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# Vegetation type alters water and nitrogen budgets in a controlled, replicated experiment on residential-sized rain gardens planted with prairie, shrub, and turfgrass

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**Abstract** Vegetation type alters surface hydrology and nitrogen (N) cycling by partitioning evapotranspiration (ET) and drainage, but has not been fully utilized for this purpose in the design of urban rain gardens. Replicated treatments of three vegetation types (prairie, shrub, and turfgrass) commonly planted in rain gardens, as well as bare soil (control), were evaluated 1 year after establishment in 12, free-drainage bioretention cells designed to specifications for residential rain gardens in the Midwestern United States. Water and N budgets were calculated to assess differences in ET, drainage, soil moisture, and N transport following three stormwater applications in July, August, and October of 2006. Evaporative demand was also estimated as potential or reference ET during the study period using Hamon, Priestley-Taylor, and FAO Penman-Monteith models. Results show that after 1 year of establishment, vegetation type changed the water budgets of rain gardens thus altering their provisioning of ecosystem services via observed tradeoffs between daytime evaporative cooling (ET), stormwater storage, N-load reduction, soil and plant N retention, and groundwater recharge (drainage). Vegetation effects on hydrology and N transport were most significant when evaporative demand was high during the middle of the growing season. Observed changes in ET, drainage, soil moisture, and N transport support the incorporation of different vegetation types into conceptual and numerical models of rain gardens to assess ecological outcomes and tradeoffs at a variety of spatiotemporal scales.

**Keywords** Rain gardens · Bioretention basins · Prairie · Shrubs · Turfgrass · Ecosystem services

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# Introduction

Impervious surfaces often increase the magnitude, speed, and chemical load of stormwater runoff to freshwater ecosystems within and surrounding cities (Arnold and Gibbons 1996). Over the past 20 years, municipalities have implemented a wide variety of green infrastructure (e.g. rain gardens, green roofs, bioswales) in efforts to control stormwater quantity and quality in and from urban environments by mimicking prede-velopment biogeochemical processes (United States Environmental Protection Agency 2012). Green infrastructure can attenuate the flashiness of stormwater flows through detention and retention, reduce pollution loads through detention and filtration, and dampen temperature extremes (urban heat island) through evaporative cooling and dissipation of heat from impervious runoff (Demuzere et al. 2014). Likewise, from a socioeconomic perspective green infrastructure can increase the aesthetics, property value, safety, psychological health, and pride of communities by creating open, active green spaces (Johnston 2005; Dunn 2010). These regulatory and cultural ecosystem services promote green infrastructure as a recurring theme in scenarios of plausible urban futures, which integrate stakeholder visions and biogeochemical modeling to predict long-term ecological responses to climate, land-use, and socioeconomic change (Carpenter et al. 2006; Gill et al. 2007; Carpenter et al. 2015). However, the ecosystem services, potential disservices, and tradeoffs from green infrastructure can be unpredict-able and require a more mechanistic understanding from controlled, replicated field studies of biogeochemical processes (Pataki et al. 2011). This study focuses on a prevalent green infrastructure practice that continues to outpace its assessment—residen-tial-sized rain gardens—and systematically explores the detailed biogeochemical pro-cesses, ecosystem services, disservices, and tradeoffs associated with vegetation type, a key design variable.

Rain gardens (4–10 m$^2$) are typically constructed by excavating and backfilling an existing topographical depression with an engineered mix of soil, sand, and compost (Thompson et al. 2008); the structures are usually planted with a variety of vegetation types, often native plants, shrubs, and grasses (Prince George's County 1993) that have been shown to alter soil physical properties (Johnston et al. 2016). The anticipated ecosystem services provided by rain gardens (e.g. maintenance of pre-development hydrology, pollution load reduction, and urban heat island mitigation) are largely an-thropocentric and require further characterization as quantifiable exchanges of water, nutrients, and energy. Key measurable exchanges include (1) soil moisture storage and drainage (deep percolation), which retain stormwater runoff and increase groundwater recharge; (2) soil nitrogen (N) retention and plant N uptake, which protect adjacent surface waters by filtering N loads from stormwater runoff; and (3) evapotranspiration (ET), which could mitigate daytime urban heat island effects through evaporative cooling. These exchanges of water, nutrients, and energy occur within the soil-plant-atmospheric continuum and vary with stormwater input, rain garden design, soil proper-ties, and vegetation type. Although soil properties and the overall design of rain gardens have been evaluated (Hsieh and Davis 2005; Davis 2008; Thompson et al. 2008; Carpenter and Hallam 2010), the isolated effect of vegetation type on the transport and cycling of water and N has received less experimental attention.

Given the long understood role of vegetation in the partitioning of stormwater into ET, storage and drainage under different land covers, the selection of vegetation type

likely causes a tradeoff between groundwater recharge and evaporative cooling in rain gardens. The design element underlying this tradeoff is the difference in rain garden storage capacity, which can be controlled by altering the drainage boundary condition or vegetation type. Wadzuk et al. (2014) altered the drainage boundary condition to include an internal water storage layer and found that ET rates were twice that of a free-drainage rain garden using weighing lysimeters. Vegetation selection could similarly modify ET and drainage partitioning by changing rain garden storage capacity. Johnston (2011) found that rain gardens planted with prairie and shrub vegetation had significantly lower soil volumetric water content at depths of 0–0.15 and 0.30–0.45 m (3–4 and 10 % lower, respectively) compared to turfgrass prior to storm events, which suggests that vegetation selection may shift the storage capacity of rain gardens during dry periods via ET-induced changes to antecedent water content. Though climatic parameters have been shown to strongly correlate to actual ET from rain gardens (Hickman et al. 2011), these parameters determine the evaporative demand or potential ET of the systems. When all climatic parameters are equal, different vegetation types will vary in their actual ET based on physiological responses to the opposing tension between soil moisture supply, conductance, and external evaporative demand (Tanner 1957).

The drainage rate from rain gardens may also be altered by vegetation type, which can change soil hydraulic conductivity through root growth and morphology. For example, Le Coustumer et al. (2012) conducted a biofiltration column study testing five different species relative to a bare soil control over 72 weeks found that a thick-rooted species, *Melaleuca ericifolia*, significantly increased soil hydraulic conductivity over the course of the experiment, which was attributed to its high mean root diameter. Although components of the water budget such as drainage and ET have been previously quantified (Davis 2008; Hatt et al. 2009; Li et al. 2009; Selbig and Balster 2010; Hickman et al. 2011; Johnston 2011; Wadzuk et al. 2014), there are limited replicated field studies experimentally isolating the effect of vegetation type on the hydrological budgets of residential-sized rain gardens.

Differences in vegetation type may modify the transport and cycling of N from rain gardens through changes in plant N uptake, soil N retention, and stormwater N detention, especially following dry periods (Mangangka et al. 2015). Different plants have been shown to vary up to 170-fold and 570-fold in nitrate or nitrite ($NO_3^-$, $NO_2^-$) and ammonium ($NH_4^+$) drainage concentrations, respectively, in a pot trial of 20 species typically used for rain gardens in Australia (Read et al. 2008). In a follow-up study, Read et al. (2009) found that rooting depth, longest root, total plant mass, and relative growth rate were negatively correlated to $NO_3^-$ and $NH_4^+$ concentrations in semi-synthetic stormwater drained from bioretention cells. These findings suggest that differences in above and below ground structure and biomass may correlate to differences in N transport from rain gardens with different vegetation types. Additionally, studies from tile-drained agricultural ecosystems have shown that plant differences in ET and rooting depth alter the coupled drainage of water and N (Johnsson et al. 1987). These known impacts of vegetation type on hydrology and N transport argue for a greater understanding of vegetation in the performance of rain gardens to inform design and assess functional tradeoffs.

Our primary objective was to evaluate the effect of vegetation type on the hydrological budgets and corresponding ecosystem services of residential-sized rain gardens by specifically investigating changes in ET, drainage, soil moisture storage, N cycling, and N transport. We compared replicated rain gardens planted with prairie, shrub, and

turfgrass to bare soil rain gardens to better understand baseline hydrology and N transport with changes in vegetation. We hypothesized that after 1 year of establishment, (1) total drainage following stormwater applications would rank in the following order: bare soil > turfgrass > shrub > prairie, explained by differences in ET and soil moisture ranking in the reverse order and (2) N load reduction following stormwater applications would rank in the following order: prairie > shrub > turfgrass > bare soil, explained by differences in N concentration observed in soil, plant tissue, and drainage.

## Materials and methods

### Experimental design

We established a 1500 m$^2$ research site in 2005 consisting of 12 field-scale bioretention cells (2.4 × 2.4 × 1.2 m, Fig. 1a) in the urban agglomeration (population: 402,000) of Madison, Wisconsin, which comprises the Yahara Watershed (U.S. Census Bureau 2012). In physical terms, the experimental bioretention cells function as free-drainage lysimeters, where the lower 1.2 m boundary is exposed to atmospheric pressure, thus allowing gravitational drainage and excluding drainage controlled by an actual or imposed matric potential gradient (Abdou and Flury 2004). The bioretention cells were replicated on level ground along a linear path with 1.2 m spacing between each cell. Each bioretention cell was sized according to the Wisconsin Department of Natural Resources (WI-DNR) specifications (depth, surface area, receiving area, soil mixture) recommended for residential rain gardens (Bannerman and Considine 2003; Dussaillant et al. 2004) that are generally applicable to the Midwestern United States.

Each bioretention cell was constructed with a centered channel drain along the bottom to facilitate free-drainage conditions (Fig. 1b). Because rain gardens are positioned in landscape depressions to receive and not contribute to surface runoff, we intentionally eliminated surface runoff from the bioretention cells by installing a 0.15 m raised rim around each bioretention cell. The inside walls and bottom of each cell were fitted with a 30 mil rubber liner; 0.05 m of wash stone was placed in the bottom of each cell to avoid clogging and anaerobic conditions. Landscaping fabric was placed over the wash stone to prevent excessive sediment loss. Finally, a ring of bentonite clay was positioned along the inside perimeter of each bioretention cell to prevent preferential flow paths along the inside walls.

