A FIELD EVALUATION OF RAIN GARDEN FLOW AND POLLUTANT TREATMENT

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Abstract. Rain gardens have been recommended as a best management practice to treat stormwater runoff. However, no published field performance data existed on pollutant removal capabilities. Replicated rain gardens were constructed in Haddam, CT, to capture shingled-roof runoff. The gardens were sized to store the first 2.54 cm (1 inch) of runoff. Influent, overflow and percolate flow were measured using tipping buckets and sampled passively. Precipitation was also measured and sampled for quality. All weekly composite water samples were analyzed for total phosphorus (TP), total Kjeldahl nitrogen (TKN), ammonia-nitrogen (NH_3-N), and nitrite+nitrate-nitrogen (NO_3-N). Monthly composite samples were analyzed for copper (Cu), lead (Pb) and zinc (Zn). Redox potential was measured using platinum electrodes. Poor treatment of NO_3-N, TKN, organic-N, and TP in roof runoff was observed. Many Cu, Pb and Zn samples were below detection limit, so statistical analysis was not performed on these pollutants. The only pollutants significantly lower in the effluent than in the influent were NH_3-N in both gardens and total-N in one garden. The design used for these rain gardens worked well for overall flow retention, but had little impact pollutant concentrations in percolate. These results suggest that if an underdrain is not connected to the stormwater system, high flow and pollutant retention could be achieved with the 2.54 cm design method.

Keywords: bioretention, nutrients, rain garden, roof runoff, stormwater, urban

1. Introduction

Urban areas contain large amounts of impervious coverage that can range from 20% in residential areas to as much as 85% in commercial areas (Novotny and Olem, 1994). Increases in peak runoff velocity and decreases in lag time due to urbanization were noted by Leopold (1968). Although several studies in the 1960s (e.g., Waananen, 1969) documented increased runoff from urban areas, this phenomenon was reported and measured more than 100 years ago (Kuichling, 1889). Urbanization also has been found to decrease low flows in streams as a result of less recharge of groundwater from precipitation (Ferguson and Suckling, 1990). Urban areas further contribute pollutants to stormwater such as sediment, nitrogen, phosphorus and heavy metals, impairing downstream habitat and water quality (Novotny and Olem, 1994).

Rain gardens, also termed bioretention areas, have been recommended as a best management practice (BMP) to reduce nonpoint source pollution from urban areas (US EPA, 2000; Prince George's County, 1993). Rain gardens are shallow depressions in the landscape that are planted with trees and/or shrubs, and covered with a bark mulch layer or ground cover. They allow stormwater to infiltrate, recharge aquifers, and reduce peak flows. In addition, they are expected to provide pollutant treatment, which has been attributed to several processes including adsorption, decomposition, ion exchange, and volatilization (Prince George's County, 1993).

While the use of rain gardens is increasing nationally, there is a surprising lack of knowledge about their retention and infiltration abilities. Prince George's County, Maryland was the first to recommend the use of a rain garden as a stormwater treatment device with the publication of the Bioretention Design Manual (Prince George's County, 1993). Since then, numerous rain garden publications and fact sheets have become available (e.g., City of Lenexa, 2003; Hunt and White, 2001; WI DNR, 2003). There is no clear consensus on design questions such as how to size the gardens and the soil matrix to use. In addition, sizing the gardens becomes more complex as the variety of land cover types within the watershed increases. The first Bioretention Manual (Prince George's County, 1993) offered two different sizing methods. The first was based on storing the first one-half inch of runoff; the second involved a calculation using the rational "C" coefficient for the watershed times 5–7% of the watershed area. The "C" coefficient is chosen based on the land use in the watershed, and examples can be found in Novotny and Olem (1994). The Natural Resources Conservation Service curve number (SCS, 1986) is used in two publications to determine the appropriate size for a rain garden based on the amount of runoff expected (Hunt and White, 2001; Prince George's County, 2002). The City of Lenexa, KS (2003) provides worksheets and graphs for sizing, but does not provide information on how the calculations were derived. The sizing method recommended by Wisconsin Department of Natural Resources (WI DNR, 2003) is based on soil type and distance from the building, and is specific to rain gardens near residential roof areas.

