deciduous leaf litter blows around extensively in open parks and lawns. This drives a net export of soil nutrients from a tree's rooting zone, since there is not an equivalent import of material from open turfgrass areas. Litter export likely decreases nutrient leaching beneath trees and increases nutrient leaching in turfgrass areas that receive the litter. Similarly, temporal

redistribution is driven by temporary storage of P in the canopy during spring and summer, reducing leaching during spring and summer but increasing it after litterfall (which we could not measure directly due to the drought).

To examine the importance of spatial and temporal redistribution driving differences in P leaching, we explored trait relationships among individual trees. The total mass of P in a deciduous tree's litter (leaf biomass x litter P concentration) is the amount stored temporarily in the canopy, dropped during litterfall, and then available for redistribution; trees with greater litter P masses should therefore have lower P leaching if spatial and/or temporal redistribution were an important mechanism. This relationship does exist for deciduous trees; however, litter P mass is confounded with litter P concentrations, which we expect to affect other mechanisms such as decomposition rates. To disentangle these mechanisms, we focused on leaf biomass and examined whether leaf biomass drove differences in P leaching among trees with similar litter P concentrations.

In a regression using litter P concentration and leaf biomass as predictors for soil water P concentrations (data not shown), leaf biomass was only statistically significant at quantiles  $\tau < 0.24$ , explaining relatively small differences among trees with low soil water P concentrations. At central and upper quantiles, which are of greater interest, leaf biomass was not a significant predictor of soil water P concentrations; only litter P concentration was a significant predictor. Because leaf biomass did not explain differences at central or upper quantiles, our data suggest that spatial and temporal redistribution are not substantial drivers of the differences we observed in soil water P concentration. In addition, for the two trees cut down during winter 2011-2012,

soil water P concentrations remained low through 2013. This suggests that lower P leaching under trees is due to a mechanism that persists at least for a couple of years, rather than to temporary seasonal redistribution.

Overall, our data do not support any of these three alternative mechanisms that could produce the differences we observed among vegetation types. As a result, we believe that trees are causing overall watershed-scale differences in P leaching to groundwater, and that planting or removing trees will affect total P leaching to groundwater.

#### *Assessing additional species and sites*

At this time, we cannot confidently recommend tree species that would most reduce nutrient leaching. We found higher than expected within-species variability in both soil water nutrient concentrations and tissue chemistry (Figures 3 and 4). For the species where we sampled multiple individuals, variation within the species was often larger than the variation among species. We recommend broader sampling of tree tissue chemistry in urban environments to understand the variability within species, and also to resolve differences among species, before making any recommendations favoring certain tree species.

With over 130 tree species identified in the Minneapolis-Saint Paul metropolitan area (Fissore et al., *unpublished data*) and a variety of different management practices that affect nutrient cycling (e.g. fertilization and irrigation), it is important to find a way of readily assessing many more trees than would be practical using lysimeter studies. The most direct method would be to



Accepted Article

extract soil water from cores for a one-time measurement of soil water N and P concentrations (Lajtha et al. 1999), but coring to at least 60 cm is fairly labor-intensive, impractical in rocky soils, and may require large volumes of soil to yield enough water for analysis in coarse soils like our study sites. Alternately, our data suggest that N mineralization and Brays-P assays using small, shallow soil cores can be useful for estimating leaching. In addition, plant trait relationships offer the possibility of leveraging extensive existing trait databases to identify species likely to have low nutrient leaching, though these databases usually contain measurements from a small number of non-urban individuals and any leaching comparisons from them would be uncertain. Measuring leaf N and P concentrations directly for species of interest (sampling multiple individuals of each species) in an urban area could then provide a more accurate, but still relatively simple, follow-up study. We recommend that further studies of nutrient leaching continue to test correlations with plant traits and soil assays in order to develop protocols that can readily assess many combinations of tree species and management practices relevant to urban-forest decision-making.

## Conclusion

Urban trees can substantially reduce P, but not N, leaching to groundwater, and have the potential to help reduce nutrient pollution of local waterbodies. While the effect of any individual tree is fairly small, their aggregate effect can be important. Across the Capitol Region Watershed (~17,400 ha, 33% tree cover), we estimated that this P reduction was worth approximately \$2-5 million per year. Future studies can help to reduce uncertainties in our estimates, especially of vertical water fluxes, retention or removal in subsoils, and variability

within tree species. Our findings highlight that trees' potential to reduce nutrient leaching warrants increased attention from both managers and scientists working to improve and protect water quality.