We chose the most common vegetation types recommended for rain gardens in the Midwestern United States: mesic native prairie plants, woody shrubs, and turfgrass (Bannerman and Considine 2003, Table 1). These three vegetation types were combined with bare soil in a complete randomized block design with three blocks and four treatments (one treatment per three bioretention cells). Per WI-DNR guidelines, all bioretention cells were filled with 6.27 m$^3$ of equal parts topsoil, sand, and compost. This bioretention soil mixture was classified as a loam texture consisting of 43 % sand, 40 % silt, and 17 % clay determined by the hydrometer method (Gee and Or 2002) after pretreatment with hydrogen peroxide (Mikutta et al. 2005). The organic matter content of the mixture was quantified at 6 % by loss on ignition (Heiri et al. 2001). To promote plant establishment, the vegetated bioretention cells were irrigated after planting, but received no irrigation in 2006, other than stormwater inputs during experimental applications. No fertilizer, pesticides, or herbicides were used throughout this study. Three
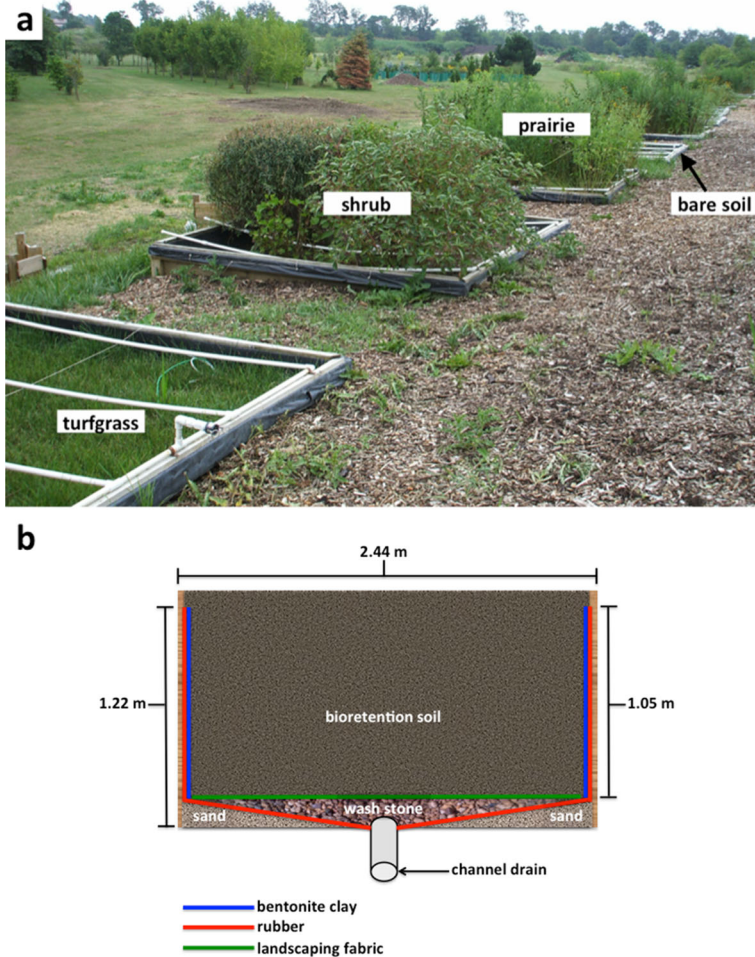
Fig. 1 (a) Photograph of 1-year old bioretention cells annotated to identify turfgrass, shrub, prairie, and bare soil treatments from bottom left to top right corner (b) Cross-sectional schematic (not to scale) of free-drainage bioretention cells (vegetation types not depicted)

event-based stormwater applications were conducted to quantify drainage, soil moisture storage, ET, and N loads among the treatments in 2006.

## Stormwater applications and data collection

We conducted three controlled stormwater applications (20 mm hr$^{-1}$) that would be of a sufficient magnitude to quantify and collect drainage. During each simulation, each bioretention cell received stormwater runoff collected from 35 m$^2$ of the contributing roof surface area (6:1 storm events) in accordance with WI-DNR guidelines that recommend sizing residential stormwater basins at 17 % of the contributing impervious area (Bannerman and Considine 2003). All three stormwater applications were conducted during the growing season of 2006 on July 15th, August 10th, and September 26th (Fig. 2). During rain events preceding

**Table 1** Species planted in each type of rain garden and initial planting density (plants per m$^{-2}$)

| Vegetation type | Scientific name | Common name | Planting density (plants m$^{-2}$) |
|---|---|---|---|
| Prairie[a] | | | 13.6 |
| | *Andropogon geradii* | Big bluestem | 0.3 |
| | *Asclepias incarnata* | Red milkweed | 0.9 |
| | *Aster novae-angliae* | New England aster | 0.9 |
| | *Baptisia bracteata* | False indigo | 0.7 |
| | *Boltonia asteroides* | False aster | 0.9 |
| | *Carex vulpinoidea* | Fox sedge | 1.7 |
| | *Echinacea pallida* | Pale purple coneflower | 0.2 |
| | *Echinacea purpurea* | Purple coneflower | 0.9 |
| | *Eupatorium perfoliatum* | Boneset | 0.9 |
| | *Helianthus occidentalis* | Ox-eye sunflower | 0.2 |
| | *Liatris pycnostachya* | Prairie blazing star | 0.9 |
| | *Monarda fistulosa* | Bergamot | 0.5 |
| | *Pancium virgatun* | Switch grass | 0.9 |
| | *Parthenium integrifolium* | Wild quinine | 0.7 |
| | *Penstemon calycosus* | Long-sepal beardtongue | 0.9 |
| | *Ratibida pinnata* | Yellow coneflower | 0.9 |
| | *Rudbeckia hirta* | Black-eyed Susan | 0.3 |
| | *Solidago rigida* | Stiff goldenrod | 0.9 |
| | *Veronicastrum virginicum* | Culver's root | 0.5 |
| Shrub[b] | | | 1.1 |
| | *Aronia melanocarpa* | Black chokeberry | 0.2 |
| | *Cornus sericea 'isanti'* | Red-twig dogwood | 0.2 |
| | *Ilex verticillata* | Winter berry | 0.2 |
| | *Salix purpurea gracilis* | Dwarf arctic willow | 0.2 |
| | *Viburnum dentatum* | Arrowwood viburnum | 0.2 |
| | *Viburnum trilobum* | American cranberry | 0.2 |
| Sod | *Poa pratensis* | Kentucky bluegrass | Complete Coverage |

[a] Prairie treatment consisting of 19 mesic species recommended by WI-DNR (Bannerman and Considine 2003) and Applied Ecological Services Inc. (Brodhead, WI)

[b] Shrub treatment consisting of 6 species recommended by Watt's Landscaping, Inc. (Verona, WI)

stormwater applications, roof runoff was collected in two large rain barrels (5,678 L) from a tin roof (417 m$^2$) on site. For the July 15th and August 10th 6:1 applications, each bioretention cell received 76 mm of stormwater, which is equivalent to a 9–15 mm rain event in Wisconsin. For the September 26th 6:1 application, the bioretention cells received 25 mm of stormwater, which is equivalent to a 3–5 mm rain event in Wisconsin. Before each controlled stormwater application, we measured initial soil volumetric moisture content ($\theta_i$) in each cell at 0.1 m increments from a depth of 0.2–0.6 m (7 depths x 2 replicates x 12 lysimeters) with a neutron hydroprobe (503 DR Hydroprobe, Campbell Pacific Nuclear). Because of the potential errors and dangers associated with using a neutron hydroprobe at the surface, we used a dielectric moisture sensor (TH2O, Dynamax Inc.) for replicated surface measurements (0–0.03 m depth) of $\theta_i$.
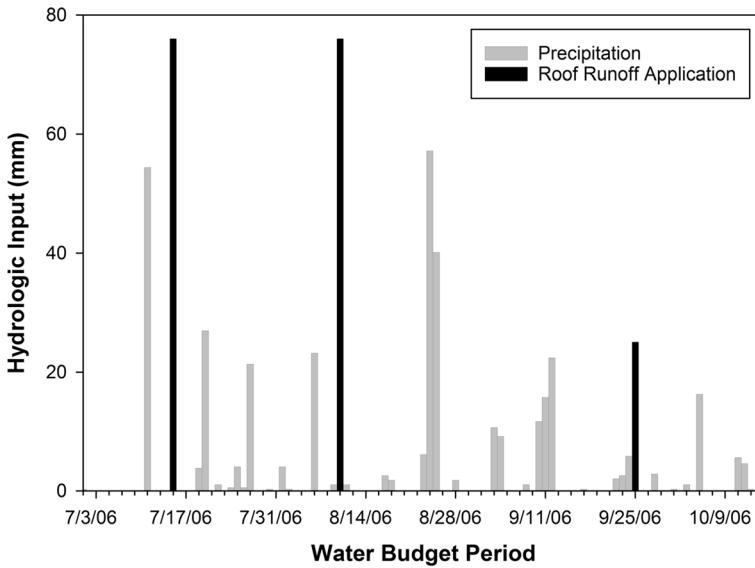
**Fig. 2** Precipitation events and 6:1 stormwater applications (mm) using collected roof runoff over the experimental time period, July 1st-October 15th, 2006 at the rain garden field site in Dane County, WI

After determining $\theta_i$ for each bioretention cell, a metered volume (Sotera 825, Tuthill Corporation) of the collected stormwater runoff was applied equally to each cell through a distribution system of PVC irrigation tubing at a rate of 20 mm hr$^{-1}$. Drainage was monitored using flow meters (Sotera 825, Tuthill Corporation) to determine the cumulative drainage at 1 hr, 24 hrs, and the designated endpoint of each stormwater application event. The designated endpoint of each application was when drainage had ceased from each cell. At the endpoint of each application, the final soil volumetric water content ($\theta_f$) was measured again at 0.1 m soil layer increments from 0.2 to 0.6 m and at the surface of each bioretention cell.

## Water budgets

Because all bioretention cells were essentially closed-system lysimeters without surface runoff, cumulative ET was quantified for the duration of each stormwater application as the residual of measured components of the water balance equation:

$$SI + P = D + \Delta S + ET \tag{1}$$

where $SI$ is stormwater input (mm) from collected roof runoff, $P$ is precipitation (mm), which was nearly negligible during stormwater applications, $D$ is cumulative drainage (mm) from each lysimeter, $\Delta S$ is the change in soil water storage (mm), and $ET$ is cumulative evapotranspiration (evaporation for bare soil, mm). We estimated $\Delta S$ for each soil layer by interpolating initial and final neutron hydroprobe and dielectric moisture sensor measurements, $\theta_i$, and $\theta_f$ (Brye et al. 2000). After quantifying the $SI$, $D$, and $\Delta S$ components of the water budget, the residual balance was attributed to $ET$ [1]. Mean daily ET rates (mm d$^{-1}$) were determined over the duration of each stormwater application for each bioretention cell.

