Environ. Sci. Technol. **2006**, *40*, 1335−1340

Saturation to Improve Pollutant Retention in a Rain Garden

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Rain gardens have been recommended as a best management practice to treat stormwater runoff. Replicate rain gardens were constructed in Haddam, CT, to treat roof runoff. The objective of this study was to assess whether the creation of a saturated zone in a rain garden improved retention of pollutants. The gardens were sized to store 2.54 cm (1 in) of runoff. Results show high retention of flow; only 0.8% overflowed. Overall, concentrations of nitrite+nitrate-N, ammonia-N, and total-N (TN) in roof runoff were reduced significantly by the rain gardens. Total-P concentrations were significantly increased by both rain gardens. ANCOVA results show significant reductions in TN (18%) due to saturation. Redox potential also decreased in the saturated garden. Rain garden mulch was found to be a sink for metals, nitrogen, and phosphorus, but rain garden soils were a source for these pollutants. The design used for these rain gardens was effective for flow retention, but did not reduce concentrations of all pollutants even when modified. These findings suggest that high flow and pollutant retention could be achieved with the 2.54 cm design method, but the use of an underdrain could reduce overall pollutant retention.

Introduction

The negative impact of urbanization on water quality is well-documented (*1*). Rain gardens have been recommended as a best management practice (BMP) to reduce urban nonpoint source pollution (*2*). Rain gardens are shallow depressions in the landscape that typically include plants and a mulch layer or ground cover. In addition to providing increased groundwater recharge, they are expected to provide pollutant treatment. Pollutant treatment in rain gardens has been attributed to adsorption, decomposition, ion exchange, and volatilization (*2*). Although some pollutants such as ammonia-N (NH_3−N), total phosphorus (TP), copper (Cu), lead (Pb), and zinc (Zn) have been found to be retained well by laboratory scale-model rain gardens (*3*), nitrite+nitrate-N (NO_3−N) has been shown to be poorly retained. Excess nitrogen in stormwater runoff continues to cause eutrophication and low dissolved oxygen in estuarine ecosystems such as the Long Island Sound (*4*).

The negatively charged NO_3−N ion travels readily through soils. One method of reducing the impacts of NO_3−N on receiving waters is to accelerate its conversion to a different form. Denitrification can occur in anaerobic environments where NO_3−N and a carbon source are present (*5*). Under these conditions, NO_3−N is converted by bacteria to nitrogen gases (*6*). Denitrification has been utilized to reduce NO_3−N concentrations in a variety of systems including septic system effluent (*7*), agricultural fields (*8*), and constructed wetlands (*9*). Denitrification in riparian zones has been noted to be a sink for NO_3−N in groundwater (*10*, *11*). In general, high rates of denitrification may be independent of the soil matrix and "hotspots" of organic matter may be responsible for observed NO_3−N removal rates where conditions are suitable for denitrification (*12*).

Event mean NO_3−N concentration in runoff from residential areas has been reported to be 0.7 mg L^{-1} (*1*). However, concentrations as high as 13 mg L^{-1} have been reported (*13*). Retention of NO_3−N and NH_3−N in a laboratory, pilot-scale rain garden was 24% and 79%, respectively (*3*). Retention of NO_3−N by rain gardens in Haddam, CT was a similar 35% (*14*). The creation of a saturated zone at the bottom of the rain garden has been recommended to promote denitrification and increase the poor retention of NO_3−N by rain gardens (*15*). However, the role of denitrification in field-based rain gardens is undocumented.

The objective of this project was to evaluate the effect of a saturated zone in a rain garden on pollutant concentrations in roof runoff. The target pollutant was NO_3−N, although we also examined the effects on other pollutants and redox potential. Two years of monitoring results from an unmodified rain garden are also presented.

Experimental Section

Materials and Methods. Two rain gardens were installed in September 2002 according to specifications in a rain garden design manual (*2*). The gardens were sized to have an aboveground storage capacity equivalent to 2.54 cm (one inch) of runoff from the contributing roof area (106.8 m^2), which resulted in a surface area of 18.4 m^2 for each 15 cm (6 in.) deep garden. Installation, sample handling details, and soil characteristics have been described in detail elsewhere (*14*). Weekly samples were analyzed for TP, TKN, NH_3−N, and NO_3−N on a Lachat colorimetric flow injection system using EPA methods (*16*). Monthly composite samples were analyzed for copper (Cu), lead (Pb), and zinc (Zn) using inductively coupled plasma-mass spectrometry (ICP−OES) method 200.7 (*17*). Precipitation and redox potential methods were also described previously (*14*).

