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MEMOIRE

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Sujet :
Asphalt parking lot runoff nutrient quality: characterization and pollutant removal by bioretention cells

Pour l'obtention du diplôme de master de sciences et technologies

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Résumé

Ce projet est centré autour de la pollution diffuse (principalement nutriments) des eaux de ruissellement provenant de surface asphaltées de parkings. Les principaux objectifs consistaient à caractériser la qualité du ruissellement, à déterminer les facteurs ayant une influence sur les concentrations (C) et les charges (M) en nutriments pour huit sites situés en Caroline du Nord (CN) (USA) et à évaluer les efficacités d'élimination des polluants de deux cellules de biorétention recevant ces eaux de ruissellement. Pour huit parkings, les C et les M moyennes par évènement de pluie ont été mesurées et statistiquement analysées pour six formes de nutriments, TN, TKN, NH_4-N, $NO_{2,3}$-N, TP et OPO_4-P, dont les C moyennes étaient 1.57, 1.19, 0.32, 0.36, 0.19 et 0.07 mg/L, respectivement. Par comparaison à des études précédentes sur le ruissellement provenant de routes ou d'autoroutes ($R_{r\text{-}a}$), les C en azote étaient légèrement plus faibles que celles trouvées par ces études alors que les C en phosphore en différaient peu. Les modèles actuels de prédiction des M en polluants pour les bassins versants urbains, généralement basés sur les C en nutriments du $R_{r\text{-}a}$, sont donc soupçonnés de sur-estimer les M en azote dans les eaux de ruissellement issues de parking asphaltés. Les C et les M les plus fortes ont été trouvées au printemps et en été, respectivement. Des différences saisonnières significatives ($p<0.05$) ont été mises en évidence entre le printemps et les deux saisons les plus froides (l'automne et l'hiver) pour les C et entre l'été et ces deux saisons pour les M. Par des tests de correlation de Pearson et des analyses de régression linéaires multiples, il a été montré que les facteurs les plus influents pour prédire les C et les M en nutriments étaient la hauteur de la pluie, la surface du bassin versant, le pourcentage en asphalte du bassin versant et une utilisation “naturelle” du sol environnant. Toutefois, ces facteurs n'étaient pas tous significatifs simultanément. Les résultats ont montré que les nutriments s'accumulaient sur les parkings asphaltés puis étaient entrainés par les eaux de pluie. De plus, il a été suggéré que le taux d'accumulation des nutriments était plus élevé que le taux de production du ruissellement. Deux cellules enherbées de biorétention chacune incluant une zone interne de stockage de l'eau ont été suivies pendant 12 mois. La profondeur des milieux filtrants étaient 0.75 m et 1.05 m pour le bassin nord (N) et sud (S), respectivement. Les types de sol sous les milieux filtrants étaient argileux (N) et sablo-limoneux (S). Les eaux de ruissellement et les eaux en sortie des cellules ont été analysées pour TN, TKN, NH_4-N, $NO_{2,3}$-N, TP, OPO_4-P et coliformes fécaux. En sortie des cellules, les volumes et les pics de débit étaient générallement inférieurs à ceux du ruissellment. Globalement, les C et les M en espèces azotées étaient significativement plus faibles que celles des eaux de ruissellement, sauf pour la cellule S et pour les M en $NO_{2,3}$-N. Les taux de réduction des espèces azotées (mis à part $NO_{2,3}$-N) étaient compris entre 34 et 85%. Mis à part pour l'automne et l'hiver où un temps de rétention hydraulique plus long semblait nécessaire, les zones internes de stockage

de l'eau ont probablement soutenu les réactions de dénitrification. Les taux de réduction des C et M en TP (de 31 à 54%) n'étaient généralement pas significatifs et les C en OPO_4-P ont augmenté (-35(N) et -45(S) %). Toutefois, les C en sorties des cellules pour les deux formes de phosphores étaient relativement faibles. Les plus forts taux de réduction des C et des M ont eu lieu au printemps et en été. En prenant en compte à la fois les taux de réduction et les C en sortie des cellules, ces systèmes enherbés ont montré des résultats prometteurs pour réduire la pollution par les coliformes fécaux et les nutriments par rapport aux systèmes de biorétention communément végétalisés (arbres, buissons, mulch) étudiés en CN. Enfin, il semble que la configuration de la cellule N était plus efficace que celle de la cellule S.