## Potential and reference evapotranspiration

We used three potential or reference ET models (i.e. Hamon, Priestley-Taylor, and FAO Penman-Monteith) to assess the evaporative demand (mm d$^{-1}$) of the environment during the experimental time period from July 1-October 15, 2006 (Hamon 1963; Priestley and Taylor 1972; Allen et al. 1998). Potential ET models (Hamon 1963; Priestley and Taylor 1972) are indices of the evaporative demand of the environment, while the FAO reference ET model (Allen et al. 1998) is an index of the hypothetical ET from a well-watered, uniform 0.12 m grass with a constant albedo of 0.23 and surface resistance of 70 s m$^{-1}$. We chose these particular models to represent a range of meteorological complexity. Hamon is the least complex and most empirical PET model because it only requires daily temperature data to compute PET, while Priestley-Taylor requires both solar radiation and temperature data (Hamon 1963; Priestley and Taylor 1972; Lu et al. 2005). The FAO Penman-Monteith model is the most complex and least empirical index we used to calculate evaporative demand, as it requires solar radiation, vapor pressure deficit, wind speed, and temperature data to compute reference ET and empirically represents canopy physiology, which is ignored in PET models (Allen et al. 1998). We calculated Hamon and FAO Penman-Monteith estimates using local climatological data from the National Climatic Data Center (NOAA) measured at the Dane County Regional Airport, 24 km from the study site. Priestley-Taylor estimates were made using a satellite-based insolation model that assumes a constant surface albedo of 0.25 and outputs daily PET estimates for Wisconsin and Minnesota (Diak et al. 1998).

## Nitrogen retention and transport

In order to quantify soil N retention over the course of the field season, replicated soil cores were collected at 0–0.3 and 0.3–0.6 m depths from each cell once in July 2006 prior to all stormwater applications and a second time in October 2006 following all applications. Molar ratios were used to calculate the change in soil storage of inorganic nitrogen (DIN) by summing nitrogen contributions from $NO_3$ and $NH_4^+$ and taking the difference between July and October soil DIN values for each bioretention cell. To quantify plant N uptake over the course of the field season, composite plant tissue samples were collected from each vegetation type in July, August, and October 2006. Finally, we collected stormwater input and drainage samples from each bioretention cell for N analyses at 1 hr, 24 hrs, and the end of each stormwater application. All soil, plant tissue, and water samples were analyzed for total Kjeldahl nitrogen (TKN) using standard methodology (Burt 1996). All soil $NO_3^-$ and $NH_4^+$ concentrations (mg L$^{-1}$) were determined using a flow-injection analysis system (Lachat QuikChem 8000 Series, Method 12-107-04-1-F).

Measured concentration and volume of $NO_3^-$ and $NH_4^+$ from stormwater input and incremental drainage (1 hr, 24 hr, final) throughout each stormwater application were used to calculate the net mass stormwater input and gross mass output (kg ha$^{-1}$) of $NO_3^-$ and $NH_4^+$ from each bioretention cell. Net mass export or net mass import (kg ha$^{-1}$) were calculated as the difference between net mass stormwater input and gross mass output of $NO_3^-$ and $NH_4^+$ (net mass export values are positive indicating overall net N loss from the bioretention cells as drainage, net mass import values are negative indicating overall net N gain into the bioretention cells as storage). Molar ratios were used to calculate the net mass export or net mass import of dissolved inorganic nitrogen (DIN) by summing N contributions from $NO_3$ and

$NH_4^+$. Finally, we calculated N removal efficiency (%) for each stormwater application as the ratio of net mass import to net mass stormwater input of DIN for each bioretention cell (negative % efficiency values indicate an overall N contribution from the bioretention cells as drainage, positive % efficiency values indicate an overall N removal by the bioretention cells as storage).

## Statistical analyses

All data were analyzed using SAS 9.2 (SAS Institute 2008) by linear mixed models (PROC MIXED) to account for blocking and assess the statistical interaction of vegetation type (treatment) and stormwater application. Because the September stormwater application differed in both the time period and magnitude of its input (25 mm as opposed to 75 mm stormwater input from the July and August stormwater applications), we cannot isolate seasonal effects (i.e. plant phenology) from the magnitude of the stormwater application when considering overall stormwater application effects compared to vegetation type; therefore, applications were treated as event-based analyses. Each treatment consisted of three replicated bioretention cells, thus standard errors were used to evaluate differences among treatments by vegetation type, stormwater application, and the statistical interaction of vegetation type and stormwater application. Post-hoc comparisons were evaluated using Tukey's significant differences at a 0.05 probability level ($HSD_{0.05}$) to limit experiment-wise error. Log-transformations were made on all data when necessary based on the Shapiro-Wilk normality test of the residuals (PROC UNIVARIATE). All figures were created using SigmaPlot 13.0 (Systat Software 2014).

## Ecosystem service comparison

In order to rank ecosystem services by vegetation type based on the largest observed responses and compare multiple services, we normalized all means for each service analog to the highest mean for each vegetation type (Raudsepp-Hearne et al. 2010; Doherty et al. 2014). We used total drainage, ET, and $\Delta S$ as ecosystem service analogs for potential recharge, daytime evaporative cooling, and stormwater retention, respectively. Likewise, we used N removal efficiency, cumulative soil $NO_3^-$ and $NH_4^+$ storage (average of 0–0.3 and 0.3–0.6 m depths), and % TKN in plant tissue as service analogs for N load reduction, soil N retention, and plant N uptake, respectively. To enable graphical comparison across different ecosystem services, we applied linear transformations to ensure that all individual means had positive values before normalization.

## Results

Though all bioretention cells had the same internal design structure and soil mixture, there were differences in hydrology, N cycling and transport, and associated ecosystem services between treatments (vegetation type: veg) 1 year after plant establishment. Similarly, there were differences in hydrology, N cycling and transport, and associated ecosystem services corresponding to each stormwater application (app), which differed in the magnitude of stormwater applied and the external environmental conditions (i.e. seasonality). In many cases, differences in hydrology, N cycling and transport, and associated ecosystem services resulted

from the combined effect of veg and app, which was not necessarily synergistic and constituted a statistical interaction.

### Drainage and potential recharge

We found that app had a statistically significant effect (F = 67.86, p = 0.0001, S.E. = ±4 mm) on 1-hr drainage (Table 2). The September application had a mean 1-hr drainage of 3 mm or 12 % of stormwater input (SI) compared to 22 mm (29 % SI) and 21 mm (28 % SI) for July and August applications, respectively. There was a statistically significant interaction (Table 2, Fig. 3) between veg and app for drainage at 24 h (F = 7.60, p = 0.0006, S.E. = ±3 mm) and total drainage (F = 9.76, p = 0.0001, S.E. = ±3 mm) such that the main effects could not be interpreted separately. The 24-hour drainage trends were generally the same as final drainage trends; differences between vegetation treatments were less pronounced after 24 h.

We found a significant statistical interaction between veg and app for final drainage, indicating that differences in the hydrologic budget depended on both vegetation type and the timing and magnitude of stormwater applications (Table 2, Fig. 3). For the July application, prairie drained 19 mm (25 % SI), which offered significantly lower drainage or potential recharge than all other vegetation types (Table 2, Fig. 3). Bare soil, shrub, and turfgrass did not statistically differ from one another in July with total drainage values of 53, 45, and 45 mm (70, 60, and 60 % SI) respectively. Drainage differences between vegetation types were most pronounced during the August stormwater application. Prairie drained 15 mm (20 % SI) in August and was significantly different from all other vegetation types. Shrub drained 37 mm (49 % SI) in August and was significantly different from both prairie and bare soil. Turfgrass drained 48 mm (63 % SI) during the August application and was significantly different from prairie, but not shrub or bare soil. Bare soil drained 58 mm (76 % SI) during the August stormwater application, which was the largest potential recharge event observed during the course of the study. There were no statistically significant differences between vegetation types in potential recharge for the September application, which was 15, 0, 4, and 9 mm (60, 0, 16, and 36 % SI) for bare soil, prairie, shrub, and turfgrass, respectively.

### ET

There was a statistically significant interaction (Table 2, Fig. 3) between veg and app for total ET over the application time period (F = 7.52, p = 0.0006, S.E. = ±3 mm) such that differences in ET depended on both vegetation type and the application magnitude and date. The July stormwater application had the highest total ET losses and therefore, the greatest potential for daytime evaporative cooling. Over the course of the July stormwater application, prairie lost 62 mm (82 % SI) as ET, which offered significantly greater evaporative cooling than bare soil, shrub, and turfgrass, which respectively lost 26, 39, and 39 mm (34, 52, and 52 % SI) to ET and were not significantly different from one another. During the August stormwater application, prairie and shrub lost a total of 44 and 33 mm (58 and 43 % SI) as ET, respectively, and were significantly greater than turfgrass and bare soil, which lost 23 and 16 mm (30 and 21 % SI) as ET, respectively. Like potential recharge, there were no statistically significant differences between vegetation types in ET for the September application, where bare soil, prairie, shrub, and turfgrass lost 12, 21, 21, and 19 mm (48, 84, 84, and 76 % SI) as ET, respectively.

Priestley-Taylor, Hamon, and FAO-Penman Monteith models of potential or reference ET followed similar seasonal trends during the stormwater application time period, though their

**Table 2** Mean water budget components determined by a statistical mixed model where vegetation type (veg), application (app), and veg x app are designated as fixed effects with each F-ratio's numerator and denominator reported as df-num/df-den. Letter (*abcdef*) differences indicate post-hoc means comparisons evaluated by Tukey's HSD with significant veg x app interactions superseding separate post-hoc means comparisons for veg or app for cumulative drainage (D), change in storage (ΔS), and actual evapotranspiration (AET). Mean values with common letters do not differ significantly at p <0.05. AET values are compared to potential and reference evapotranspiration (PET, RET) values generated by the Priestly-Taylor, Hamon, and FAO-Penman-Monteith models for each application time period referred to as PET-Pri-Tay, PET-Hamon, and RET-FAO, respectively