An underdrain may be installed beneath a rain garden and connected to a traditional stormwater system or daylight to facilitate drainage when native soils are not sufficient to infiltrate water quickly enough. Yet, an underdrain is not consistently recommended. An underdrain is not recommended by the WI DNR (2003). The use of an underdrain allows a portion of percolated stormwater to enter the traditional stormwater system. Peak flow rates and pollutant concentrations may be reduced, but depending on the local soil conditions and rainfall pattern, a large volume of water could still be discharged to local waterways. The impact of the underdrain on overall pollutant retention in a rain garden has not been investigated.

The purposes of encouraging infiltration are to maximize treatment and avoid standing water, which would encourage mosquito growth. The earlier manual from Prince George's County (1993) recommended a ponding time of less than four days, however, this recommendation was changed to less than three to four hours in the newer version of the manual (Prince George's County, 2002). Interestingly, the City of Lenexa (1993) includes an option for a permanent pool in their worksheet, and they state that some people introduce tadpoles or dragonflies into them. This recommendation seems to differ greatly from the majority of the other sources, as most seem to suggest not having standing water in the garden. Although the methods and recommendations vary from simple to complex, none has been substantiated with field monitoring.

Davis *et al*. (2001) performed a laboratory column study of two scale-model rain gardens. They reported concentration reductions of greater than 90% for copper (Cu), lead (Pb), and zinc (Zn), 68% for total Kjeldahl-N (TKN), and 87% for ammonia-N (NH_3-N). Nitrite+nitrate-N (NO_3-N) reductions were generally low (24%), and NO_3-N outflow concentrations from the upper ports (closer to the surface) in the system were higher than inputs, indicating that NO_3-N was actually produced in the system. Application of synthetic stormwater to a field rain garden in Maryland by Davis (US EPA, 2000) resulted in slightly lower reductions for all measured pollutants. No published data exist on the effectiveness of field scale rain gardens in reducing pollutants under actual working conditions.

The objective of the present study was to evaluate the ability of a field-installed rain garden to retain pollutants and reduce runoff.

2. Methods

2.1. Construction

Two rain gardens were constructed in September 2002 in Haddam, CT, according to specifications in the most recent design manual from Prince George's County, MD (Prince George's County, 2002). Native soil at the site was Penwood (Typic Udipsaments) loamy sand (USDA, 1979). The soil was also classified as loamy sand by the Soil Testing Laboratory at the University of Connecticut. This soil was adequate to provide the rapid infiltration (3.8 cm hr^{-1}) recommended in the bioretention manual (Prince George's County, 2002). A 1,143 μm (45-mil) EPDM liner (Firestone PondGard, Unit Liner Company, Shawnee, OK) was installed under the two rain gardens (Figure 1). The liner is not normally a component of a rain garden, however, for sampling purposes, it was used here to seal the bottom and collect all percolate water. A 10.2 cm diameter perforated PVC pipe collected percolate water under each rain garden (above the liner) and was covered with 15 cm of 2.54 cm round stone. The native soil was installed in both gardens and covered with a layer of shredded hardwood bark mulch approximately 5 cm thick. The physical characteristics of the soil and mulch in the two gardens were similar (Table I). Plantings in the rain garden include three native species. Five chokeberry (*Aronia prunifolia)* shrubs, and four each of winterberry (*Ilex verticillata)* and compact inkberry (*Ilex*

TABLE I
Soil and mulch characteristics, rain gardens, Haddam, CT

	Rain garden no.		
	1	2	Mulch
Bulk density (g/cc)	1.63	1.66	0.2
Organic matter (% LOI)	1.6	1.9	–
CEC ($cmol_c$/kg)	16.8	22.7	166
pH	6.1	6.3	–
Sand (%)	84.4	83.6	–
Silt (%)	7.6	10.0	–
Clay (%)	8.0	6.4	–
Infiltration capacity (cm/hr)	12.6	10.3	–