## **Acknowledgments**

We are grateful to Adam Robbins and Rachel Coyle of Saint Paul Parks & Recreation, for helping us obtain permission and navigate logistics of sampling in city parks; to Tess Carley, Sandy Brovold, Chris Buyarski, Max Grossman, and Charlotte Riggs for field and lab assistance; and to Jared Trost and John Nieber for assistance with lysimeters. Jacques Finlay, Rebecca Montgomery, Casey Peel, Bill Currie, and two anonymous reviewers also provided valuable comments on earlier drafts. This study was supported by the University of Minnesota Center for Urban and Regional Affairs Dissertation Grant, UMN Institute on the Environment Discovery Grant, Plant Traits Course Small Grant, and the Bell Museum Dayton Fellowship Fund, as well as fellowships to D. A. Nidzgorski from the National Science Foundation Graduate Research Fellowship, the UMN Graduate School Fellowship, and the Kence Foundation.

## **Literature Cited**

- Amador, J., R. Hull, E. Patenaude, J. Bushoven, and J. Gorres. 2007. Potential nitrate leaching under common landscaping plants. *Water Air and Soil Pollution* 185:323–333.
- Bray, R. H., and L. T. Kurtz. 1945. Determination of total, organic, and available forms of phosphorus in soils. *Soil science* 59:39–46.

- Brown, A. E., L. Zhang, T. A. McMahon, A. W. Western, and R. A. Vertessy. 2005. A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. *Journal of Hydrology* 310:28–61.
- Cade, B. S., and B. R. Noon. 2003. A gentle introduction to quantile regression for ecologists. *Frontiers in Ecology and the Environment* 1:412–420.
- Capitol Region Watershed District. 2010. Stormwater BMP performance assessment and cost-benefit analysis. Page 107. Saint Paul, MN.
- Crow, P. 2005. The Influence of Soils and Species on Tree Root Depth. Page 8. Information Note, Forestry Commission.
- DeMott, W. R., R. D. Gulati, and K. Siewertsen. 1998. Effects of phosphorus-deficient diets on the carbon and phosphorus balance of *Daphnia magna*. *Limnology and Oceanography* 43:1147–1161.
- Dewalle, D. R., and G. M. Heisler. 2012. Use of Windbreaks for Home Energy. *Agriculture, Ecosystems, and the Environment* 22:243–260.
- Dijkstra, F. A., K. Wrage, S. E. Hobbie, and P. B. Reich. 2006. Tree Patches Show Greater N Losses but Maintain Higher Soil N Availability than Grassland Patches in a Frequently Burned Oak Savanna. *Ecosystems* 9:441–452.
- Doane, T. A., and W. R. Horwath. 2003. Spectrophotometric Determination of Nitrate with a Single Reagent. *Analytical Letters* 36:2713–2722.
- Dubrovsky, N. M., and P. A. Hamilton. 2010. The Quality of Our Nation's Water: Nutrients in the Nation's Streams and Groundwater; National Findings and Implications. US Geological Survey, National Water-Quality Assessment Program.
- Erickson, J. E., J. L. Cisar, G. H. Snyder, D. M. Park, and K. E. Williams. 2008. Does a Mixed-

Species Landscape Reduce Inorganic-Nitrogen Leaching Compared to a Conventional St. Augustinegrass Lawn? *Crop Science* 48:1586.

Erickson, J. E., J. L. Cisar, G. H. Snyder, and J. C. Volin. 2005. Phosphorus and Potassium Leaching under Contrasting Residential Landscape Models Established on a Sandy Soil. *Crop Science* 45:546.

Federer, C. A., C. Vörösmarty, and B. Fekete. 2003. Sensitivity of Annual Evaporation to Soil and Root Properties in Two Models of Contrasting Complexity. *Journal of Hydrometeorology* 4:1276–1290.

Fissore, C., L. A. Baker, S. E. Hobbie, J. Y. King, J. P. McFadden, K. C. Nelson, and I. Jakobsdottir. 2011. Carbon, nitrogen, and phosphorus fluxes in household ecosystems in the Minneapolis-Saint Paul, Minnesota, urban region. *Ecological Applications* 21:619–639.