| Fixed effect | Mean water budget component (mm) | | | | | Modeled potential or reference evapotranspiration (mm) | | |
|---|---|---|---|---|---|---|---|---|
| | D 1-hr | D 24-hr | D final | ΔS | AET | PET-Pri-Tay | PET-Hamon | RET-FAO |
| Vegetation type (*Veg*) | | | | | | | | |
| df-num/df-den | 3/6 | 3/6 | 3/6 | 3/6 | 3/6 | | | |
| *F*-ratio | 4.06 | 29.82 | 36.86 | 14.46 | 19.71 | | | |
| *p*-value | 0.068 | 0.0005 | 0.0003 | 0.0037 | 0.0016 | | | |
| Bare soil | 21 | 34 | 42 | 1 | 18 | | | |
| Prairie | 9 | 10 | 11 | 7 | 42 | | | |
| Shrub | 17 | 26 | 29 | 1 | 31 | | | |
| Turfgrass | 15 | 27 | 34 | 0 | 27 | | | |
| Standard error veg | 3 | 2 | 2 | 2 | 2 | | | |
| Application time period, magnitude (*App*) | | | | | | | | |
| df-num/df-den | 2/16 | 2/16 | 2/16 | 2/16 | 2/16 | | | |
| *F*-ratio | 67.86 | 233.84 | 280.69 | 77.78 | 79.63 | | | |
| *p*-value | <0.0001 | <0.0001 | <0.0001 | <0.0001 | <0.0001 | | | |
| Jul 13–19, 76 mm | 22a | 35 | 40 | −4 | 42 | 43 | 41 | 48 |
| Aug 9–14, 76 mm | 21a | 34 | 40 | 9 | 29 | 27 | 25 | 30 |
| Sep 25-Oct 2, 25 mm | 3b | 4 | 7 | 1 | 18 | 16 | 18 | 24 |
| Standard error app | 4 | 2 | 2 | 2 | 1 | | | |
| Veg x App | | | | | | | | |
| df-num/df-den | 6/16 | 6/16 | 6/16 | 6/16 | 6/16 | | | |
| *F*-ratio | 1.13 | 7.60 | 9.76 | 5.55 | 7.52 | | | |
| *p*-value | 0.3889 | 0.0006 | 0.0001 | 0.0028 | 0.0006 | | | |
| Bare soil x Jul | 29 | 45ab | 53a | −1cde | 26cde | 43 | 41 | 48 |
| Bare soil x Aug | 25 | 49a | 58a | 4bcd | 16ef | 27 | 25 | 30 |
| Bare soil x Sep | 7 | 10 cd | 15 cde | 0bcde | 12f | 16 | 18 | 24 |
| Prairie x Jul | 12 | 17c | 19c | −3de | 62a | 43 | 41 | 48 |
| Prairie x Aug | 14 | 15c | 15 cd | 19a | 44b | 27 | 25 | 30 |
| Prairie x Sep | 0 | 0d | 0e | 6bc | 21def | 16 | 18 | 24 |
| Shrub x Jul | 24 | 40ab | 45ab | −5e | 39bc | 43 | 41 | 48 |
| Shrub x Aug | 24 | 34b | 37b | 7b | 33bcd | 27 | 25 | 30 |
| Shrub x Sep | 3 | 3 cd | 4de | 1bcde | 21ef | 16 | 18 | 24 |
| Turfgrass x Jul | 22 | 38ab | 45ab | −6e | 39bc | 43 | 41 | 48 |
| Turfgrass x Aug | 23 | 40ab | 48ab | 7b | 23def | 27 | 25 | 30 |
| Turfgrass x Sep | 1 | 4 cd | 9cde | −2cde | 19def | 16 | 18 | 24 |
| Standard error veg x app | 4 | 3 | 3 | 2 | 3 | | | |

magnitudes differed on a daily basis (Table 2). Average daily ET for shrub and turfgrass was at or near the predicted ET for all three applications (Fig. 4). Prairie had higher average daily ET rates than calculated PET and RET for both the July and August stormwater applications at 9 and 7 mm d$^{-1}$, respectively, while bare soil evaporation was markedly lower than PET and RET in July and August with rates of 4 and 3 mm d$^{-1}$ (Fig. 4), respectively. All vegetation types decreased in average daily ET rates as the season progressed except shrub, which maintained nearly the same average daily ET of 6 mm d$^{-1}$ between July and August stormwater applications.

## ΔS and soil moisture

There was a statistically significant interaction (Table 2, Fig. 3) between veg and app for the change in soil moisture storage (ΔS) over the application time period (F = 5.55, $p = 0.0028$, S.E. = ±2 mm). There were no statistically significant differences in ΔS between vegetation types during the stormwater application in July. During the July application, bare soil, prairie, shrub, and turfgrass had ΔS values of −1, −3, −5, and −6 mm, respectively, indicating that all vegetation types lost soil moisture over the course of the application when seasonal evaporative demand was also at its peak. Prairie had the greatest ΔS during the August stormwater application by storing 19 mm, which was significantly different from bare soil, shrub, and turfgrass, which stored 4, 7, and 7 mm, respectively. There were no statistically significant differences in ΔS during the September stormwater application and ΔS was 0, 6, 1, and −2 mm for bare soil, prairie, shrub, and turfgrass, respectively.

We also found a statistically significant interaction between veg and app for both $\theta_i$ and $\theta_f$ at all measured depths between 0.03 and 0.6 m, with the exception of $\theta_i$ at 0.03 m (Table 3, Figs. 5, 6 and 7). The $\theta_i$ of turfgrass at 0.03 m, 0.306 m$^3$ m$^{-3}$, was significantly greater (F = 75.87, $p < 0.0001$, S.E. = ±0.013) than bare soil, prairie, and shrub treatments, which were 0.171, 0.180, and 0.184 m$^3$ m$^{-3}$, respectively. Though there was an interaction between veg and app (F = 3.36, $p = 0.0245$, S.E. = ±0.01) at 0.03 m, we found that turfgrass still retained a significantly higher $\theta_f$ at 0.03 m with values of 0.263, 0.332, and 0.281 m$^3$ m$^{-3}$ for July, August, and September applications compared to the other vegetation types, which ranged from 0.158 to 0.190 m$^3$ m$^{-3}$. Turfgrass was not significantly different from bare soil in both $\theta_i$ and $\theta_f$ at all depths greater than 0.03 m and soil moisture profiles are nearly indistinguishable from between turfgrass and bare soil at depth (Fig. 5). Prairie had a distinctively dry soil
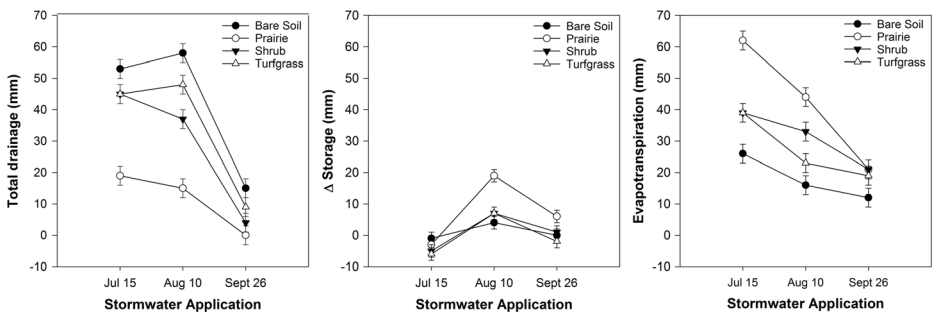


**Fig. 3** Statistical interactions between stormwater application (app; time period and magnitude) and vegetation type (veg) for rain garden water budgets including total drainage, change in storage (ΔS), and evapotranspiration (mm). Error bars represent one standard error of the mean
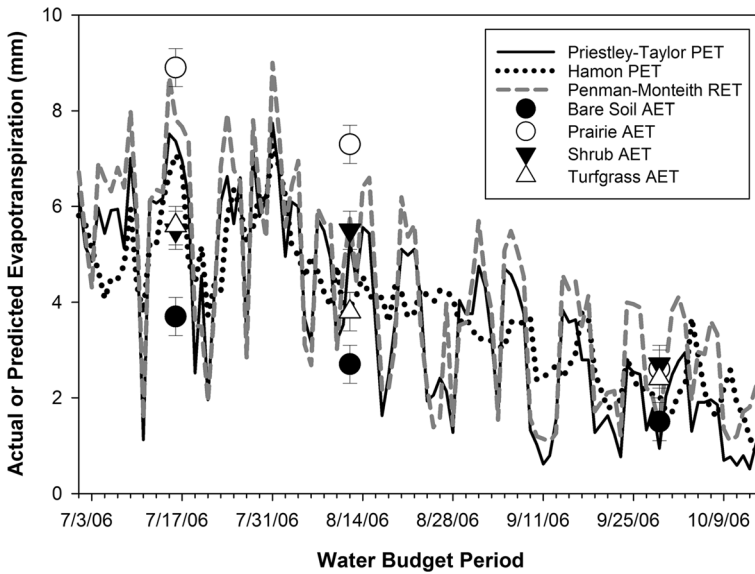
**Fig. 4** Average daily evapotranspiration (AET, mm) inferred using lysimetry method for each vegetation treatment following stormwater applications, as well as daily potential or reference evapotranspiration (PET, RET, mm) calculated using Priestley-Taylor, Hamon, and Penman-Monteith models. Error bars represent one standard error of the mean

moisture profile (Figs. 5, 6 and 7) and was almost always significantly lower than all other vegetation types between depths of 0.2–0.6 m in both antecedent and post-drainage soil moisture (Table 3). The shrub treatment had the most distinctive $\theta_i$ and $\theta_f$ profiles (Figs. 5, 6 and 7) during the August and September stormwater applications and often had significantly higher soil moisture than prairie and lower soil moisture than turfgrass or bare soil at 0.2–0.4 m depths both preceding and following stormwater applications (Table 3).

## Soil N retention, plant N uptake, and N transport

We found no statistically significant differences in soil nitrogen (%TKN, $NO_3^-$, and $NH_4^+$) pools between vegetation types. However, we found significant differences between the July and October sampling time periods (*time*) across all bioretention cells for soil $NO_3^-$ at both the 0–0.3 and 0.3–0.6 m depths and soil $NH_4^+$ at the 0.3–0.6 m depth (Table 4). In July, prior to stormwater applications, mean soil $NO_3^-$ from all bioretention cells was 2.4 and 2.6 mg $L^{-1}$ for 0–0.3 and 0.3–0.6 m depths, respectively. In October, following all stormwater applications mean soil $NO_3^-$ from all bioretention cells was 0.3 and 0.4 mg $L^{-1}$ at 0–0.3 and 0.3–0.6 m depths, respectively. These reductions in soil $NO_3^-$ content between July and October were statistically significant at both the 0–0.3 m depth (F = 156.48, $p < 0.0001$, S.E. = ±0.1) and the 0.3–0.6 m depth (F = 90.69, $p < 0.0001$, S.E. = ±0.2). In contrast, mean soil $NH_4^+$ from all bioretention cells at the 0.3–0.6 m depth increased from 0.3 mg $L^{-1}$ in July to 0.5 mg $L^{-1}$ in October. This addition of $NH_4^+$ between July and October at the 0.3–0.6 m depth was statistically significant (F = 6.77, $p = 0.0315$, S.E. = ±0.09).