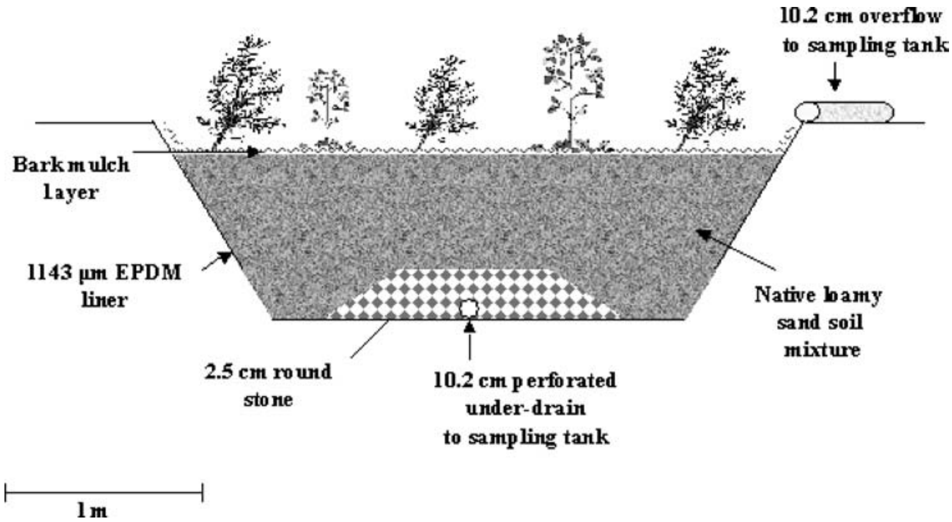

Figure 1. Cross-section of rain garden in Haddam, CT.

glabra compacta) were planted in each garden. The shrubs and the mulch were obtained from a local nursery.

The study rain garden was sized to accept the first 1.27 cm (one-half inch) of runoff from the roof. The addition of the replicate garden increased the aboveground storage capacity to the first 2.54 cm (one-inch) of runoff from 106.8 m^2 of the asphalt shingle roof. The surface dimensions of the rain gardens were 2.74 m by 3.35 m. The soil was approximately 0.6 m deep, but bottom dimensions varied slightly, so rain garden 1 contained 2.66 m^3 of soil, and rain garden 2 contained 2.33 m^3 of soil. The paired rain gardens were designed to have a combined aboveground storage depth of 15.24 cm (6 inches). Aluminum gutters and downspouts in place at the site prior to the study were used to collect roof runoff. Influent sampling was conducted in a concrete tipper distribution box; flow was measured continuously using a tipping bucket. Influent roof runoff flowed to the rain gardens through 10.2 cm slotted-pipe spreaders. In the chamber, flow from each pipe was measured using tipping buckets.

2.2. Soils

Organic matter content of the soils was determined at the beginning of the study by loss on ignition (NEC-67, 1995). Textural analysis was performed using the hydrometer method (Gee and Bauder, 1979). Both organic matter content and soil texture analysis were performed on a subsample from 10 2.54 cm diameter by 6 cm soil cores from each garden. Average bulk density was determined using the core method on 4 samples from each garden (SSSA, 2002) Average infiltration capacity was determined using a double-ring infiltrometer (SSSA, 2002). Cation exchange capacity was determined using the ammonium acetate (pH 7) method (Sumner and Miller, 1996).

2.3. Sampling

Subsurface and overflow samples were collected in a 3785-liter (1000 gal) pump chamber adjacent to the rain gardens (Figure 2). Water samples were collected passively from the tipping bucket using PVC tubing, and split. Collection bottles were prepared by washing with a phosphate-free soap, acid washing with a nitric acid solution, and triple rinsing with distilled water. One-half of the sample was collected in bottles acidified with nitric or sulfuric acid, for metals analysis and nutrient analysis, respectively. Water samples were stored at 4 °C in a refrigerator until they were collected weekly. They were then transported in a cooler to the water quality laboratory at the University of Connecticut, and stored at 4 °C until analysis. Holding times were $<$28 days for nutrient samples, and $<$6 months for metal samples (APHA, 1998).

2.4. Analysis

Weekly composite samples were analyzed for total phosphorus (TP), TKN, NH_3-N, and NO_3-N on a Lachat colorimetric flow injection system using EPA methods (US EPA, 1983a). Organic nitrogen (ON) was calculated by subtracting the NH_3-N concentration from the TKN concentration for every sample. Total nitrogen (TN) was calculated by adding TKN concentrations to NO_3-N concentrations for every sample.

Monthly composite samples were analyzed for Cu, Pb and Zn using Inductively Coupled Plasma-Mass Spectrometry (ICP-MS) method 200.7 (US EPA, 2001).