Abstract

The objectives of this study were to characterize asphalt parking lot (APL) runoff quality, to determine influent factors to predict nutrient concentrations and loads from eight sites in North Carolina and to assess two grassed bioretention cell efficiencies to remove APL runoff pollutants. From the eight APL sites, event mean concentrations (EMCs) and loads were measured and statistically analyzed for six nutrient forms, TN, TKN, NH_4-N, $NO_{2,3}$-N, TP and OPO_4-P, whose average EMCs were 1.57, 1.19, 0.32, 0.36, 0.19 and 0.07 mg/L, respectively. Nitrogen EMCs were slightly lower than those from highway runoff; whereas, phosphorus EMCs were not very different. Current load prediction models, generally based on highway or roadway nutrient concentrations, are therefore expected to over-estimate nitrogen loads from asphalt parking lots. Spring and summer presented the highest EMCs and loads, respectively. Significant seasonal differences ($p<0.05$) were found mainly between spring and both fall and winter for concentrations and between summer and fall and winter for most loads. In an attempt to determine the factors affecting EMCs and loads, Pearson correlation tests and multiple linear regression analyses were performed. Rainfall depth, catchment area, the percentage of asphalt and natural land use were good predictors of nutrient EMCs and loads. However, the factors were not all significant simultaneously. Results indicated that nutrients accumulated before being washed-off from impervious surfaces and that pollutant build-up rate was greater than the rate of runoff production. Two grassed bioretention cells including internal storage zones (ISZs) were monitored for 12 months in central North Carolina. Fill media depths were 0.75 and 1.05 m for the north (N) and the south (S) cells, respectively and in-situ underdrain soils were clay (N) and sandy loam (S). Asphalt parking lot runoff and outflows were analyzed for TN, TKN, NH_4-N, $NO_{2,3}$-N, TP, OPO_4-P and fecal coliforms (FC). Outflow volumes and peak flows were generally lower than those of inflow runoff. Overall, effluent nitrogen species concentrations and loads were significantly ($\alpha=0.05$) lower than those of the runoff, except for the south cell EMCs and loads for $NO_{2,3}$-N. Except for $NO_{2,3}$-N, nitrogen species load reductions ranged from 34 to 85 %. Apart from fall and winter during which a longer hydraulic contact time seemed to be needed, the ISZs appeared to improve denitrification. TP EMC and load reductions (from 31 to 54 %) were generally not significant and OPO_4-P EMCs increased (-35 (N) and -45 (S) %). However, effluent concentrations for both phosphorous species were quite low. The best nutrient EMC and load reductions occurred during spring and summer. When considering effluent EMCs in addition to removal rates, the grassed cells showed promising efficiencies for FC and nutrient pollution abatement when compared to conventionally vegetated bioretention cells (tress, shrubs and mulch) previously studied in North Carolina. Finally, it appeared that a bioretention system comprising a 0.75 m fill media depth over a clayey soil was more efficient than a 1.05 m fill media depth over a sandy loam soil.

Introduction

Impermeable surfaces resulting from urbanization increase runoff volumes, limit groundwater recharge and deteriorate water quality (Paul and Meyer 2001, US EPA 2000). While pollution remediation systems are commonly implemented, stormwater runoff continues to contribute to surface water pollution. The major sources of pollutants are dust fall, rainfall, soil erosion and surrounding activities in the watershed including the use of pesticides, fertilizers, domestic chemicals, gas emissions, vehicle liquid leaks, pet wastes, animal and vegetation decomposition or direct disposal onto the drainage area (Pitt et al. 2004; Hall and Ellis 1985; Barnes et al. 2000; Rushton 2001). The limited infiltration of asphalt surfaces enables pollutant build-up and prevents pollutants from being degraded. Thus, during a storm event, precipitation washes them into receiving waters. To minimize stormwater pollution, the National Pollutant Discharge Elimination System (NPDES) Stormwater Permit requires departments of transportation, municipalities, counties and select industries to use Best Management Practices (BMPs) or to pay a fund used to build retrofits elsewhere to mitigate urban non-point source pollution (NC DENR 2004).