Geiger, S. C., R. J. Vandenberg, and A. Manu. 1994. Variability in the growth of *Faidherbia albida*: the soils connection. *Soil Science Society of America journal (USA)* 58:227–231.

Gill, R. A., and I. C. Burke. 1999. Ecosystem consequences of plant life form changes at three sites in the semiarid United States. *Oecologia* 121:551–563.

Gold, A. J., W. R. DeRagon, W. M. Sullivan, and J. L. Lemunyon. 1990. Nitrate-nitrogen losses to groundwater from rural and suburban land uses. *Journal of Soil and Water Conservation* 45:305–310.

Gordon, H., P. M. Haygarth, and R. D. Bardgett. 2008. Drying and rewetting effects on soil microbial community composition and nutrient leaching. *Soil Biology and Biochemistry* 40:302–311.

Groffman, P., C. Williams, R. Pouyat, L. Band, and I. Yesilonis. 2009. Nitrate Leaching and

Nitrous Oxide Flux in Urban Forests and Grasslands. *Journal of Environmental Quality* 38:1848–1860.

Guillard, K., and K. L. Kopp. 2004. Nitrogen Fertilizer Form and Associated Nitrate Leaching from Cool-Season Lawn Turf. *Journal of Environment Quality* 33:1822.

Gundersen, P., I. Callesen, and W. de Vries. 1998. Nitrate leaching in forest ecosystems is related to forest floor C/N ratios. *Environmental Pollution* 102:403–407.

Hobbie, S. E. 1992. Effects of plant species on nutrient cycling. *Trends in Ecology & Evolution* 7:336–339.

Iseman, T. M., D. R. Zak, W. E. Holmes, and A. G. Merrill. 1999. Revegetation and nitrate leaching from lake states northern hardwood forests following harvest. *Soil Science Society of America Journal* 63:1424.

Jackson, R. B., J. L. Banner, E. G. Jobbagy, W. T. Pockman, and D. H. Wall. 2002. Ecosystem carbon loss with woody plant invasion of grasslands. *Nature* 418:623–626.

Jackson, R. B., J. Canadell, J. R. Ehleringer, H. A. Mooney, O. E. Sala, and E. D. Schulze. 1996. A Global Analysis of Root Distributions for Terrestrial Biomes. *Oecologia* 108:389–411.

Janke, B. D., J. C. Finlay, S. E. Hobbie, L. A. Baker, R. W. Sterner, D. Nidzgorski, and B. N. Wilson. 2013. Contrasting influences of stormflow and baseflow pathways on nitrogen and phosphorus export from an urban watershed. *Biogeochemistry*:1–20.

Kilberg, D., and M. E. Bauer. 2011. Mapping Capitol Region Watershed District Urban Tree Canopy. Remote Sensing and Geospatial Laboratory, University of Minnesota, Saint Paul, MN.

Koenker, R. 1994. Confidence intervals for regression quantiles. Pages 349–359 *Asymptotic statistics*. Springer.

- Koenker, R., and K. Hallock. 2001. Quantile regression: An introduction. *Journal of Economic Perspectives* 15:43–56.
- Kuo, F. E., and W. C. Sullivan. 2001. Environment and crime in the inner city: Does vegetation reduce crime? *Environment and Behavior* 33:343–367.
- Lajtha, K., W. M. Jarrell, D. W. Johnson, and P. Sollins. 1999. Collection of soil solution. Pages 166–182 *Standard soil methods for long-term ecological research*. New York: Oxford University Press.
- Larson, G. A. 1985. Phosphorus concentrations in surface water, ground water, and soils in the Anoka sand plain and peatlands. M.S. Plan B Paper, University of Minnesota, Saint Paul, MN.
- Loper, S. J., A. L. Shober, C. Wiese, G. C. Denny, and C. D. Stanley. 2013. Nutrient Leaching during Establishment of Simulated Residential Landscapes. *Journal of Environment Quality* 42:260.
- Lovett, G. M., K. C. Weathers, and M. A. Arthur. 2002. Control of Nitrogen Loss from Forested Watersheds by Soil Carbon:Nitrogen Ratio and Tree Species Composition. *Ecosystems* 5:712–718.
- McCarron, J. K., A. K. Knapp, and J. M. Blair. 2003. Soil C and N responses to woody plant expansion in a mesic grassland. *Plant and Soil* 257:183–192.
- Milesi, C., S. W. Running, C. D. Elvidge, J. B. Dietz, B. T. Tuttle, and R. R. Nemani. 2005. Mapping and Modeling the Biogeochemical Cycling of Turf Grasses in the United States. *Environmental Management* 36:426–438.
- Miller, R. W. 1997. *Urban forestry: planning and managing urban greenspaces*. Second edition. Prentice Hall.