Plantings in each garden included chokeberry (*Aronia prunifolia*), winterberry (*Ilex verticillata*), and compact inkberry (*Ilex glabra compacta*). A layer of shredded hardwood mulch approximately 5 cm thick was applied. Plants and mulch were sampled for metals, N, and P. Fifteen samples of approximately 10 g each were taken from the bulk mulch at the beginning of the study, and from each garden at the end of the study, and composited. Plant samples were taken from aboveground biomass of each shrub (leaf and twig) during leaf-on conditions. Mulch and plant tissue samples were ground using a Cyclotec sample mill, and a subsample from each was analyzed for total Cu, Pb, and Zn using ICP-MS method 6010b (*18*). Mulch and plant TN and TP was determined using a salicylic−sulfuric acid digestion and colorimetric analysis (Technicon II, Industrial Method 334-74W/B). Adsorption of metals, TN, and TP on mulch particles was calculated based on measured bulk density and concentrations, where mass adsorbed = mass measured at end

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10.1021/es051644f CCC: $33.50
Published on Web 12/28/2005

TABLE 1. Flow Mass Balance (2002−2004) for Haddam, CT Rain Gardens

	volume (L)	depth[a] (cm)	% of inflow
inflow			
roof runoff	312,920	1202	79.7
precipitation	79,607	306	20.3
total	392,527	1507	
outflow			
underdrain	374,580	1438	95.4
overflow	3326	13	0.8
total	377,906	1451	15
residual	14,621	56	3.7

[a] Depth represents depth of water in the rain gardens.

of study − mass measured at beginning of study. Equations relating aboveground biomass to shrub basal diameter were developed for the woody shrubs used in this study based on the methods developed by Telfer (*19*). Plant uptake of TN, TP, and metals was calculated from dry matter yield based on calculated biomass estimates and measured concentrations. Below-ground biomass was not sampled or estimated.

Statistical Analysis. Analysis of variance (ANOVA) was used to determine whether significant concentration differences existed between precipitation, roof runoff, underdrain outflow, and overflow. Water quality concentrations were log-transformed prior to analysis as pollutant concentrations were found to be log-normally distributed. Duncan's Multiple Range Test was used to determine differences among means.

A value of one-half the detection limit was substituted for any analyte reported as "nondetect" (ND). However, if greater than 50% of samples were ND, a test of proportion was used to evaluate concentration differences (*20*). The Chi-square test was used to determine if differences existed in the proportion of samples below detection between precipitation, roof runoff, and underdrain outflow. SAS version 9.1 was used for all statistical analyses (*21*). Percent retention was calculated for each pollutant based on the following formula: % retention = (mass in − mass out) × 100/mass in.

The effect of the saturation treatment was analyzed using the paired watershed design (*22*). The benefit of this method is that variance due to individual differences between rain gardens can be controlled. The two rain gardens were monitored for a one year calibration period, and a regression relationship was developed between the two rain gardens for each of the constituents measured. During the treatment period the outlet of the underdrain pipe was raised, creating a saturated condition in 0.5 m of one rain garden (0.25 m in the stone and 0.25 m in the soil mixture). Analysis of covariance (ANCOVA) was used to determine whether significant differences due to treatment existed between slopes and intercepts of the regressions for the calibration and treatment periods for all constituents measured.

Results and Discussion

Precipitation. During the 12-month calibration period (December 2002 to December 2003) 172.8 cm of precipitation was measured at the Haddam site, and 134.9 cm of precipitation was measured during the treatment period (January 2004 to December 2004). Measured precipitation at a NOAA station in Groton, CT 40 km away was equal to the 30-year normal for that station during the study period (*23*).

Flow. Most roof runoff left the rain gardens as subsurface flow (95.4%). Only 0.8% of the inflow water overflowed during the entire study period (Table 1). The residual volume (3.7%) was assumed to be evapotranspiration (ET) from the gardens. The 56 cm evapotranspired during the study period (28 cm yr^{-1}) was lower than expected. Mean annual pan evaporation at Norfolk, CT is 52.1 cm yr^{-1} (*24*). Using a pan coefficient of 0.7 (*25*), ET was estimated to be 36.5 cm. Given that the pan value reported is a mean, year to year variation could likely explain the difference from the 28 cm yr^{-1} residual found at the Haddam rain garden. The difference may also be attributed to the shady location of the gardens and the layer of bark mulch on the gardens, which could have reduced ET from the rain gardens.