Watershed load prediction models are commonly used by engineers to chose and design a BMP. Several types of models exist of varying complexities. Simple stormwater pollutant load models include those from the Total Maximum Daily Load (TMDL) Program in which load predictions are based on the Simple Method (Schueler 1987). The Tar-Pamlico river basin model (North Carolina), the PLOAD model (Oregon), the Spreadsheet Tool for Estimating Pollutant Load (STEPL, national) are some examples. The Simple Method and the Tar-Pamlico river basin model are described in appendix I. Impervious surface nutrient concentrations, which are usually based on highway or roadway nutrient concentrations, are key inputs in such models. Their appropriateness for asphalt parking lot watersheds is arguable. More complex models are generally based on equations describing pollutant buildup and washoff processes (Chen and Adams 2006). A better understanding of the factors most affecting nutrient concentrations on asphalt parking lot areas could improve the accuracy of such models.

Several studies assessed urban area, and particularly highway or roadway runoff quality, to characterize nutrient concentrations and loads (Barrett et al. 1995; Irish et al 1995; Wu et al. 1998; Brezonik and Stadelmann 2002; Choe et al. 2002; Kayhanian et al. 2003; Kayhanian et al. 2007). Asphalt parking lot runoff pollution is much less documented. Hope et al. (2004) studied three highly impervious sites in Arizona by carrying out simulated storm experiments after a very dry season. Large variations and substantially high values were observed for nitrate and ammonium concentrations. Results from a one-year study of two asphalt parking lots at the Florida Aquarium in Tampa are presented by Rushton (2001). Average event mean concentrations (EMCs) were nearly 0.55 mg/L for total nitrogen and 0.105 mg/L for total phosphorous. For highways or mixed land use watersheds, the major factors found to affect runoff quality were rainfall amount, intensity, drainage area (Brezonik and Stadelmann 2002), antecedent dry days, land use and average annual daily traffic (Kayhanian et al. 2003; Barrett et al. 1995; Driscoll et al. 1990). Asphalt parking lot surfaces occupy a substantial portion of urban watersheds, and they are assumed to show similar trends on nutrient accumulation and wash-off phenomena.

Many BMPs are able to reduce peak flows, runoff volumes and ground or surface water pollution by increasing evapotranspiration, infiltration and providing pollutant removal processes. Among BMPs, bioretention cells are used for infiltration and filtration. This BMP is generally considered a shallow depression in the landscape acting as a vegetated soil filter. They are commonly covered with hardwood mulch and planted with different kinds of vegetation to meet aesthetic needs. Sub-surface underdrains facilitate infiltration through a fill media preventing the cells from being flooded during a long time. Oil, grease, total suspended solids (Hsieh and Davis 2005) and heavy metals were found to be highly captured by bioretention cells (Davis et al. 2001, Dietz and Clausen 2005). Nutrient removal efficiency of bioretention systems is less systematically high (Hsieh and Davis 2005, Birch et al. 2005, Dietz and Clausen 2005). Phosphorous desorption was found to occurred for high phosphorus index (P-Index) fill media (Hunt et al. 2006). Up-turned underdrains can force water to remain longer into the fill media thus creating internal storage zones (ISZs). ISZs were previously shown to enhance denitrification on column studies (Kim et al. 2000). On field studies, Hunt et al. (2006) found that this effect was not significant; whereas, Dietz and Clausen (2006) found that the presence of an ISZ could reduce TN and TP concentrations but no effect was found for nitrate concentrations. Grass has been increasingly considered as an alternative cover, for a different aesthetic value, simpler maintenance, and reduced installation costs. However, a lack of research data on its pollutant removal efficiency generally prevents grass from being allowed by regulators.

The objectives of this study were (1) to summarize monitoring data from asphalt parking lot runoff quality at eight sites in North Carolina, (2) to statistically characterize event mean concentrations and loads, (3) to determine major factors affecting EMCs and loads, (4) to compare the data to the existing literature, particularly to discern differences between asphalt parking lot and highways, (5) to analyze preliminary results on the efficiency of two grassed bioretention cells utilizing ISZs to capture nutrients and fecal coliforms and (6) to discuss changes in pollutant concentrations and loads on global and seasonal bases.

I. Methods

I.1.Asphalt parking lot runoff nutrient quality characterization for eight sites in North Carolina, USA

I.1.1. Site selection

Eight sites were included into the analysis. Each site met two criteria. First, a minimum of ten sampled storm events was required, and, secondly, the minimum percentage of runoff produced by the asphalt part of the watershed was 70% of the total runoff generated. This percentage was chosen so that a majority of the pollutants came from the asphalt portion of the drainage area. The National Resources Conservation Service Curve Number (NRCS CN) methodology (SCS 1986; Mishra and Singh 2003) was employed to calculate the runoff volumes generated by each land cover of the watersheds (Appendix II). As suggested recently (Woodward et al. 2003; Lim et al 2006), a lower initial abstraction coefficient (λ=0.05) and associated modified curve numbers (CNs) were used.