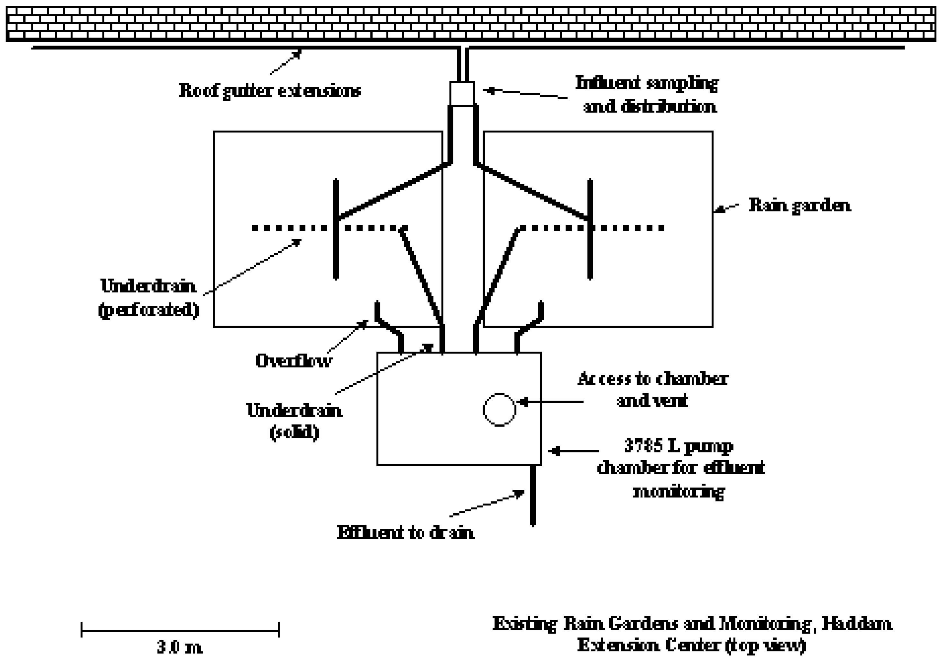

Figure 2. Aerial view of rain gardens and monitoring, Haddam, CT.

Bacteria grab samples were taken when water was flowing at a site visit and analyzed for fecal coliform bacteria using the membrane filtration method (US EPA, 1978).

2.5. PRECIPITATION

Precipitation was measured using a heated tipping-bucket rain gauge. Total weekly precipitation was also measured using a non-recording rain gauge. Monthly totals were compared to 30-year normal precipitation at the NCDC station in Groton, CT, which is approximately 38 km from the study site. Ground water level was measured in 5.1 cm screened wells in each rain garden using a pressure transducer. A Campbell Scientific CR-10 datalogger was used to record water level, flow (tips) from the inlet and under drain locations. Water level was measured every minute and a 30-minute average was recorded. Tips were counted and totaled for the same period. Tips from the overflow pipes were recorded using Onset Computer Hobo^{TM} event loggers.

A weekly composite bulk deposition sample was collected using a funnel collector similar to the throughfall sampler used by Eaton *et al.* (1973). Precipitation samples were analyzed for the same constituents as other samples.

2.6. Temperature and redox potential

Water temperature was measured using a Campbell Scientific model 107 thermistor temperature probe at the inlet and the outlet of the two underdrains, when flow was occurring. Thirty-minute averages were recorded using the CR-10 datalogger.

Redox potential was measured at two depths in each garden with platinum welded electrodes (Jensen Instuments, Inc., Tacoma, WA). Replicate shallow electrodes were installed to a depth of 28 cm, and replicate deep electrodes were installed to a depth of 55 cm in each garden. Redox potential measurements (mv) were taken weekly using a millivolt meter and an Ag/AgCl reference electrode. Direct measurements were corrected to Normal Hydrogen Electrode potential by adding +0.199 volt.

2.7. Statistical analysis

Analysis of variance (ANOVA) was used to determine if significant differences existed in measured nutrient and metals concentrations between precipitation, inlet, and underdrain. Duncan's multiple range test was used to determine significant differences among means. A value of one-half the detection limit was substituted for any analytes reported as "non-detect." The water quality data were log-transformed prior to analysis as the data were found to be log-normally distributed. The SAS™ version 9.0 statistical package was used for all analyses (SAS, 2002). Nutrient percent retention was calculated as the difference between the product of influent and effluent concentrations and flows. ANOVA was performed on the difference between weekly inlet and underdrain outflow average temperatures, to determine if a significant difference existed among seasons. Seasons were defined as follows: winter = December, January, February; spring = March, April, May; summer = June, July, August; fall = September, October, November.

3. Results

3.1. Precipitation

From December 2002 through December 2003, 167.4 cm of precipitation fell at the study site, which was 24.2% above the 30-year normal at the Groton NCDC station (NOAA, 2003).