- Nassauer, J. I., and P. Opdam. 2008. Design in science: extending the landscape ecology paradigm. *Landscape Ecology* 23:633–644.
- Nowak, D., D. Crane, R. Hoehn, J. Stevens, and J. Walton. 2006a. Assessing Urban Forest Effects and Values: Minneapolis' Urban Forest. US Forest Service, Northeastern Research Station.
- Nowak, D. J., D. E. Crane, and J. C. Stevens. 2006b. Air pollution removal by urban trees and shrubs in the United States. *Urban Forestry & Urban Greening* 4:115–123.
- Peters, E. B., R. V. Hiller, and J. P. McFadden. 2011. Seasonal contributions of vegetation types to suburban evapotranspiration. *Journal of Geophysical Research: Biogeosciences* 116:n/a–n/a.
- Peters, E. B., and J. P. McFadden. 2010. Influence of seasonality and vegetation type on suburban microclimates. *Urban Ecosystems* 13:443–460.
- Peters, E. B., J. P. McFadden, and R. A. Montgomery. 2010. Biological and environmental controls on tree transpiration in a suburban landscape. *Journal of Geophysical Research: Biogeosciences* 115:n/a–n/a.
- Qin, Z., A. L. Shober, R. C. Beeson, and C. Wiese. 2013. Nutrient Leaching from Mixed-Species Florida Residential Landscapes. *Journal of Environment Quality* 42:1534.
- Rhoades, C. 1997. Single-tree influences on soil properties in agroforestry: lessons from natural forest and savanna ecosystems. *Agroforestry Systems* 35:71–94.
- Saint Paul Department of Parks and Recreation. 2013. Park and boulevard tree planting list.
- Strahm, B. D., R. B. Harrison, T. A. Terry, B. L. Flaming, C. W. Licata, and K. S. Petersen. 2005. Soil solution nitrogen concentrations and leaching rates as influenced by organic matter retention on a highly productive Douglas-fir site. *Forest Ecology and Management*

218:74–88.

Trojan, M. D., J. S. Maloney, J. M. Stockinger, E. P. Eid, and M. J. Lahtinen. 2003. Effects of land use on ground water quality in the Anoka Sand Plain Aquifer of Minnesota. *Ground Water* 41:482–492.

Vetaas, O. 1992. Micro-site effects of trees and shrubs in dry savannas. *Journal of vegetation science* 3:337–344.

Wein, S. P., K. C. Nelson, L. A. Baker, S. E. Hobbie, J. Y. King, J. P. McFadden, C. Fissore, I. Jakobsdottir, D. A. Nidzgorski, and D. Burk. 2010. Homeowner behaviors that affect nitrogen and phosphorus fluxes in household landscapes: Watershed District and Watershed Management Organization Report. University of Minnesota, Saint Paul, MN.

Willis, R. B., G. J. Schwab, and C. E. Gentry. 1993. Elimination of interferences in the colorimetric analysis of ammonium in water and soil extracts. *Communications in Soil Science and Plant Analysis* 24:1009–1019.

Zhang, L., W. R. Dawes, and G. R. Walker. 2001. Response of mean annual evapotranspiration to vegetation changes at catchment scale. *Water Resources Research* 37:701–708.

## **Tables**



Table 1: Summary statistics for soil water nutrient concentrations (all samples, all years) by vegetation type.