Nitrogen and Phosphorus. *Overall Concentration Results.* NO_3-N concentrations in underdrain outflow from the saturated garden were significantly (p = 0.001) lower than roof runoff concentrations during the entire period of study (Table 2). Overflow concentrations of NO_3-N were significantly higher than underdrain outflow concentrations for both gardens. NH_3-N concentrations in outflow from both of the underdrains were significantly (p = 0.001) lower than roof runoff, precipitation, and overflow concentrations. However, a large percentage of NH_3-N concentrations were below detection for the underdrains (73% and 70% for the saturated and unsaturated gardens, respectively). Based on a Chi-square test, a significantly ($\chi^2 = 42.97$, $p = 0.001$) higher proportion of NH_3-N samples were below detection in underdrain outflow than in precipitation and roof runoff. TN concentrations in underdrain outflow from the treatment garden were significantly (p = 0.001) lower than roof runoff concentrations (Table 2). In addition, TN concentrations in outflow from both underdrains were significantly lower than overflow concentrations. Interestingly, TP concentrations in underdrain outflow from both gardens were significantly (p

TABLE 2. Geometric Mean Pollutant Concentrations (2002−2004) Haddam CT Rain Garden[a]

						underdrain		overflow[b]	
variable	*n*	DL[c]	unit	bulk deposition	roof runoff	saturated	unsaturated	saturated	unsaturated
NO_3-N[d]	73	0.2	mg L^{-1}	0.7 bc ±*0.9*	0.9 ab ±*1.5*	0.3 c ±*0.4*	0.4 bc ±*0.4*	2.0 a ±*2.9*	2.1 a ±*0.6*
NH_3-N[d]	78	0.01	mg L^{-1}	0.04 a ±*0.12*	0.04 a ±*0.19*	0.01 b ±*0.02*	0.01 b ±*0.11*	0.08 a ±*0.04*	0.04 a ±*0.02*
TKN[e]	79	0.1	mg L^{-1}	0.5a ±*0.5*	0.6a ±*0.7*	0.4a ±*0.2*	0.5a ±*0.3*	0.6a ±*0.3*	0.3a ±*0.4*
TN[d]	72	0.1	mg L^{-1}	1.3 abc ±*1.0*	1.6 ab ±*1.6*	0.7 c ±*0.5*	0.9 bc ±*0.6*	2.7 a ±*3.1*	2.4 a ±*1.1*
ON[e]	77	0.1	mg L^{-1}	0.4 a ±*0.5*	0.5 a ±*0.6*	0.4 a ±*0.2*	0.5 a ±*0.4*	0.5 a ±*0.4*	0.2 a ±*0.5*
TP[d]	80	0.005	mg L^{-1}	0.009 b ±*0.015*	0.015 b ±*0.032*	0.039 a ±*0.049*	0.043 a ±*0.055*	0.009 b ±*0.018*	0.016 b ±*0.026*

[a] Means followed by the same letter for a given variable are not significantly (p = 0.05) different from each other using Duncan's Multiple Range Test. Standard deviation values are below each mean. [b] n = 4 for overflow samples. [c] DL = Detection limit. [d] p = 0.001. [e] ns = ANOVA comparison nonsignificant.

= 0.001) higher than precipitation, roof runoff, or overflow concentrations. Although some differences were found among overflow concentrations and other measurements, these comparisons should be interpreted with caution due to the low number (4) of overflow samples.