3.2. Flow

Most of influent left the rain gardens as subsurface flow (98.8%). Only 0.8% of the inflow water overflowed during the entire study period (Table II), which included an unusually cold and snowy winter with frequent frost in the soil. Overflow

TABLE II

Flow mass balance for rain gardens, Haddam, CT. Depth values are based on total rain garden area (26.1 m^2)

	Volume (L)	Depth (cm)	% of inflow
Inflow			
Roof runoff	170,063	653	84
Precipitation	32,241	123	16
Total	202,304	776	100
Outflow			
Underdrain	199,933	767	98.8
Overflow	1,645	6	0.8
Total	201,578	774	
Residual	726	3	0.4

occurred four times, and three times, for garden 1 and garden 2, respectively, during the 56-week study period. The residual volume (0.4%) was assumed removed by evapotranspiration (ET) from the gardens.

Precipitation and flow data for one event on October 1, 2003 demonstrate the ability of the rain garden to reduce peak flow rates and increase lag time (Figure 3). The timing and shape of the inlet (roof runoff) hydrograph follow the precipitation hyetograph closely. However, the underdrain outflow shows a lower peak flow rate

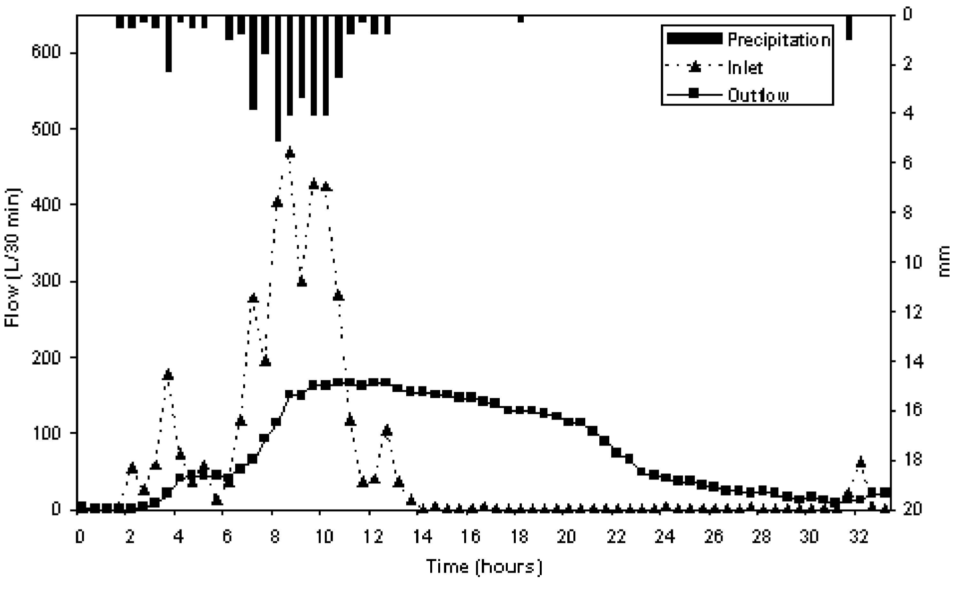

Figure 3. Precipitation, inflow, and outflow (underdrain) for one event, Haddam rain garden.

and a delayed response to the precipitation event (Figure 3). No overflow occurred during this 42 mm event.

3.3. Nitrogen and phosphorus

No significant differences in NO_3-N, TKN or ON concentrations were found among precipitation, inlet or underdrain concentrations (Table III). However, NH_3-N concentrations from the underdrains were significantly ($p = 0.001$) lower than NH_3-N concentrations in the precipitation and the inlet. NH_3-N mean concentrations were near detection limit for precipitation and inlet, and at detection limit for the underdrains (Table III). TN concentrations from the underdrain of rain garden 1 were significantly ($p = 0.05$) lower than precipitation and inlet TN concentrations; TP concentrations at the inlet were significantly ($p = 0.001$) greater than in the precipitation. Interestingly, TP concentrations in effluent from both underdrains were significantly greater than both precipitation and inlet concentrations. Although TP concentrations were higher from the underdrains than the inlet when analyzed using ANOVA, a trend can be seen when the concentrations are compared over time (Figure 4). The geometric mean TP concentration at the inlet (regression not significant) was 0.019 mg L^{-1}, whereas the average of the two underdrains showed

TABLE III

Summary of geometric means and ANOVA results for pollutants measured in rain gardens, Haddam, CT