Table 1. General characteristics of the eight monitored sites

Site location	Site code	Monitoring period	No.[a] SE	DA[a]	PA[a]	SLU[a]	ARP[a]	Rfdepth[a]	Rfint[a]	ADP[a]
Charlotte	C	Feb 2004-March 2006	26	3700	100	Commercial	100	24.5	3.6	162
Kinston	K1	June 2006-Feb 2007	15	111.5	100	Commercial	100	17.2	1.5	87
	K2	July 2006-Feb 2007	11	111.5	100	Commercial	100	17.6	2.6	105
Greensboro	Gre	Sept 2002-August 2004	25	2000	90	Commercial	90	36.6	2.4	166
Goldsboro	Go	June 2003-Dec 2004	14	615	100	Commercial	100	17.7	1.7	89
Louisburg	L1	May 2004-Nov 2004	12	3600	95	Natural area/Park	99	36.0	2.7	151
	L2	June 2004-Feb 2007	16	2200	45	Natural area/Park	88	34.9	1.2	139
Graham	Gra	Apr 2006-March 2007	23	6950	33	Residential	79	21.6	2.8	144

[a] SE = storm event; DA = drainage area (m^2); PA = percentage of asphalt (%); SLU = surrounding land use; ARP = asphalt runoff percentage (%); Rfdepth = average rainfall depth (mm); Rfint = average rainfall intensity (mm/h); ADP = antecedent dry period (h)

I.1.2. Site descriptions

The eight sites were located in six counties in North Carolina (Fig. 1).

Fig. 1. Map of North Carolina highlighting the sites' counties

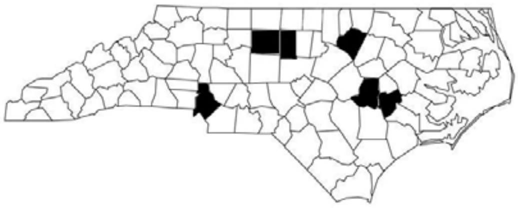


Table 1 summarizes the general characteristics of the sites' drainage areas. Some photos of the drainage areas are presented in Appendix III. The monitoring periods covered from 6 (site L1) to 25 (site C) months in the early 2000s. No data were collected for site L2 from December 2004 through October 2006 and for site Go from March to July 2004. Catchment areas ranged from 111.5 to 6950 m². The major surrounding land use was commercial except for Louisburg sites L1 and L2 which were principally surrounded by a park and Graham which was located in a residential and school area. Most of the drainage areas were nearly 100% asphalt except for sites L2 and Gra. In all cases, between 79% and 100% of the total runoff volume was contributed by the asphalt portion of the watershed. The highest average rainfall depths (Rfdepth, [mm]) during the monitoring periods were nearly 35 mm for Gre, L1 and L2. C presented the highest average rainfall intensity (Rfint, [mm/h]) reaching 3.6 mm/h; whereas, for the other sites, Rfint varied from 1.18 and 2.8 mm/h. Finally, antecedent dry period (ADP, [h]) varied from 87 to 166 h.

Table 2. Analytical methods used by the laboratories

Constituent	U.S. EPA	A.W.W.A.
TP	EPA 365.4	SM 4500-P F
OPO_4-P	EPA 365.1 or EPA 365.2	SM 4500-P F
TN	calculated by TKN+NO_{3+2}-N	
TKN	EPA 351.2	SM 4500Norg B
NH_4-N	EPA 351.2 or EPA 350.1	SM 4500-NH3 G
$NO_{2,3}$-N	EPA 353.2	SM 4500-NO3-E

I.1.3. Data collection and description

Composite samples were collected by automated samplers situated at the outlets of the parking lot drainage areas. In nearly all cases, the lab analyses provided EMCs that were analyzed for six common nutrient forms: total nitrogen (TN), total Kjeldahl nitrogen (TKN), ammonia (NH_4-

N), nitrate plus nitrite ($NO_{2,3}$-N), ortho-phosphorus (OPO_4-P) and total phosphorus (TP). Laboratory analytical procedures followed U.S. Environmental Protection Agency (U.S. EPA 1993) Standard Methods (Table 2) except for Greensboro and Louisburg sites conformed to AWWA Standard Methods for Water and Wastewater Analysis (AWWA 1992). Rainfall and runoff measurement techniques varied per site. Table 3 summarizes sampling procedures and data collection.