Although TP concentrations were higher from the underdrains than roof runoff when analyzed using ANOVA, a time plot revealed a decaying trend in underdrain outflow TP concentrations (*14*). The decay trend continued through the second year of study, and TP concentrations in roof runoff and underdrain outflow toward the end of the study period were similar (data not shown). Similar results were reported in a preliminary study in North Carolina (*26*), where a median TP concentration of 0.13 mg L^{-1} in roof runoff was found for 21 events, and the median outflow concentration from a bioretention cell was 2.40 mg L^{-1}. A loamy sand in Sweden was also found to be a source of TP in percolate water, as compared to a loam and a silty clay loam, which were TP sinks (*27*). We suspect that the physical disturbance of the soil at the beginning of the study period resulted in flora and soil fauna destruction, with a subsequent release of phosphorus. The phosphorus was then easily leached out as the low ionic strength roof runoff passed through the soils, resulting in increased TP concentrations in underdrain outflow. In general, TP concentrations found in roof runoff at the Haddam site (Table 2) were lower than others reported. In Wisconsin, geometric mean concentrations of 0.15 mg L^{-1} (*31*) and 0.07 mg L^{-1} (*32*) in residential roof runoff were found. A geometric mean TP concentration of 0.06 mg L^{-1} was reported for residential roof runoff in Michigan (*28*). It is not known whether rain garden soils would show a greater TP concentration reduction if inflow TP concentrations were higher.

Concentrations of NH_3-N in bulk deposition and roof runoff from the Haddam study site were lower than those measured at several other study locations. A higher geometric mean of 0.44 mg L^{-1} was reported for residential roof runoff in Michigan (*28*) (Table 2). A geometric mean of 0.35 mg L^{-1} was reported in runoff from a residential roof in Pennsylvania with asphalt shingles (*29*). A median NH_4-N concentration of 0.859 mg L^{-1} in runoff from an asphalt composition shingle roof in Texas was not significantly different from the 0.799 mg L^{-1} NH_4-N measured in open rainfall (*30*).

In contrast, NO_3-N concentrations in precipitation and roof runoff at the Haddam site were higher than those reported elsewhere. For example, a geometric mean NO_3-N concentration of 0.46 mg L^{-1} in residential roof runoff in Michigan has been reported (*28*). The geometric mean for NO_3-N of 0.68 mg L^{-1} reported in runoff from a residential roof in Pennsylvania with asphalt shingles (*29*) was only slightly lower than the concentrations measured at the Haddam site. Bulk deposition rates of 8.3 and 0.04 kg $ha^{-1}yr^{-1}$ have been reported for TN and TP, respectively, in Connecticut (*33*). This is less than the 28 (TN) and 0.27 kg $ha^{-1}yr^{-1}$ (TP) deposition rates measured at the study site. It is suspected that several trees in close proximity to the deposition sampler at the rain garden site contributed the TN and TP mass.

For the entire study period, no significant differences in the concentrations of ON and TKN were found among bulk deposition, roof runoff, overflow, or underdrain outflow concentrations (Table 2).

Mass Export. Percent mass retentions for the two-year study period followed concentration results; low retention of TKN and ON were found for both the unsaturated and saturated gardens (Table 3), corresponding to nonsignificant concentration comparisons (Table 2). Retention of TKN and TP was lower than the reported removals of 68% and 81%, for TKN and TP, respectively in laboratory pilot scale rain gardens (*3*). NH_3-N and NO_3-N retention was high for the entire study period (Table 3). However, only 36% retention of NO_3-N occurred during the calibration period (Table 3). Higher NO_3-N retention by the saturated (87%) and unsaturated (81%) gardens was noted during the treatment period (Table 3). Higher roof runoff NO_3-N concentrations were found during the treatment period than during the calibration period (data not shown). Less precipitation (−22%) also fell during the treatment period as compared to the calibration period. It is possible that during the calibration period the lower NO_3-N concentrations in roof runoff and the higher volume of water that passed through the gardens led to a lower retention of NO_3-N. Retention of NO_3-N by the Haddam rain gardens was greater than the 24% removal of NO_3-N reported for laboratory rain gardens (*3*). NO_3-N retention includes possible transformation to another nitrogen form. Losses such as the conversion of NO_3-N to gaseous forms were not quantified in this study.

TABLE 3. Percent Retention for Haddam, CT Rain Gardens[a]

period	NO_3-N	NH_3-N	TKN	TP	TN	ON
calibration						
unsaturated garden	36	84	29	−104	31	19
saturated garden[a]	36	86	34	−117	34	24
treatment						
unsaturated garden	81	86	22	−104	68	6
saturated garden	87	69	5	−98	69	−9
overall (both periods)	67	82	26	−108	51	14

[a] Both gardens were unsaturated during the calibration period, and the treatment garden was saturated during the treatment period. Overall retention is for the entire period of study.