Variable	n	DL	Unit	Bulk deposition	Roof runoff	Underdrain Garden 1	Underdrain Garden 2
NO_3-N^{ns}	47	0.2	mg L^{-1}	0.5 a	0.5 a	0.3 a	0.4 a
				±0.5	*±0.6*	*±0.4*	*±0.5*
NH_3-N***	47	0.01	mg L^{-1}	0.03 a	0.04 a	0.01 b	0.01 b
				±0.12	*±0.19*	*±0.01*	*±0.14*
TKN^{ns}	47	0.1	mg L^{-1}	0.5 a	0.7 a	0.4 a	0.6 a
				±0.7	*±0.8*	*±0.3*	*±0.4*
TN*	47	0.1	mg L^{-1}	1.2 a	1.2 a	0.8 b	1.0 ab
				±0.8	*±1.1*	*±0.6*	*±0.6*
ON^{ns}	47	0.1	mg L^{-1}	0.4 a	0.5 a	0.4 a	0.6 a
				±0.6	*±0.7*	*±0.3*	*±0.4*
TP***	47	0.005	mg L^{-1}	0.012 a	0.019 b	0.058 c	0.060 c
				±0.018	*±0.038*	*±0.036*	*±0.064*

$^{*}p = 0.05$.
$^{***}p = 0.001$.
ns = ANOVA comparison non significant.
DL = Detection limit.

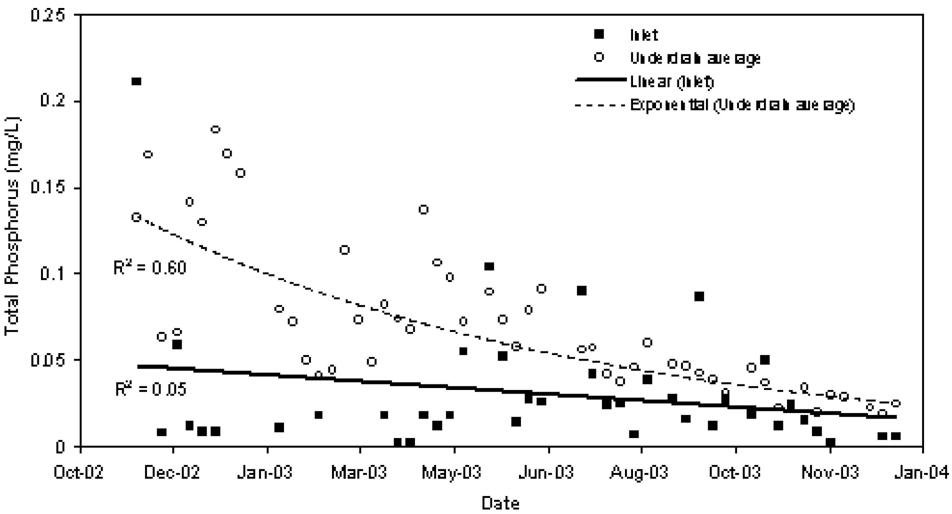

Figure 4. Rain garden TP concentrations over time, Haddam, CT.

an exponential decay trend ($R^2 = 0.6$). Total phosphorus concentrations at the inlet and the underdrains towards the end of the study period were similar. We suspect that the disturbance of the soil at the beginning of the study period was the cause of the increase in phosphorus concentrations at the outlet.

The geometric mean concentration of 0.5 mg L^{-1} for NO_3-N in both the bulk deposition samples and in the runoff from the roof (Table III) was only slightly lower than the 0.7 mg L^{-1} event mean concentration for NO_3-N reported by the Nationwide Urban Runoff Program (NURP) for runoff from urban areas (EPA, 1983b). The NURP event mean concentrations of 1.9 and 2.4 mg L^{-1} for TKN and TP, respectively (US EPA, 1983b), were higher than the mean concentrations of those pollutants found in this study (Table III). Chang and Crowley (1993) reported median NH_4-N concentrations in runoff from an asphalt composition shingle roof in Texas of 0.859 mg L^{-1}, which was not significantly different from the 0.799 mg L^{-1} NH_4-N measured in open rainfall at the site. Concentrations of NH_3-N in precipitation and roof runoff from the Haddam study site were lower than those measured in Texas (Chang and Crowley, 1993), and were not significantly different from each other.