Table 3. Rainfall origin, runoff volume determination and composite sample types

Site code	Rainfall origin[a]	Runoff	Composite sample
C	on site or USGS 351320080502645[b]	Measured	Flow-weighted
K1	on site	Measured	Flow-weighted
K2	on site	Measured	Flow-weighted
Gre	on site	Estimated	Rainfall dependent
Go	Cherry Research Station[c]	Estimated	Time dependent
L1	on site	Estimated	Rainfall dependent
L2	on site	Estimated	Rainfall dependent
Gra	on site or USGS 02096500[d]	Estimated and Measured	Time dependent and Flow-weighted

[a] USGS numbers correspond to United States Geological Survey monitoring stations whereas Cherry Research Station was monitored by the State Climate Office of North Carolina (SCO NC)

[b] 2.4 km from C site; [c] 8.4 km from Go site; [d] 3.1 km from Gra site

Referring to a certified analytical problem with one of the laboratories, some data transformations or suppression were performed (Appendix IV). Runoff volumes were either measured on site or estimated by NRCS CN (λ=0.05) methodology (Mishra and Singh 2003; Woodward et al. 2003; Lim et al 2006). Automated samplers took samples either at a constant time-step (time-dependent), or according to a constant amount of passed volume (flow-weighted) or each time a certain amount of rainfall was measured (rainfall weighted). On highly impervious surfaces, rainfall depth is directly correlated to runoff depth. Composite samples triggered by rainfall amounts are therefore a good approximation of flow-weighted composite samples for highly impermeable catchments. Rainfall duration (Rfdur [h]), event average rainfall intensity and antecedent dry period were determined. Event average rainfall intensity was calculated by dividing the total amount of rainfall of every storm by the corresponding Rfdur. Loads (L) and mass flow rates (M) were calculated to express them in area- and time-normalized units.

Loads were determined by multiplying every EMC by the corresponding runoff depth (1):

$$L_{ij} = EMC_{ij} * ROdepth_j \qquad (1)$$

where:

L_{ij} is the load of pollutant i for storm event j (mg/m²),

EMC_{ij} is the event mean concentration of pollutant i for storm event j (mg/L),

$ROdepth_j$ is the total runoff depth of storm event j (L/m²).

Mass flow rates were given by (2):

$$M_{ij} = \frac{L_{ij}}{RFdur_j} \qquad (2)$$

where:

M_{ij} is the mass flow rate of pollutant i for strom event j (mg/(m².h)),

$Rfdur_j$ is the rainfall duration of storm event j (h).