TABLE 4. Mulch, Soil, and Plant Average Concentration Changes (mg kg^{-1}) from Beginning of Study (November 2002) to End of Study (December 2004), Haddam, CT

	mulch	soil	*I. verticillata*	*I. glabra*	*A. prunifolia*
TN					
2002	1800	337	8650	8900	8700
2004	4726	276	11920	8842	8739
% change	163	−18	38	−1	0
TP					
2002	335	274	965	815	1430
2004	362	188	1532	783	1842
% change	8	−31	59	−4	29
Cu					
2002	4	17	13	4	ND
2004	52	15	18	6	17
% change	1200	−12	38	50	
Pb					
2002	3	54	ND	ND	ND
2004	12	41	ND	ND	ND
% change	300	−24			
Zn					
2002	17	68	21	28	20
2004	42	39	344	65	35
% change	147	−43	1538	132	75

TN was retained (51%) by the rain gardens during the entire study period (Table 3). Mulch retained 33% of the total TN inputs, but plants retained only 0.3% of TN inputs. Overall TP retention was negative (Table 3), indicating that more phosphorus left the system than entered. Plant uptake of TP was only 3% of total TP inputs. However, mulch retained 117% of the total TP inputs. Retentions greater than 100% could be caused by uptake of available soil P or by sampling errors. One composite sample of the mulch from each garden was used for the initial and final concentrations.

Over the entire study period, TN and TP concentrations increased in the mulch, and decreased in the soils (Table 4). These results indicate that although the mulch was a sink for

TABLE 5. Geometric Means and ANCOVA Results for Underdrain Outflow (mg/L) during Calibration and Treatment Periods, Haddam, CT Rain Gardens[a]

	calibration period (n = 41)		treatment period (n = 32)			
				saturated		
	unsaturated	saturated[b]	unsaturated	observed	predicted	% change
NO_3-N	0.4	0.3	0.3	0.2	0.2	0
NH_3-N						
TKN	0.6	0.4	0.4	0.4	0.4	0
TN	1	0.8	0.8	0.6	0.7	−18[c]
ON	0.5	0.4	0.4	0.4	0.4	−3
TP	0.059	0.058	0.028	0.022	0.040	−82[d]

[a] Percent change represents the percent difference between the expected value for the saturated garden during the treatment period, and the observed value. [b] Both gardens were unsaturated during the calibration period. [c] P value = 0.05. [d] P value = 0.001.

both TN and TP, rain garden soils were a source of both pollutants.

The nutrient most highly retained by both gardens was NH_3-N (Table 3). This level of retention may be expected, as high retention of NH_3-N has been reported even in gravel soils (*34*). Nitrification and adsorption of NH_4-N were reported to be the primary mechanisms responsible for the NH_3-N removal. For the Haddam rain garden, adsorption was assumed to be the primary mechanism responsible for the decrease in NH_3-N concentrations, as a corresponding increase in NO_3-N concentrations in underdrain outflow was not found. However, it is also possible that concurrent nitrification/denitrification caused the decrease in NH_3-N concentrations, with no noticeable increase in NO_3-N concentrations.

The input to these gardens was roof runoff with low concentrations of pollutants. These water quality results may not apply to source water with higher contaminant concentrations.

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, the rain garden could be an effective BMP in reducing flow and pollutant loads if an underdrain were not connected to the stormwater system.

Effects of Saturation. ANCOVA showed no net change in NO_3-N outlet concentrations from the treatment garden during the treatment period as compared to the calibration period (Table 5). One explanation for the lack of change in NO_3-N concentrations in underdrain outflow from the saturated garden may be related to the low level of the concentrations. During the calibration period, 19% of samples were at or below the detection limit of 0.2 mg L^{-1} for both the saturated and unsaturated gardens. However, during the treatment period, 56% of samples were at or below the detection limit for the saturated garden, whereas only 25% of samples from the unsaturated garden were at or below the detection limit. The existence of a large number of samples below detection limit can limit the regression relationship, thereby providing inaccurate estimates of percent change. When the frequencies of samples above and below detection were analyzed using Chi-square analysis, a significant (χ^2 = 11.03, p = 0.01) difference between the calibration and treatment periods was detected in the saturated garden, whereas no difference was detected in the unsaturated garden. A greater proportion (56% vs 19%) of underdrain outflow samples from the saturated garden were below detection in the treatment period.