**Table 3** Mean antecedent and post-drainage volumetric soil water content ($m^3\ m^{-3}$) at 0.03, 0.2, 0.3, 0.4, 0.5, and 0.6 m depths determined by a statistical mixed model where vegetation type (veg), application time period/magnitude (app), and veg x app are designated as fixed effects with degrees of freedom of each F-ratio's numerator and denominator reported as df-num/df-den. Letters *abcdef* indicate post-hoc means comparisons evaluated by Tukey's HSD with significant veg x app interactions superseding separate post-hoc means comparisons for veg or app. Mean values with common letters do not differ significantly at p <0.05

| Fixed effect | Antecedent θ at depth ($m^3\ m^{-3}$) | | | | | | Post-drainage θ at depth ($m^3\ m^{-3}$) | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 0.03 m | 0.2 m | 0.3 m | 0.4 m | 0.5 m | 0.6 m | 0.03 m | 0.2 m | 0.3 m | 0.4 m | 0.5 m | 0.6 m |
| **Vegetation type (veg)** | | | | | | | | | | | | |
| df-num/df-den | 3/6 | 3/6 | 3/6 | 3/6 | 3/6 | 3/5 | 3/6 | 3/6 | 3/6 | 3/6 | 3/6 | 3/5 |
| F-ratio | 75.87 | 42.08 | 286.27 | 74.95 | 50.77 | 12.55 | 96.69 | 24.20 | 69.27 | 35.52 | 33.72 | 14.27 |
| p-value | <0.0001 | 0.0002 | <0.0001 | <0.0001 | 0.0001 | 0.009 | <0.0001 | 0.0009 | <0.0001 | 0.0004 | 0.0004 | 0.0069 |
| Bare soil | 0.171b | 0.379 | 0.396 | 0.394 | 0.393 | 0.395 | 0.168 | 0.384 | 0.398 | 0.395 | 0.394 | 0.397 |
| Prairie | 0.180b | 0.316 | 0.335 | 0.348 | 0.349 | 0.361 | 0.172 | 0.336 | 0.356 | 0.365 | 0.361 | 0.361 |
| Shrub | 0.184b | 0.360 | 0.371 | 0.378 | 0.389 | 0.396 | 0.176 | 0.364 | 0.375 | 0.383 | 0.393 | 0.397 |
| Turfgrass | 0.306a | 0.377 | 0.390 | 0.394 | 0.400 | 0.403 | 0.292 | 0.379 | 0.394 | 0.397 | 0.404 | 0.408 |
| Standard error of veg | 0.013 | 0.006 | 0.002 | 0.002 | 0.003 | 0.006 | 0.007 | 0.005 | 0.003 | 0.003 | 0.003 | 0.006 |
| **Application time period, magnitude (app)** | | | | | | | | | | | | |
| df-num/df-den | 2/16 | 2/16 | 2/16 | 2/16 | 2/16 | 2/12 | 2/16 | 2/16 | 2/16 | 2/16 | 2/16 | 2/13 |
| F-ratio | 11.07 | 25.42 | 26.94 | 20.48 | 10.56 | 7.37 | 15.57 | 2.08 | 7.65 | 9.29 | 9.67 | 3.40 |
| p-value | 0.001 | <0.0001 | <0.0001 | <0.0001 | 0.0012 | 0.008 | 0.0002 | 0.1569 | 0.0047 | 0.0021 | 0.0018 | 0.0647 |
| Jul 13–19, 76 mm | 0.230a | 0.367 | 0.381 | 0.385 | 0.389 | 0.398 | 0.185 | 0.364 | 0.384 | 0.390 | 0.393 | 0.397 |
| Aug 9–14, 76 mm | 0.192b | 0.351 | 0.366 | 0.373 | 0.380 | 0.385 | 0.221 | 0.369 | 0.382 | 0.385 | 0.388 | 0.388 |
| Sep 25–Oct 2, 25 mm | 0.209b | 0.356 | 0.372 | 0.377 | 0.380 | 0.383 | 0.201 | 0.365 | 0.377 | 0.380 | 0.381 | 0.387 |
| Standard error of app | 0.012 | 0.005 | 0.002 | 0.002 | 0.002 | 0.003 | 0.006 | 0.003 | 0.003 | 0.002 | 0.002 | 0.004 |
| **Veg x App** | | | | | | | | | | | | |
| df-num/df-den | 6/16 | 6/16 | 6/16 | 6/16 | 6/16 | 6/12 | 6/16 | 6/16 | 6/16 | 6/16 | 6/16 | 6/13 |
| F-ratio | 1.89 | 15.74 | 16.49 | 20.63 | 8.15 | 4.42 | 3.36 | 3.38 | 10.48 | 10.57 | 7.41 | 5.12 |
| p-value | 0.1443 | <0.0001 | <0.0001 | <0.0001 | 0.0004 | 0.0138 | 0.0245 | 0.024 | <0.0001 | <0.0001 | 0.0006 | 0.0066 |

**Table 3** (continued)

| Fixed effect | Antecedent θ at depth (m³ m⁻³) | | | | | | Post-drainage θ at depth (m³ m⁻³) | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 0.03 m | 0.2 m | 0.3 m | 0.4 m | 0.5 m | 0.6 m | 0.03 m | 0.2 m | 0.3 m | 0.4 m | 0.5 m | 0.6 m |
| Bare soil x Jul | 0.170 | 0.377ab | 0.396a | 0.393ab | 0.395a | 0.396a | 0.158c | 0.376ab | 0.398a | 0.392ab | 0.394a | 0.396a |
| Bare soil x Aug | 0.160 | 0.378ab | 0.396a | 0.395ab | 0.393a | 0.394a | 0.172c | 0.389a | 0.398a | 0.397a | 0.397a | 0.401a |
| Bare soil x Sep | 0.182 | 0.382ab | 0.396a | 0.395ab | 0.390ab | 0.394ab | 0.174c | 0.386a | 0.399a | 0.395a | 0.390a | 0.395a |
| Prairie x Jul | 0.217 | 0.338c | 0.357c | 0.373 cd | 0.370b | 0.389a | 0.142c | 0.341 cd | 0.370 cd | 0.386ab | 0.383ab | 0.388a |
| Prairie x Aug | 0.157 | 0.293e | 0.312e | 0.324f | 0.336c | 0.353bc | 0.190c | 0.333d | 0.353ef | 0.360 cd | 0.355b | 0.349b |
| Prairie x Sep | 0.167 | 0.319d | 0.338d | 0.346e | 0.342c | 0.340c | 0.185c | 0.335d | 0.347f | 0.350d | 0.345b | 0.346b |
| Shrub x Jul | 0.212 | 0.377a | 0.384ab | 0.385abcd | 0.392a | 0.401a | 0.175c | 0.367abc | 0.380bc | 0.386ab | 0.393a | 0.401a |
| Shrub x Aug | 0.167 | 0.356bc | 0.368bc | 0.378bcd | 0.388ab | 0.392a | 0.189c | 0.370abc | 0.379bc | 0.387ab | 0.397a | 0.398a |
| Shrub x Sep | 0.173 | 0.346 cd | 0.360c | 0.372d | 0.387ab | 0.396a | 0.164c | 0.354bcd | 0.366de | 0.376bc | 0.388a | 0.392a |
| Turfgrass x Jul | 0.322 | 0.377ab | 0.389a | 0.390abc | 0.398a | 0.404a | 0.263b | 0.371ab | 0.388ab | 0.396a | 0.401a | 0.403a |
| Turfgrass x Aug | 0.283 | 0.377ab | 0.389a | 0.393ab | 0.401a | 0.402a | 0.332a | 0.383ab | 0.397a | 0.397a | 0.404a | 0.406a |
| Turfgrass x Sep | 0.314 | 0.379ab | 0.393a | 0.397a | 0.401a | 0.404a | 0.281b | 0.384a | 0.397a | 0.399a | 0.403a | 0.414a |
| Standard error of veg x app | 0.016 | 0.005 | 0.003 | 0.003 | 0.004 | 0.008 | 0.01 | 0.006 | 0.004 | 0.003 | 0.004 | 0.008 |
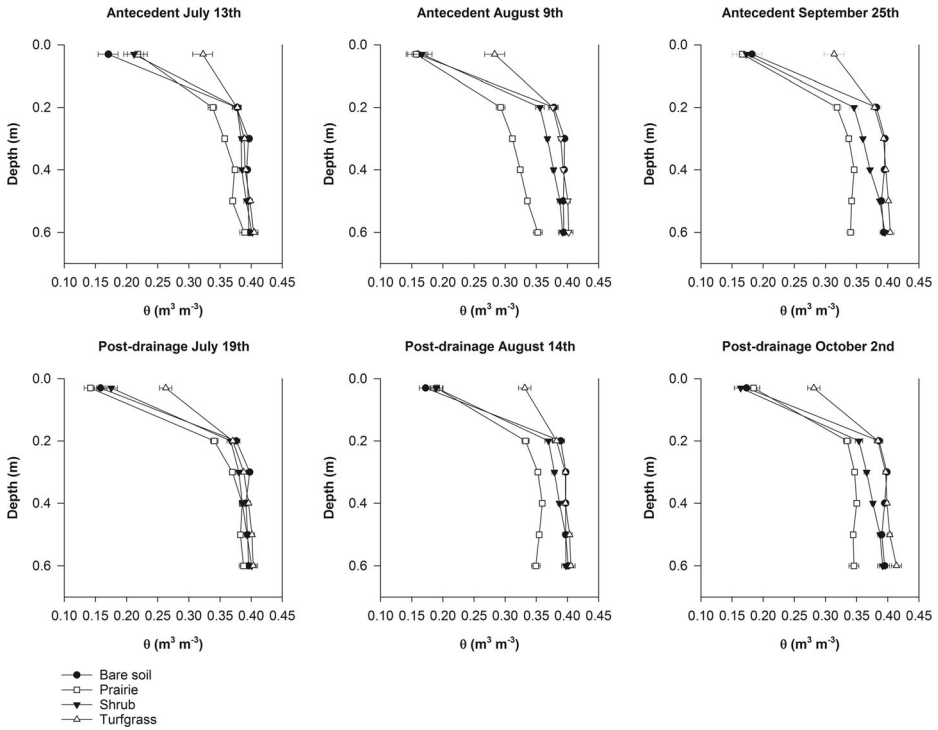
**Fig. 5** Antecedent and post-drainage soil volumetric water content (θ, m³m⁻³) of each vegetation treatment at six soil depths 0.03–0.6 m preceding and following each stormwater application. Error bars represent one standard error of the mean

We found a statistically significant interaction (Table 4) between veg and app in plant N uptake over the July, August, and October sampling periods ($F = 4.81$, $p = 0.015$, S.E. = ±0.02 mm). Turfgrass had 0.4 % TKN during all three sampling periods, which was significantly greater than prairie and shrub (0.2–0.3 % TKN).