I.1.4. Statistical analyzes

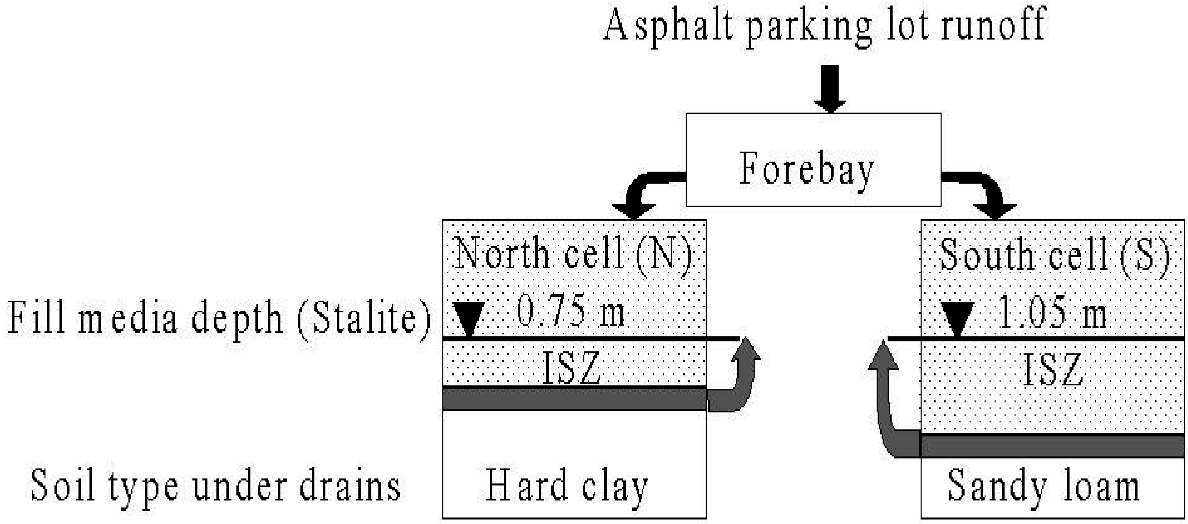
SAS 9.1.3 w/SPA™ software was used for statistical analysis. For each site individually and for all the eight sites, descriptive statistics were performed on EMCs, loads and mass flow rates. Log-normal distributions were tested by probability plot observation and Cramer-von Mises tests implementation on all sites. All statistical analyses were conducted for confidence intervals of 95% (level of significance α=0.05). The results were then compared to the existing literature on urban runoff quality. Seasonal concentrations and loads were compared to assess seasonal variability by descriptive statistics and paired t-tests. Seasons were defined according to solstice and equinox days. For example, Spring started on the Spring Equinox (March 23) and ended on the Summer Solstice (June 21). The effects of select influent factors were assessed on EMCs and loads. Factor statistical distributions were first assessed and Pearson correlation tests were then performed in order to detect possible dependence between the predictive explanatory variables. Correlations among pollutant loads and concentrations were then assessed with the predictors. A multiple linear regression (MLR) analysis was finally implemented and MLR models were computed to obtain standardized coefficients. Forward, backward and stepwise SAS procedures were first employed to select the factors that seemed to best describe constituent concentrations or loads. Then, other factors were added or removed until a predictive MLR model with the highest R-squared value was determined. Residual normal distribution was checked by means of cumulative probability plot observation and Cramer-von Mises tests, and residual constant variance assumption was evaluated examining residuals versus predictive value plots.

I.2. Field grassed bioretention cell efficiencies to remove urban stormwater runoff pollutants

I.2.1. Site and data descriptions

Two grassed (centipede sod, from Living Landscape, Inc.) bioretention cells (Fig. 2) were constructed in the summer 2005 at Graham High School (Cape Fear river basin, Alamance County, North Carolina) and monitored from April 2006 to April 2007.

Fig. 2. Graham bioretention cell diagram showing fill media depths, native soils and internal storage zones.



The 7100 m² drainage area was 33% impervious (asphalt parking lot) and included a large lawn portion (60%) separated from the cells by the parking lot (Appendix V). The BMP surface area to watershed area ratio was around 3.2% and the surrounding neighborhood was residential and a school. For every storm event, a forebay first collected the runoff and equally distributed it to the northern (N) and southern (S) cells. Expanded slate (80% Stalite, 15% sand and 5% topsoil) fill media depths were 0.75 and 1.05 m, for the N and S cells, respectively. The native subsoils were hard clay (N) and sandy loam (S). Up-turned underdrains forced water to remain high in the fill media thus inducing ISZs at the bottom of the cells. The nominal ponding of the cell was approximately 0.25 m and the surface hydraulic conductivity was around 15 cm/hour. ISCO 730 bubbler modules were used for flow rate measurements at the two outlet 90 degree V-notch weirs and at the inlet rectangular weir. The inflow rate was only measured from September 24, 2006; it was estimated by NRCS CN (λ=0.05) methodology (Mishra and Singh 2003; Woodward et al. 2003; Lim et al 2006) beforehand. Rainfall was measured on site by an ISCO tipping bucket rain gauge placed at a 2 m height. Bioretention cell hydrology was characterized by analyzing inflow and outflow volumes and antecedent dry periods. ISCO 6712 automatic samplers collected rainfall- and flow-weighted composite samples at the inlet and both outlets, respectively, to determine event mean concentrations (EMCs). They were analyzed for $NO_{2,3}$-N, NH_4-N, TKN, TN, OPO_4-P and TP.