	Veg. type	Mean	Median	Std. Error	n	significance
Total dissolved N (mg/L)						$p=8.0 \times 10^{-4}$
	turfgrass	7.32	2.56	1.08	94	a
	evergreen	7.07	2.39	0.95	94	a
	deciduous	3.75	1.18	0.55	195	b
NO <sub>3</sub> <sup>-</sup> -N (mg/L)						$p=3.2 \times 10^{-4}$
	turfgrass	5.63	1.97	1.00	77	a
	evergreen	5.95	1.37	0.97	77	a
	deciduous	2.46	0.25	0.42	146	b
Total dissolved P (µg/L)						$p=5.3 \times 10^{-11}$
	turfgrass	159.26	72.70	19.88	94	a

	evergreen	84.61	36.94	12.95	97	b
	deciduous	49.86	31.07	4.23	191	c
Soluble reactive P ( $\mu\text{g/L}$ )						$p=5.2 \times 10^{-11}$
	turfgrass	131.23	54.45	19.92		a
	evergreen	59.42	13.55	11.16		b
	deciduous	24.98	12.80	3.12		c

Note: Different lowercase letters (a, b, c) indicate statistically discernable pairwise comparisons at  $p < 0.05$ .

Table 2: Total nutrient leaching fluxes below 60 cm depth for each year's sampling period, and total for the overall sampling period (2011-2013), averaged by vegetation type.

		C		N		P		
2011		(g m <sup>-2</sup> sampling period <sup>-1</sup> )						
	turfgrass	2.58		0.27		0.0115		
	evergreen	11.98		0.51		0.0051		
	deciduous	18.74		0.18		0.0041		
2012								
	turfgrass	2.70		1.50		0.0175		
	evergreen	15.14		1.49		0.0152		
	deciduous	12.00		0.46		0.0069		
2013								
	turfgrass	7.14		1.37		0.0409		

	evergreen	11.80		2.63		0.0267	
	deciduous	10.90		2.15		0.0180	
Overall			$p=6.5 \times 10^{-8}$		$p=1.5 \times 10^{-3}$		$p=8.3 \times 10^{-10}$
	turfgrass	12.42 a		3.14 a		0.0700 a	
	evergreen	38.91 b		4.62 b		0.0471 b	
	deciduous	41.64 b		2.78 a		0.0289 c	

Notes: Different lowercase letters (a, b, c) indicate statistically discernable pairwise comparisons at  $p < 0.05$ .

The sampling period differed for each individual tree or turfgrass area, spanning the dates during that calendar year when we were able to collect soil water in the lysimeter. The 2011 sampling period began with the study start in late July, and hence only included a portion of annual leaching fluxes. The 2012 and 2013 sampling periods spanned from soil wet-up in spring to dry-down in summer, and these values approximate annual leaching fluxes.

Table 3: Species codes for Figures 3 and 4

Code	Species
Ac	<i>Abies concolor</i>
Ap	<i>Acer platanoides</i>
As	<i>Acer saccharinum</i>
Co	<i>Celtis occidentalis</i>
Fp	<i>Fraxinus pennsylvanica</i>
Gb	<i>Ginkgo biloba</i>
Gt	<i>Gleditsia triacanthos</i>
Pp	<i>Pinus ponderosa</i>
Pr	<i>Pinus resinosa</i>
Ps	<i>Pinus strobus</i>
Py	<i>Pinus sylvestris</i>
Qm	<i>Quercus macrocarpa</i>
Tc	<i>Tilia cordata</i>

## Figure captions:

**Figure 1:** Timeseries of soil water total dissolved N (TDN) and total dissolved P (TDP) concentrations, averaged by vegetation type.

**Figure 2:** Average daily N and P fluxes for each calendar month, averaged by vegetation type.

**Figure 3:** Average springtime (first two sampling dates for 2012 and 2013) soil water total dissolved N (TDN) concentrations for each focal tree were best predicted by root %N for deciduous trees, and leaf biomass for evergreens. Solid lines are median regression lines (excluding *Gleditsia triacanthos*). Species codes are listed in Table 3; intercepts and slopes are plotted across all quantiles in Appendix A, Figure A5.

**Figure 4:** Average soil water total dissolved P (TDP) concentrations for each focal tree were best predicted by litter C:P ratios. Solid lines are median regression lines; dashed lines are regression lines near the upper and lower extent of statistical significance. For deciduous trees, these are at quantiles  $\tau=0.3$  and  $\tau=0.9$ ; for evergreens, these are at  $\tau=0.35$  and  $\tau=0.7$ . Species codes are listed in Table 3; intercepts and slopes are plotted across all quantiles in Appendix A, Figure A6.