There were no significant differences in $NO_3^-$ and $NH_4^+$ net mass input across vegetation types, however the September stormwater application received 0.04 kg ha⁻¹ of $NO_3^-$ and 0.04 kg ha⁻¹ of $NH_4^+$ in roof runoff, which was significantly less ($F = 184842$, $p < 0.0001$, S.E. = ±0.0001) than the 0.13 kg ha⁻¹ of $NO_3^-$ and 0.14 kg ha⁻¹ of $NH_4^+$ received during the July and August applications (Table 5). Bare soil leached the greatest amount of DIN observed in the study during the July and August applications, losing 0.57 and 0.96 kg ha⁻¹, respectively. Prairie was the only vegetation type that imported (negative values, Table 5) DIN for all three applications. Shrub had a net mass export (or import) of 0.34, 0.06, and −0.03 kg ha⁻¹ for July, August, and September applications, respectively. Turfgrass remained at almost the same overall N content by having a net mass export of 0.00, 0.14, and 0.00 kg ha⁻¹ for July, August, and September applications, respectively. Averaged across all applications, nitrogen removal efficiency was 72, −36, −109, and −500 % for prairie, turfgrass, shrub, and bare soil, respectively, with prairie being the only vegetation type to offer N load reduction as an ecosystem service.
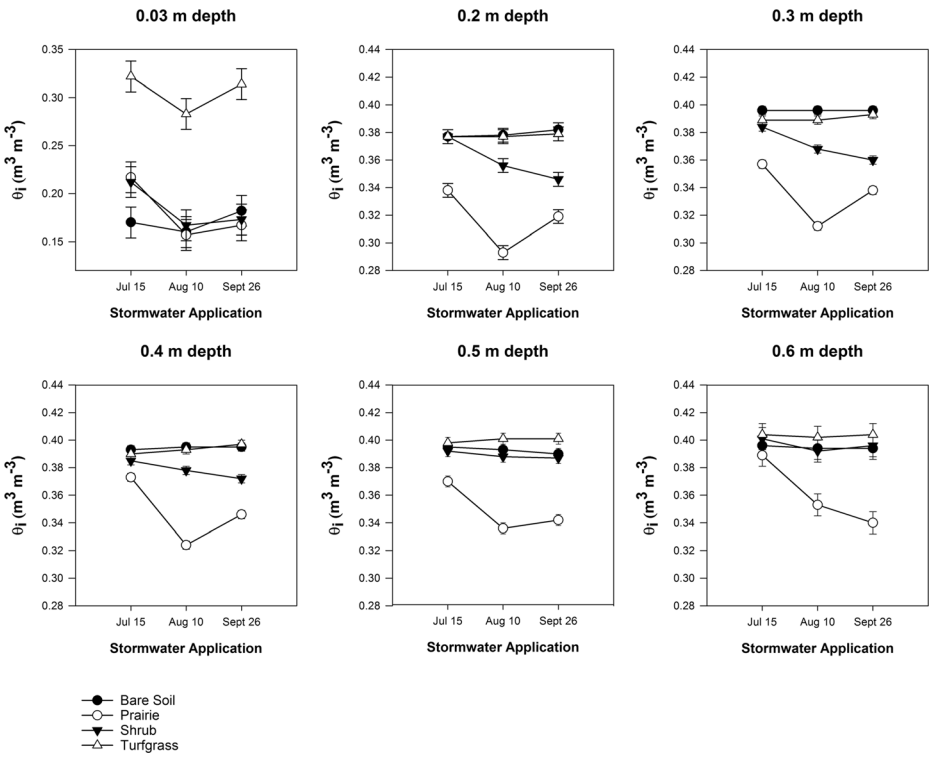
**Fig. 6** Statistical interactions between stormwater application (app; time period and magnitude) and vegetation type (veg) for antecedent volumetric water content (θ, m³m⁻³) at six soil depths between 0.03 and 0.6 m. Error bars represent one standard error of the mean

## Ecosystem services

All vegetation types differed in their provisioning of the seven ecosystem services evaluated in this study (Fig. 8). Prairie ranked first in runoff reduction, evaporative cooling, N load reduction, soil $NH_4^+$ and $NO_3^-$ retention; second (tied with shrub) for plant N uptake; and last for potential recharge services. Shrub did not rank first for providing any ecosystem service, but ranked second in runoff reduction (tied with bare soil), evaporative cooling, soil $NH_4^+$ retention, and plant N uptake (tied with prairie); third in N load reduction and potential recharge; and last in soil $NO_3^-$ retention. Turfgrass ranked first in plant N uptake; second in potential recharge and N load reduction; third in evaporative cooling, soil $NH_4^+$ retention (tied with bare soil), and $NO_3^-$ retention; and last in runoff reduction. Bare soil ranked first in potential recharge; second (tied with shrub) in runoff reduction and soil $NO_3^-$ retention, third in soil $NH_4^+$ retention (tied with turfgrass); and last in evaporative cooling, N load reduction, and plant N uptake (by default as there were no plants present).

## Discussion

Vegetation type altered the hydrological and N budgets of residential-sized rain gardens by altering drainage, ET, soil moisture, and the cycling and transport of N. Within individual
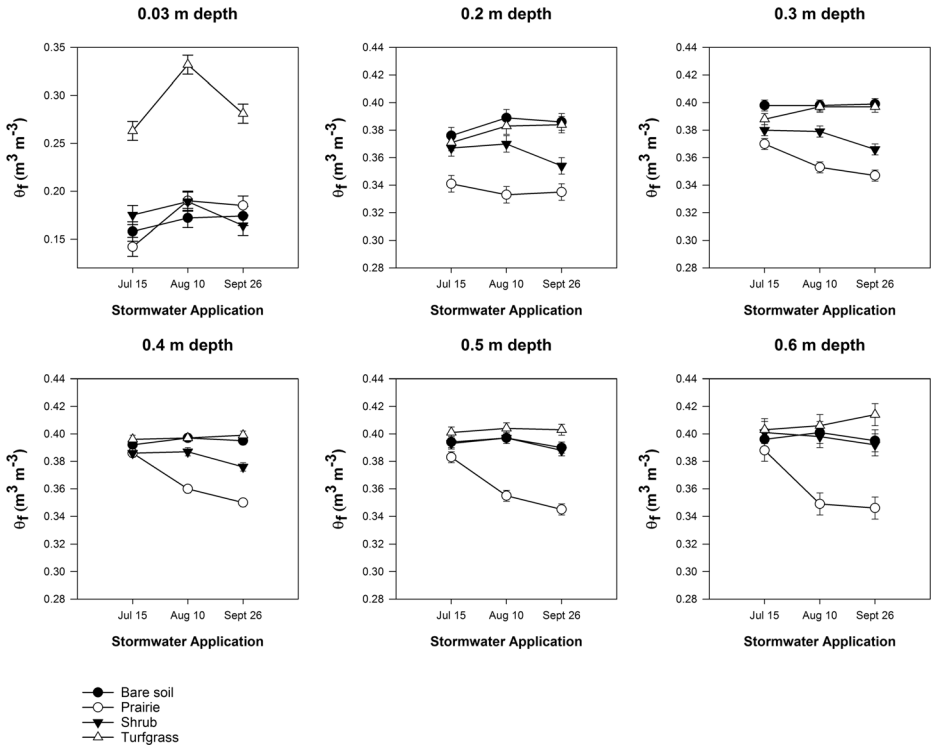
**Fig. 7** Statistical interactions between stormwater application (app; time period and magnitude) and vegetation type (veg) for post-drainage volumetric water content (θ, m³m⁻³) at six soil depths between 0.03 and 0.6 m. Error bars represent one standard error of the mean

applications, drainage and ET varied up to 2.5-fold and 4-fold, respectively, across the vegetation types assessed in this study. Additionally, differences in antecedent soil moisture preceding stormwater applications yielded up to a six-fold increase in storage across the vegetation types assessed in this study. Furthermore, rain gardens receiving the same stormwater N-load functioned as either net N exporters (via leaching) or net N importers (via retention) depending solely on their vegetation type. In this study, lower antecedent soil moisture coincided with high levels of storage, ET, and N removal, which we observed as concomitant hydrological processes in residential-sized rain gardens. Therefore, we posit that antecedent soil moisture modified by vegetation type may be an important mechanism underlying rain garden function and corresponding ecosystem services.

In this study, each vegetation type developed a characteristic antecedent soil moisture profile at depth, which contributed to its overall storage, ET, and N transport. Though antecedent soil moisture and ET were not explicitly measured, a bioretention study in Queensland, Australia demonstrated a similar relationship between antecedent dry periods, storage, and N retention where longer antecedent dry periods between storm events resulted in greater N retention (Mangangka et al. 2015). Mangangka et al. (2015) hypothesized that longer antecedent dry periods increase the storage capacity of rain gardens by drying soil pore space and reducing antecedent soil moisture prior to storm events. It has also been suggested that prior rainfall and biophysical conditions (i.e. evaporative demand) control antecedent soil moisture (Davis et al. 2011) and that vegetation type could be maximized as a rain garden

**Table 4** Mean soil nitrogen (N) content as Total Kjeldahl Nitrogen (TKN), nitrate ($NO_3^-$), and ammonium ($NH_4^+$) at 0–0.3 and 0.3–0.6 m depths and plant N content (TKN) determined by a statistical mixed model where vegetation type (veg), application (app) and veg x app are designated as fixed effects with degrees of freedom of each F-ratio's numerator and denominator reported as df-num/df-den. Letter differences (*abcd*) indicate post-hoc means comparisons evaluated by Tukey's HSD with significant veg x app interactions superseding separate post-hoc means comparisons for veg or app. Mean values with common letters do not differ significantly at $p < 0.05$. Soil N samples were collected in July and October 2006, while plant N samples were taken in July, August, and October 2006