Mass retention was less than 36% for NO_3-N, TN, TKN, and ON (Table IV). Mason *et al.* (1999) reported that NO_3-N was not well retained by gravel soils, with concentrations in percolate water following concentrations in influent water. The only nutrient that was retained well by the system was NH_3-N at 84.6%. This corresponds well with results presented by Mason *et al.* (1999), who reported that NH_4-N concentrations decreased as a function of infiltration path length through gravel soils. The mechanisms responsible for the decrease were reported to be nitrification in the soil and adsorption of NH_4-N to soil particles. For the Haddam

TABLE IV

Pollutant percent retention summary for rain gardens, Haddam, CT

	NO_3 (g)	NH_3 (g)	TKN (g)	TP (g)	TN (g)	ON (g)
Inlet	108.2	13.01	147.3	6.39	252.5	125.6
Bulk deposition	27.0	3.60	25.1	0.77	52.6	20.8
In total	135.2	16.61	172.4	7.16	305.2	146.4
Garden 1 (under)	43.6	1.18	56.6	7.78	100.3	55.5
Garden 2 (under)	43.2	1.34	61.3	7.29	105.8	59.2
Overflow	0.5	0.04	0.7	0.02	1.3	0.6
Out total	87.3	2.55	118.6	15.09	207.4	115.3
% Retention	35.4	84.6	31.2	−110.6	32.0	21.3

rain garden, adsorption is assumed to be the primary mechanism responsible for the decrease in NH_3-N concentrations, as a corresponding increase in NO_3-N concentrations, indicative of nitrification, was not found. Simultaneous nitrification and denitrification could also have caused the decrease in NH_3-N concentrations without increasing NO_3-N concentrations, however denitrification was not measured directly. Retention for TP was −110.6%, indicating that more phosphorus left the system than entered. Retention of NO_3-N and NH_3-N was slightly greater than that reported by Davis *et al.* (2001), who reported 24 and 79% removal for NO_3-N and NH_3-N, respectively in a laboratory pilot scale rain garden. However, retention of TKN and TP was lower than that presented by Davis *et al.* (2001), who found removals of 68 and 81%, for TKN and TP, respectively. When synthetic runoff was applied to a rain garden in a field setting in Maryland, similar results to those presented by Davis *et al.* (2001) were found, except for a slightly lower retention of 65% for TP (US EPA, 2000). It should be noted, however that both of the other studies presented involve the application of synthetic runoff for six hour durations, which may or may not be representative of field conditions for an actual rain garden.

Bulk total nitrogen deposition estimates of 8.3 (Yang *et al.*, 1996), 9.7 (Carley *et al.*, 2001), and 10.1 kg N ha^{-1} yr^{-1} (Nadim *et al.*, 2001) have been reported in Connecticut. These estimates are based on summing wet and dry deposition values, except for Nadim *et al.* (2001), who also measured and reported bulk deposition directly. Air samples were taken to estimate dry deposition based on particle deposition rates (Carley *et al.*, 2001; Yang *et al.*, 1996). These values are all less than the 24.6 kg $ha^{-1}yr^{-1}$ measured at the site during the study period. Bulk total phosphorus deposition reported by Yang *et al.* (1996) was 0.04 kg $ha^{-1}yr^{-1}$, which is less than the 0.3 kg ha^{-1} P measured at the study site. The study site had several trees within 15 m, and although gross particles were filtered from the bulk deposition sampler, pollen was observed on the sampling funnel during spring months. This added mass could have increased the concentrations of TN and TP measured at the site.

3.4. Metals

Since 64% of all monthly metals samples were below detection limits (5, 5, and 10 μg L^{-1} for Cu, Pb, and Zn, respectively), ANOVA and mass retention were not calculated for metals.

3.5. Bacteria

Fecal coliform concentrations for the inlet and both underdrains were <10 FCU 100 mL^{-1} for each of six events sampled. Fecal coliform bacteria concentrations in the roof runoff (inlet) were less than the geometric mean of 294 FCU 100 mL^{-1} measured by Bannerman *et al.* (1993) in runoff from residential roofs.

3.6. Temperature

Water temperatures in roof runoff and rain garden effluent followed each other closely through the year (Figure 5). Seasonal means are presented in Table V. ANOVA results on the difference between roof runoff temperature and underdrain outflow temperature indicated no significant differences among seasons. Although the gardens warmed roof runoff more during the fall and winter, the average difference for all seasons except spring was negative (Table V). The lack of a cooling effect in the summer was surprising and may be due to the relatively shallow (0.6 m) soil depth and the rapid infiltration of water through the system. In addition, the