Due to the high percentage of samples below detection for NH_3-N (>59% for both gardens during calibration and treatment), ANCOVA was not performed for NH_3-N. However, Chi-square frequency analysis revealed significantly ($\chi^2 = 5.43$, $p = 0.05$) fewer ND values in underdrain outflow from the treatment garden during the treatment period as compared to the calibration period. No difference between periods was detected for the control garden. TN concentrations decreased by 18% due to saturation, with significant (p = 0.05) changes in both the slopes and intercepts of the calibration and treatment regressions (Table 5). The regression equation for TN for the calibration period was based on observed concentrations, and was the following: saturated garden TN = 0.806 × unsaturated garden TN∧(0.83814). The mean observed TN concentration in the unsaturated garden during the treatment period was then entered into this equation to determine a predicted mean concentration. The percent change is the percent difference between the predicted and observed mean concentration. No significant differences were found for TKN or ON concentrations between the calibration and treatment periods. Despite the export of phosphorus from the system noted in Table 3, TP concentrations were reduced by 82% due to the saturation treatment (Table 5). Although the calibration period regression was highly significant for TP, the treatment period regression was not significant ($R^2 = 0.04$). A reduction in TP concentrations due to anaerobic conditions in a rain garden has been noted in a preliminary study in North Carolina (*26*), where outflow TP concentrations were 2.40 and 0.65 mg L^{-1}, for a traditional and a partially saturated bioretention cell, respectively. The median TP concentrations in the roof runoff were 0.13 mg L^{-1}. Trapping of sediment-bound phosphorus in the sump area of the saturated zone was cited as the mechanism for the higher TP retention by the saturated bioretention cell (*26*). In general, TP concentrations would be expected to increase as a result of lower redox potentials, as TP bound in the Fe redox couple becomes soluble (*35*). Clausen and Spooner (*22*) caution that in paired watershed analysis, both watersheds should be in steady-state conditions. It is possible that the decay trend noted for TP (*14*) biased ANCOVA results, and caused an erroneous result.

The creation of the saturated zone reduced concentrations of TN and TP, and increased the frequency of ND samples for NO_3-N. However, saturation decreased the frequency of ND samples for NH_3-N.

Metals. *Overall Results.* Due to the large number of samples at or below detection limits for Cu, Pb, and Zn, ANOVA and percent retention were not performed on these pollutants. Geometric mean concentrations of 15, 21, and 149 μg L^{-1} for Cu, Pb, and Zn, respectively, were reported for residential roof runoff in Wisconsin (*31*). The presence of metals in roofing materials such as flashing and gutters seems to be the primary cause of high metals concentrations in runoff from certain roofs (*31*, *34*, *36*−*38*). There was no exposed flashing on the roof in Haddam.

Mulch retained 98%, 36%, and 16% of Cu, Pb, and Zn inputs, respectively. Plants only removed 0.1, 0, and 0.2% of the inputs of Cu, Pb, and Zn, respectively.

Soils metals concentrations decreased in the rain gardens during the study (Table 4). Mulch is a likely sink for metals, but rain garden soils were a source of all metals measured. For laboratory prototype rain gardens, increases in mulch metals concentrations by factors of 6, 1.7, and 17 for Cu, Pb, and Zn, respectively, have been found (*3*), although the percentage of input mass was not reported. At the Haddam rain gardens, average (treatment and control gardens) mulch metals concentrations increased by factors of 13.1, 4.1, and 2.4 for Cu, Pb, and Zn, respectively.

Effects of Saturation. Due to the large number of samples at or below detection (60%, 89%, and 43% for Cu, Pb, and