| Fixed Effect | Soil Nitrogen 0–0.3 m | | | Soil Nitrogen 0.3–0.6 m | | | Plant Nitrogen |
|---|---|---|---|---|---|---|---|
| | TKN (%) | $NO_3^-$ (mg L$^{-1}$) | $NH_4^+$ (mg L$^{-1}$) | TKN (%) | $NO_3^-$ (mg L$^{-1}$) | $NH_4^+$ (mg L$^{-1}$) | TKN (%) |
| **Vegetation type (veg)** | | | | | | | |
| df-num/df-den | 3/6 | 3/6 | 3/6 | 3/6 | 3/6 | 3/6 | 2/4 |
| F-ratio | 3.20 | 3.30 | 3.87 | 2.20 | 1.24 | 1.42 | 35.70 |
| *p*-value | 0.1051 | 0.0992 | 0.0746 | 0.1889 | 0.3752 | 0.3251 | 0.0028 |
| Bare soil | 0.1 | 1.2 | 0.3 | 0.2 | 1.7 | 0.4 | N/A |
| Prairie | 0.2 | 0.9 | 0.4 | 0.2 | 1.1 | 0.5 | 0.3 |
| Shrub | 0.2 | 1.6 | 0.4 | 0.2 | 1.4 | 0.4 | 0.3 |
| Turfgrass | 0.3 | 1.6 | 0.4 | 0.2 | 1.7 | 0.4 | 0.4 |
| Standard error veg | 0.04 | 0.2 | 0.04 | 0.01 | 0.3 | 0.08 | 0.02 |
| **Application time period, magnitude (App)** | | | | | | | |
| df-num/df-den | 1/8 | 1/8 | 1/8 | 1/8 | 1/8 | 1/8 | 2/12 |
| F-ratio | 0.75 | 156.48 | 0.44 | 0.25 | 90.69 | 6.77 | 9.88 |
| *p*-value | 0.4117 | <0.0001 | 0.5278 | 0.6305 | <0.0001 | 0.0315 | 0.0029 |
| July 2006 | 0.2 | 2.4a | 0.3 | 0.2 | 2.6a | 0.3b | 0.3 |
| August 2006 | N/A | N/A | N/A | N/A | N/A | N/A | 0.3 |
| October 2006 | 0.2 | 0.3b | 0.4 | 0.2 | 0.4b | 0.5a | 0.4 |
| Standard error time | 0.04 | 0.1 | 0.04 | 0.01 | 0.2 | 0.09 | 0.01 |
| **Veg x App** | | | | | | | |
| df-num/df-den | 3/8 | 3/8 | 3/8 | 3/8 | 3/8 | 3/8 | 4/12 |
| F-ratio | 0.08 | 3.76 | 0.54 | 0.25 | 1.56 | 0.05 | 4.81 |
| *p*-value | 0.9672 | 0.0596 | 0.6669 | 0.8592 | 0.2727 | 0.9834 | 0.0151 |
| Bare soil x Jul | 0.1 | 2.2ab | 0.3 | 0.2 | 3.1 | 0.2 | N/A |
| Bare soil x Oct | 0.1 | 0.3c | 0.2 | 0.2 | 0.4 | 0.5 | N/A |
| Prairie x Jul | 0.1 | 1.6bc | 0.3 | 0.2 | 1.8 | 0.3 | 0.3bcd |
| Prairie x Aug | N/A | N/A | N/A | N/A | N/A | N/A | 0.2 cd |
| Prairie x Oct | 0.2 | 0.3c | 0.5 | 0.2 | 0.4 | 0.6 | 0.3ab |
| Shrub x Jul | 0.2 | 3.0a | 0.4 | 0.2 | 2.6 | 0.3 | 0.2d |
| Shrub x Aug | N/A | N/A | N/A | N/A | N/A | N/A | 0.3bcd |
| Shrub x Oct | 0.2 | 0.2c | 0.5 | 0.2 | 0.2 | 0.5 | 0.3bc |
| Turfgrass x Jul | 0.2 | 2.7ab | 0.4 | 0.2 | 2.9 | 0.3 | 0.4a |
| Turfgrass x Aug | N/A | N/A | N/A | N/A | N/A | N/A | 0.4a |
| Turfgrass x Oct | 0.3 | 0.4c | 0.4 | 0.2 | 0.5 | 0.5 | 0.4a |
| Standard error veg x time | 0.05 | 0.3 | 0.08 | 0.02 | 0.4 | 0.12 | 0.02 |

**Table 5** Net mass input (kg ha$^{-1}$), 1-hour drainage concentration (mg L$^{-1}$), 24-hour drainage concentration (mg L$^{-1}$), and net mass export (kg ha$^{-1}$) of NO$_3^-$, NH$_4^+$, and Dissolved Inorganic Nitrogen (DIN), determined by a statistical mixed model where vegetation type (veg), application time period/magnitude (app), and veg x app are designated as fixed effects with degrees of freedom each F-ratio's numerator and denominator reported as df-num/df-den. Letters *abcd* indicate post-hoc means comparisons evaluated by Tukey's HSD with significant veg x app interactions superseding separate post-hoc means comparisons for veg or app. Mean values with common letters do not differ significantly at p <0.05

| Fixed effect | Net mass input (kg ha$^{-1}$) | | | 1-hr concentration (mg L$^{-1}$) | | 24-hr concentration (mg L$^{-1}$) | | Net mass export or import (kg ha$^{-1}$) | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | NO$_3^-$ | NH$_4^+$ | DIN | NO$_3^-$ | NH$_4^+$ | NO$_3^-$ | NH$_4^+$ | NO$_3^-$ | NH$_4^+$ | DIN |
| **Vegetation type (veg)** | | | | | | | | | | |
| df-num/df-den | 3/6 | 3/6 | 3/6 | 3/6 | 3/6 | 3/6 | 3/6 | 3/6 | 3/6 | 3/6 |
| F-ratio | 0.64 | 0.63 | 0.60 | 75.98 | 7.12 | 9.57 | 1.35 | 46.73 | 3.22 | 56.60 |
| p-value | 0.6163 | 0.6221 | 0.6383 | <0.0001 | 0.0210 | 0.0105 | 0.34 | 0.0001 | 0.1035 | <0.0001 |
| Bare soil | 0.10 | 0.11 | 0.11 | 9.61 | 0.18 | 2.64 | 0.14 | 2.63 | −0.06 | 0.55 |
| Prairie | 0.10 | 0.11 | 0.11 | 0.57 | 0.20 | 1.23 | 0.29 | −0.03 | −0.10 | −0.08 |
| Shrub | 0.10 | 0.11 | 0.11 | 1.89 | 0.15 | 3.14 | 0.26 | 0.74 | −0.06 | 0.12 |
| Turfgrass | 0.10 | 0.11 | 0.11 | 2.86 | 0.36 | 0.75 | 0.12 | 0.41 | −0.06 | 0.04 |
| Standard error veg | 0.0001 | 0.0002 | 0.0002 | 0.54 | 0.04 | 0.44 | 0.08 | 0.22 | 0.01 | 0.05 |
| **Application time period, magnitude (app)** | | | | | | | | | | |
| df-num/df-den | 2/16 | 2/16 | 2/16 | 2/15 | 2/15 | 2/14 | 2/14 | 2/14 | 2/14 | 2/14 |
| F-ratio | 184842 | 184218 | 188687 | 7.90 | 10.86 | 1.13 | 9.07 | 23.19 | 15.00 | 15.12 |
| p-value | <0.0001 | <0.0001 | <0.0001 | 0.0045 | 0.0012 | 0.3513 | 0.003 | <0.001 | 0.0003 | 0.0003 |
| Jul 13–19, 76 mm | 0.13a | 0.14a | 0.14a | 3.11 | 0.23 | 2.28 | 0.10 | 1.18 | −0.07b | 0.21 |
| Aug 9–14, 76 mm | 0.13a | 0.14a | 0.14a | 5.22 | 0.13 | 1.83 | 0.14 | 1.48 | −0.09b | 0.26 |
| Sep 25–Oct 2, 25 mm | 0.04b | 0.04b | 0.05b | 2.87 | 0.30 | 1.72 | 0.36 | 0.15 | −0.03a | 0.01 |
| Standard error app | 0.0001 | 0.0001 | 0.0001 | 0.50 | 0.04 | 0.34 | 0.06 | 0.17 | 0.01 | 0.04 |
| **veg x app** | | | | | | | | | | |
| df-num/df-den | 6/16 | 6/16 | 6/16 | 6/15 | 6/15 | 6/14 | 6/14 | 6/14 | 6/14 | 6/14 |
| F-ratio | 1.07 | 1.05 | 1.07 | 6.37 | 4.16 | 4.04 | 3.68 | 13.44 | 2.27 | 13 |
| p-value | 0.4225 | 0.4332 | 0.4200 | 0.0017 | 0.0116 | 0.0148 | 0.02 | <0.0001 | 0.0964 | <0.0001 |

**Table 5** (continued)

| Fixed effect | Net mass input (kg ha$^{-1}$) | | | 1-hr concentration (mg L$^{-1}$) | | 24-hr concentration (mg L$^{-1}$) | | Net mass export or import (kg ha$^{-1}$) | | |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| | NO$_3^-$ | NH$_4^+$ | DIN | NO$_3^-$ | NH$_4^+$ | NO$_3^-$ | NH$_4^+$ | NO$_3^-$ | NH$_4^+$ | DIN |
| Bare soil x Jul | 0.13 | 0.14 | 0.14 | 8.25b | 0.12b | 2.55ab | 0.04b | 2.87b | −0.10 | 0.57b |
| Bare soil x Aug | 0.13 | 0.14 | 0.14 | 14.51a | 0.18b | 3.24ab | 0.12ab | 4.45a | −0.06 | 0.96a |
| Bare soil x Sep | 0.04 | 0.04 | 0.05 | 6.07bc | 0.23b | 2.14b | 0.27ab | 0.58 cd | −0.01 | 0.12 cd |
| Prairie x Jul | 0.13 | 0.14 | 0.14 | 0.50d | 0.27b | 0.81b | 0.04b | −0.006d | −0.11 | −0.09d |
| Prairie x Aug | 0.13 | 0.14 | 0.14 | 0.39d | 0.06b | 1.30b | 0.11b | −0.05d | −0.13 | −0.12d |
| Prairie x Sep | 0.04 | 0.04 | 0.05 | 0.81d | 0.27ab | 1.60ab | 0.72a | −0.05 cd | −0.05 | −0.05 cd |
| Shrub x Jul | 0.13 | 0.15 | 0.14 | 2.62 cd | 0.24b | 5.25a | 0.28ab | 1.63bc | −0.04 | 0.34bc |
| Shrub x Aug | 0.13 | 0.14 | 0.14 | 1.80 cd | 0.08b | 2.16b | 0.32ab | 0.57 cd | −0.09 | 0.06 cd |
| Shrub x Sep | 0.04 | 0.04 | 0.05 | 1.26d | 0.12b | 2.01b | 0.17ab | 0.006d | −0.04 | −0.03d |
| Turfgrass x Jul | 0.13 | 0.14 | 0.14 | 1.07d | 0.27b | 0.53b | 0.06ab | 0.23 cd | −0.07 | 0.00d |
| Turfgrass x Aug | 0.13 | 0.14 | 0.14 | 4.18bcd | 0.22b | 0.61b | 0.02b | 0.94 cd | −0.09 | 0.14 cd |
| Turfgrass x Sep | 0.04 | 0.04 | 0.05 | 3.33 cd | 0.59a | 1.12b | 0.27ab | 0.07d | −0.02 | 0.00d |
| Standard error veg x app | 0.0002 | 0.0002 | 0.0003 | 1.06 | 0.07 | 0.95 | 0.15 | 0.46 | 0.03 | 0.10 |
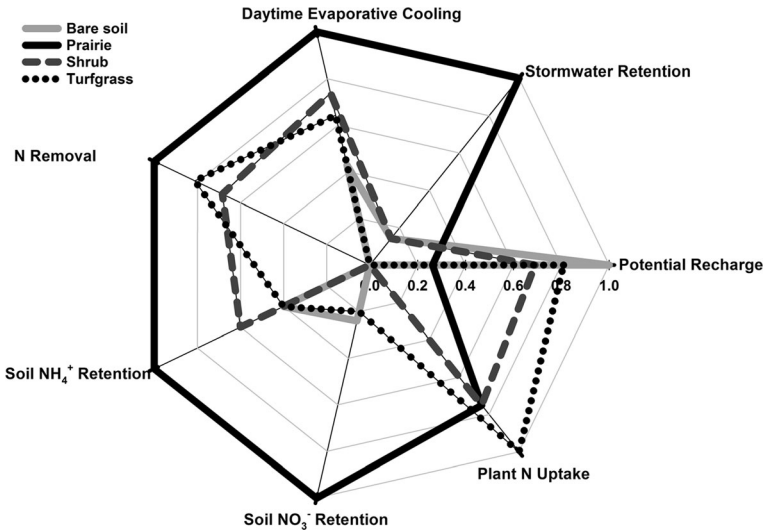
**Fig. 8** Relative provisioning of seven ecosystem services from four vegetation treatments based on analogous measured parameters. Total drainage, evapotranspiration, and the change in storage were used as ecosystem service analogs for groundwater recharge, daytime evaporative cooling, and stormwater retention, respectively. N removal efficiency, soil $NO_3^-$ and $NH_4^+$ storage, and % total Kjeldahl nitrogen in plant tissue were used as ecosystem service analogs for N removal, soil N retention, and plant N uptake, respectively. All data were normalized to the vegetation treatment that had the maximum mean for each measured parameter. Spatial arrangement of ecosystem services is arbitrary, thus polygon shapes are nonunique to these normalized data. Linear interpolation between and across ecosystem services is inappropriate in this context