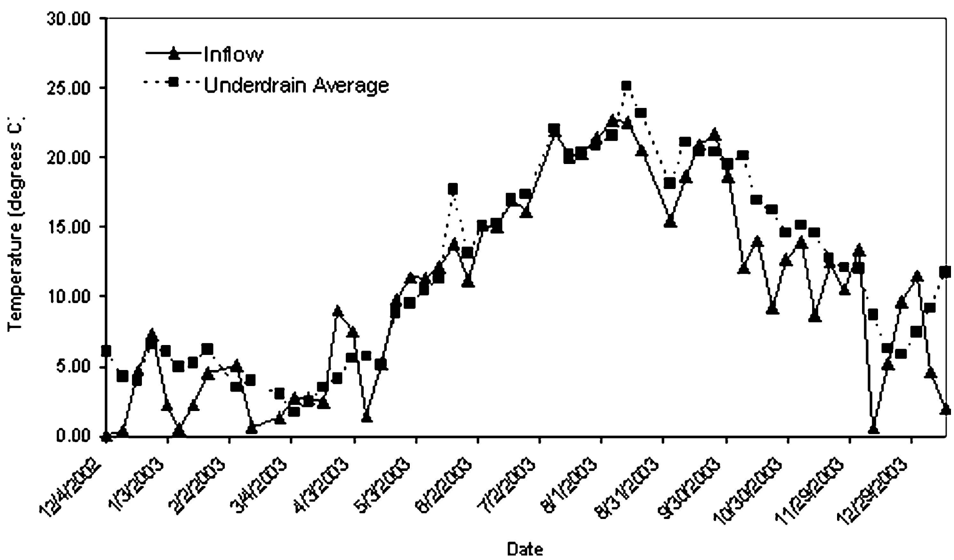

Figure 5. Temperature of influent and effluent water in rain garden, Haddam, CT.

TABLE V
Seasonal mean water temperatures for rain gardens, Haddam, CT

Season	Roof runoff (°C)	Underdrain outflow (°C)	Difference
Fall	14.5	17.1	−2.5 a
Winter	4.3	5.9	−1.5 a
Spring	7.8	7.6	0.1 a
Summer	19.3	19.8	−0.5 a

Mean differences followed by the same letter for each season were not significantly different at $p = 0.05$.

roof faces north, so the summer rains may not have picked up as much heat as if the roof had a more southerly exposure.

3.7. Redox potential

The average shallow probe (28 cm) measurements for both gardens were 503 and 555 mv, for garden 1 and garden 2 respectively (Table VI). The average deep probe (55 cm) measurements were 545, and 123 mv for garden 1 and garden 2, respectively. The minimum measurements taken were all less than 96 mv. Denitrification reactions are expected to take place in soils between 500 and 200 mv (Bohn *et al.*, 2001). Two of the four average measurements for both gardens were within this range, and the minimum measurements were below this range. This indicates that during the period of measurement, conditions did exist that would favor denitrification. It is not understood why NO_3-N concentrations did not change after passing through the gardens. Although the redox potential was suitable for denitrification, it is possible that soil-water contact time was too short due to rapid infiltration rates, or a suitable electron donor was not present in large enough concentrations to allow denitrification to occur. Kim *et al.* (2003) reported NO_3-N removal of up

TABLE VI
Redox potential measurements for Haddam rain garden ($n = 23$)

	Minimum (mv)	Average (mv)
Rain garden 1		
Deep	32	545
Shallow	−58	503
Rain garden 2		
Deep	−344	123
Shallow	96	555

to 80% when shredded newspaper was added to rain garden soils in laboratory soil columns.

4. Conclusions

Only 0.8% of inflow left the gardens as overflow, indicating high infiltration of roof runoff. The vast majority (98.8%) of inflow left the gardens as subsurface flow. The rain gardens reduced the peak flow rate and increased the lag time of influent water.

The only nutrient well-retained by the rain gardens was NH_3-N. Only NH_3-N and TN concentrations from the underdrains were significantly ($p = 0.001, 0.05$) lower than inlet concentrations. Although the concentration of TN was reduced significantly ($p = 0.05$) from the inlet to the underdrain of rain garden 1, the mass retention of TN was poor. The export of TP from the system was not expected, and we believe that disturbance of the soil was the cause.

Overall, these rain gardens provided runoff control, but water quality renovation was not good. Installing a rain garden without an underdrain may not be appropriate in all situations. However, given the high overall retention of flow found for the 2.54 cm design method used in this study, a rain garden could be an effective BMP in reducing flow and pollutant loads if an underdrain were not connected to the stormwater system.

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