Analytical methods followed U.S. EPA (U.S. EPA 1993) or APHA Standard Methods (APHA et al. 1998). On September 24, 2006, a change in the laboratory occurred. Prior to this date, only $NO_{2,3}$-N, NH_4-N and TN data were available. Because of a certified laboratory analytical problem, TN values were transformed to remove a 0.2 mg/L bias for events monitored prior to September 24, 2006 (Appendix IV). Organic nitrogen (ON) concentrations were estimated by substracting NH_4-N from TKN. For 7 storm events, grab samples were collected and immediately analyzed by the City of Graham Wastewater Treatment Plant to determine the amount of fecal coliforms (FC). Loads were calculated according to the equation (1) previously cited.

As suggested by Strecker et al. (2001), bioretention cell efficiencies (η_i) were determined not only by calculating removal rates for both EMCs and loads (equation (3)), but by providing and discussing effluent quality for EMCs and loads as well:

$$\eta_i = \frac{(Xin_i - Xout_i)}{Xin_i} * 100 \qquad (3)$$

where Xin_i and $Xout_i$ are the means of EMCs or loads for the inlet and the outlets (N or S), respectively, for constituent i.

Seasons were determined on a solstice and equinox basis. Precipitation and temperature records for climate characterization were obtained using the Haw River 1E 313919 SCO NC station, located 1.6 km from the bioretention cells. Commonly, spring and summer are the wettest seasons and summer is the warmest season. During the monitoring period, the wettest months were June, November and July 2006 with 205, 160 and 145 mm, respectively; whereas, January 2007, February and March 2006 were the driest months with 20, 30 and 32 mm of rainfall, respectively. Concentration and load reductions were discussed on a seasonal basis. Due to the limited number of storms monitored, spring and summer were compared with winter and fall.

I.2.2. Statistical analyzes

Paired t-tests were performed with SAS 9.1.3 w/SPA^{TM} software to detect potentially significant differences between N and S cell outflow concentrations and between the inlet and the outlets. A level of significance (α) of 0.05 was considered. Concentration and load log-normality was assessed by Cramer-von Mises tests and probability plot observations. To avoid log-transformation problems, a 0.001 mg/L value (which is 5 times smaller than the lowest value recorded) was substituted for null values.

II. Results and discussion

II.1. Asphalt parking lot runoff nutrient quality characterization for eight sites (NC, USA)

II.1.1. Individual site water quality results

Table 4 shows average EMCs, loads and mass flow rates for each monitored site. Mean TP concentrations ranged from 0.07 to 0.33 mg/L and mean TN concentrations varied from 1.13 to 2.19 mg/L.

Table 4. Average EMCs (mg/L), loads (mg/m²) and mass flow rates (mg/(m²h)) per site.

Site code and variable value	Constituent					
	TP	OPO_4-P	TN	TKN	NH_4-N	$NO_{2,3}$-N
C						
EMC	0.20	-	1.83	1.37	0.39	0.42
Load	4.13	-	36.80	28.13	8.48	9.19
Mass flow rate	1.00	-	7.01	5.37	1.29	1.19
K1						
EMC	0.10	0.09	1.13	0.46	0.26	0.36
Load	0.78	0.70	8.12	3.28	1.96	2.77
Mass flow rate	0.04	0.04	1.04	0.19	0.27	0.37
K2						
EMC	0.07	0.05	1.14	0.38	0.30	0.37
Load	0.79	0.64	15.48	2.70	3.76	6.34
Mass flow rate	0.05	0.04	2.71	0.16	0.69	1.07
Gre						
EMC	0.18	0.05	1.57	1.29	0.36	0.28
Load	5.88	2.55	48.08	40.28	7.48	7.60
Mass flow rate	0.32	0.13	3.33	2.77	0.71	0.55
Go						
EMC	0.20	0.06	1.52	1.22	0.35	-
Load	1.39	0.19	9.98	7.67	1.92	-
Mass flow rate	0.10	0.01	0.99	0.66	0.13	-
L1						
EMC	0.33	0.20	1.84	1.48	0.27	0.36
Load	9.07	4.99	47.01	34.39	3.57	12.67
Mass flow rate	0.57	0.29	2.83	2.20	0.23	0.62
L2						
EMC	0.23	0.02	2.19	1.37	0.29	0.43
Load	3.43	0.23	43.93	29.07	5.08	5.04
Mass flow rate	0.11	0.01	0.88	0.63	0.09	0.10
Gra						
EMC	0.08	0.01	1.43	0.68	0.27	0.36
Load	0.91	0.19	11.69	8.40	2.10	3.51
Mass flow rate	0.13	0.01	1.44	1.06	0.20	0.36

Compared to values found in the literature, TP concentrations are in the range of published data on highway or roadway runoff TP concentrations; whereas, TN results are slightly lower (Table 5).