design parameter to draw down antecedent moisture and increase storage capacity during dry periods (Mangangka et al. 2015).

We further develop these hypotheses using the differences in ET and antecedent soil moisture observed in this study to propose the following hydrological mechanism: rain gardens lose stored soil moisture to ET during dry periods between storm events. Dry-period ET consistently lowers soil moisture at depths accessible to plant root systems. During and immediately following storm events (when drainage is still occurring), rain gardens with large dry-period ET losses (having depleted water throughout the soil profile) also have a higher storage capacity for incoming stormwater, which leads to increased stormwater storage, N retention, decreased drainage, and large wet-period ET losses. We speculate that different vegetation types will differ in their partitioning of these concomitant hydrological processes based on their ET rates during both dry and wet periods. Though it has been suggested that wet-period ET may not be an important component of the water budget during and immediately following storm events (He and Davis 2010), we found it to be a significant component of the water budgets in this experiment, depending on vegetation type. To the best of our knowledge, this is the first controlled, replicated study to demonstrate that when exposed to the same rainfall patterns and evaporative demand, certain vegetation types, such as prairie, will decrease the antecedent soil moisture and increase the storage capacity of rain gardens compared to other vegetation types, such as shrubs and turfgrass.

In this study, vegetation-induced changes to the antecedent soil moisture of rain gardens drove differences in N retention and transport (primarily as $NO_3^-$), as opposed to biofiltration processes (e.g. plant N uptake, soil N retention, denitrification), which would have been

observed as a dilution in the concentration of N in drainage relative to stormwater inputs. We acknowledge that input N concentrations in this study were low, ranging from 0.16 to 0.18 mg $L^{-1}$ $NO_3^-$, compared to the higher range of $NO_3^-$ (0.23–1.42 mg $L^{-1}$) used in previous studies investigating stormwater N reduction (Houdeshel et al. 2015; Taylor et al. 2005; Hunt et al. 2006; Dietz and Clausen 2005; Sharkey 2006). Additionally, it has been demonstrated in unsaturated rain gardens that stormwater input N exhibits a positive linear relationship with drainage N concentration and that the slope of this relationship may be modified by vegetation type (Houdeshel et al. 2015). Though adding a saturated zone to promote denitrification may significantly dampen or remove this linearity (Kim et al. 2003; Yang et al. 2013; Gilchrist et al. 2013; Houdeshel et al. 2015), saturated rain gardens have been shown to emit higher levels of methane (McPhillips and Walter 2015) and often drain slower than their unsaturated counterparts (Yang et al. 2013). Therefore, in urban areas receiving high stormwater runoff rates and volumes or in localized hot spots where additional greenhouse gas emissions are a concern, it may be preferable to construct high-infiltration, unsaturated rain gardens planted with vegetation that supports high ET rates. Based on our findings, we contend that N transport from unsaturated rain gardens will always be contingent on storage capacity, which may be modified by vegetation type via antecedent soil moisture.

An increased understanding of the timing and magnitude of plant N uptake and mineralization in different vegetation types may provide better control of biofiltration processes in unsaturated rain gardens where denitrification is not a treatment option. Because of the low stormwater N concentration, as well as the nascent growth of vegetation in this study, observed net mass export of N may be indicative of the leaching potential of rain gardens with different vegetation types. It has been established that plant detritus can be mineralized into $NO_3^-$ and leached from rain gardens (Li and Davis 2014), which likely explains why drainage N concentration was often greater than input N concentration in this study. Considering these results, stormwater practitioners may expect newly established rain gardens to leach N even when incoming stormwater N concentration is low. Much progress has been made identifying regional plant species that reduce N loads via N uptake processes in rain gardens (Hunt et al. 2015; Read et al. 2008, 2009). For control of N transport from unsaturated rain gardens, we propose that further regional work is needed to identify specific plants or plant communities that maximize the combination of high N uptake, low N leaching, high ET, and thus greater storage capacity.

Though seasonal partitioning of ET and drainage is inevitable because of evaporative demand, our findings suggest that vegetation type is a rain garden design parameter that may be manipulated to diminish or amplify seasonal tradeoffs between ET and drainage. Similar to other studies (Denich and Bradford 2010; Hickman et al. 2011; Wadzuk et al. 2014), we observed the highest ET from rain gardens during peak evaporative demand in July, though July and August stormwater applications had the same stormwater input rate and magnitude. During peak evaporative demand in this study, rain gardens planted with prairie partitioned 80 % of incoming stormwater to ET, while gardens planted with shrub and turfgrass partitioned only 50 % of incoming stormwater to ET indicating that vegetation type can mediate the hydrological tradeoff between ET and drainage and corresponding ecosystem service tradeoff between evaporative cooling and groundwater recharge. Though maximizing ET from rain gardens and other stormwater infrastructure has been suggested as an urban heat island (UHI) mitigation practice (Endreny 2008; Clark et al. 2010), daytime ET cooling may not necessarily mitigate nighttime UHI effects (Gober et al. 2009). In our Yahara Watershed, two studies using a dense array of meteorological sensors found that the Madison UHI is the

strongest during the summer months at nighttime and that the continuous high nighttime UHI could increase the potential for adverse human health impacts during heat waves (Schatz and Kucharik 2014, 2015). Though vegetated urban surfaces with high ET positively correlate to nighttime cooling because of reductions in heat capacity; this relationship is nonlinear (Gober et al. 2009) and spatially-dependent (Mitchell et al. 2008), thus we cannot extrapolate the vegetation-mediated differences in ET from this study to translate to nighttime cooling outcomes. Additional research is needed to understand the contribution of rain gardens to nighttime cooling and whether manipulating vegetation type or spatial arrangement could make rain gardens a viable UHI mitigation strategy.

In residential-sized rain gardens, vegetation selection supported concomitant (storage, evapotranspiration, and N transport) and antithetical hydrological processes (drainage). These processes are fundamentally linked to ecosystem services that can occur from green infrastructure in bundles (stormwater retention, daytime evaporative cooling, and N removal) and tradeoffs (potential recharge) (Raudsepp-Hearne et al. 2010). All ecosystem services cannot be maximized from a single vegetation type and stormwater practitioners will have to select rain garden vegetation that addresses site-specific problems. Though this is the first replicated study to identify vegetation-driven bundles and tradeoffs from small-scale rain gardens, previous green infrastructure studies have identified bundles and tradeoffs arising from differences in design parameters. A study comparing four mesoscale (400–1410 m$^2$) saturated and unsaturated bioretention basins in Ithaca, New York identified a tradeoff between N-removal (via denitrification) and greenhouse gas emissions arising from saturated, slow-draining conditions in bioretention basins (McPhillips and Walter 2015). Likewise, a study comparing three mesoscale (1100 m$^2$) bioretention swales in the Midwestern U.S. with different clay layer thicknesses identified a tradeoff between drainage (which resulted in a five-service hydrological bundle) and net primary productivity (Doherty et al. 2014). A consistent idea emerging from our study and others (Doherty et al. 2014; McPhillips and Walter 2015) is that to some degree, ecosystem service bundles and tradeoffs will always result from differences in green infrastructure design such that all services cannot be maximized from a single archetype.

In order to optimize the versatility of ecosystem services in urban ecosystems, we recommend incorporating vegetation type in biogeochemical models of green infrastructure used for predicting water quantity and quality in future climate, land-use, and socioeconomic scenarios. Validated models representing the bundles and tradeoffs associated with vegetation selection are critical to assess how small, spatially-distributed, green infrastructure practices perform across different scales including individual parcels, residential developments, and entire urban landscapes (Potter 2006). Though biogeochemical models have been developed to simulate various flow and soil textural configurations (Dussaillant et al. 2004; He and Davis 2010; Brown et al. 2013), no current models of green infrastructure practices explicitly represent different vegetation types and their associated biogeochemical processes and impacts. Pataki et al. 2011 provides an experimental framework for assessing ecosystem services, disservices, and net efficacy of green infrastructure practices using biogeochemical methods. Although our study used this framework to identify bundles and tradeoffs in ecosystem services from small-scale rain gardens during the growing season, we did not quantify the effects of spatial arrangement and interannual climate variability on these outcomes, as differences should occur across space and time (Raudsepp-Hearne et al. 2010). Additionally, our study quantified the differences between nascent vegetation, the effects of which will likely increase over time via

changes in soil structure and hydraulic connectivity (Johnston et al. 2016; Angers and Caron 1998). Therefore, we suggest future work should incorporate observed processes into modeled urban scenarios, while conducting additional controlled, replicated studies to isolate long-term effects and feedbacks between vegetation type, soil texture, and other key design elements that may alter biogeochemical cycles in green infrastructure.

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