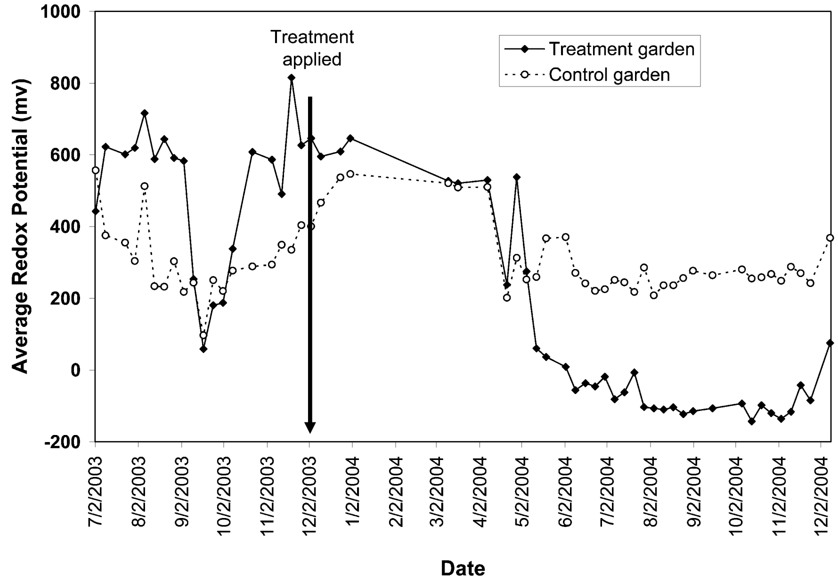

FIGURE 1. Average E_h (mv) over time, Haddam, CT rain gardens. Treatment period began January 2004.

Zn, respectively) ANCOVA was not performed. During the calibration period, 14% and 21% of Zn concentrations were below detection for the control and saturated gardens, respectively. However, during the treatment period, 17% of Zn samples were below detection in the control garden, whereas 67% of Zn samples in the saturated garden were below detection. Chi-square frequency analysis showed a significant ($\chi^2 = 5.42$, $p = 0.05$) increase in samples below detection for the saturated garden between the calibration and treatment periods. No changes were detected for the control garden. It is possible that the reducing conditions in the treatment garden caused Zn to be complexed, and therefore less mobile. In a laboratory study, Zn was over 90% complexed with soluble organic matter under reduced conditions (*39*). No significant differences were detected using Chi-square analysis for Cu and Pb.

Redox Potential. During the treatment period, the average shallow and deep probe E_h measurements in the treatment garden decreased (Figure 1). Average E_h in the control garden remained above 200 mv during the entire treatment period, whereas E_h in the treatment garden dropped beginning in May 2004 (Figure 1). Denitrification reactions are expected to take place in soils between 500 and 200 mv (*35*). During the treatment period, on average, conditions existed that were conducive to greater denitrification in the treatment garden. This is consistent with the Chi-square results for NO_3-N concentrations.

Temperature. Water temperatures in roof runoff and underdrain outflow were similar throughout the study period (Figure 2). The lack of a cooling effect in the summer was surprising. The relatively shallow (0.6 m) soil depth and the short contact time with soils may have reduced the cooling effect of the soils. In addition, the roof faced northeast. Heat gain by the runoff would likely have been greater if the roof had a southerly exposure.

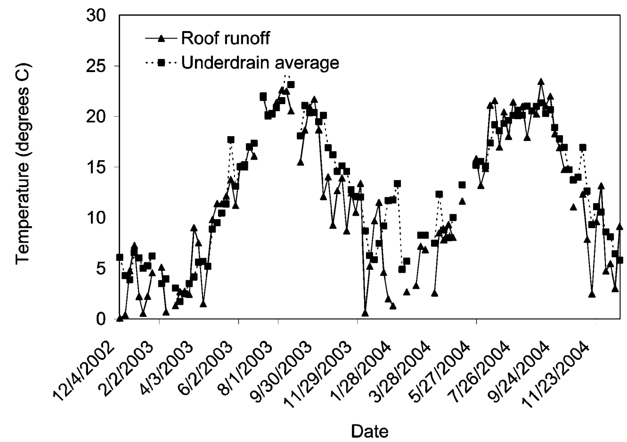

FIGURE 2. Roof runoff and underdrain average water temperature, Haddam, CT rain gardens.

Acknowledgments

Funding for this project was provided by the Long Island Sound Fund administered through sale of Long Island Sound license plates and contributions. The project was also funded in part by the DEP through a USEPA nonpoint source grant under §319 of the Clean Water Act.

Supporting Information Available

Detailed rain garden cross section, table with soil and mulch characteristics, sample ANCOVA regression, additional monitoring data for temperature, and an unexpected finding for manganese. This material is available free of charge via the Internet at http://pubs.acs.org.

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Received for review August 18, 2005. Revised manuscript received November 15, 2005. Accepted November 28, 2005.

ES051644F