Table 5. The results of this study and previous studies average concentrations (mg/L) in urban, highway, roadway or asphalt parking lot runoff.

Reference	TP	OPO_4-P	TN	TKN	NH_4-N	$NO_{2,3}$-N
These 8 parking lots	0.21	0.07	1.63	1.24	0.32	0.36
Hope et al. 2004					0.8	26.6
					9.6	14.2
					6.7	3.4
Rushton 2001	0.106	0.044	0.556		0.133	0.273
	0.105	0.062	0.548		0.123	0.28
Barrett et al. 1995	0.42					1.25
	0.13					0.96
	0.1					0.36
Wu et al. 1996	0.14	0.10		0.88	0.22	
Wu et al. 1998	0.43	0.15			0.83	
	0.52	0.30			0.67	
	0.47	0.17			0.52	
Kayhanian et al. 2003	0.3	0.1		2	1.1	1.1
Kayhanian and Borroum 2000	0.7			5.2	1.9	1.8
Kayhanian et al. 2007	0.29	0.11		2.06		1.07
Irish et al.1995	0.41 (med)[a]					1 (med)[a]
	0.08 (med)[a]					0.73 (med)[a]
Brezonik and Stadelmann 2002	0.58		3.08	2.62		
Choe et al. 2002	1.96			6.76		
Driscoll et al. 1990		0.16 (med)[a]		0.87 (med)[a]		
U.S. EPA 1983	0.62 (res)[a]			2.85 (res)[a]		
	0.29 (com)[a]			1.50 (com)[a]		
	0.33 (med)[a]			1.50 (med)[a]		

[a] med = median, res and com are average EMCs for residential and commercial sites, respectively.

L1 and L2 sites had the highest means for TN, TKN and TP concentrations; whereas, the lowest concentrations for TN, TKN and TP were found for K1, K2 and Gra. This is consistent with the fact that L1 and L2 were not entirely impervious and were surrounded by a natural (park) land use. These sites may have received more nutrients than commercial sites probably due to the use of fertilizers or by plant material decomposition. TN and TKN highest loads were measured in C, Gre, L1 and L2 sites. These four sites were those which received the highest average rainfall depths during the monitoring period (Table 1). It seems that rainfall amount is a non-negligible source of nitrogen species. This trend is not as clear for phosphorus species. OPO_4-P and TP highest loads were found for L1; whereas, for the other sites, small loads with few differences were noted. It does not support a strong link between phosphorus inputs and rainfall amounts. Furthermore, the highest concentrations for OPO_4-P were found in L1; whereas, L2, Gre and K2 had the lowest OPO_4-P concentrations. As noted previously, L2 also had the highest value for TP average EMC. It suggests that the major form of phosphorus for L2 site runoff could be particulate-bounded phosphorus. Among the eight sites, NH_4-N and $NO_{2,3}$-N concentration and load variations were low.

All constituents considered, the Charlotte site presented the highest mass flow rate values which may be explained by this site also having the highest average rainfall intensity (Table 1). Similar to loads, L1 site also presented high values of TP, OPO_4-P, TN and TKN mass flow rates.

II.1.2. Descriptive statistics for EMCs, loads and mass flow rates for the set of the eight sites

The results of the descriptive statistics for the runoff quality for the eight sites combined are presented in Table 6.

Table 6. Descriptive statistics for EMCs (mg/L), loads (mg/m²) and mass flow rates (mg/(m²h)) for the eight sites.

Constituent	N	Mean	Median	SD	Range
EMC					
TP	116	0.19	0.12	0.21	0.005-1.40
OPO_4-P	74	0.07	0.04	0.10	0.005-0.64
TN	143	1.57	1.31	1.16	0.100-7.30
TKN	115	1.19	0.97	0.98	0.05-5.70
NH_4-N	141	0.32	0.22	0.35	0.01-2.10
$NO_{2,3}$-N	110	0.36	0.30	0.34	0.02-3.00
Load					
TP	113	4.01	1.48	7.20	0.026-60.84
OPO_4-P	72	1.89	0.36	5.69	0.015-44.25
TN	140	28.17	14.40	38.48	0.32-201.73
TKN	112	25.00	12.16	35.37	0.22-177.00
NH_4-N	139	4.86	2.11	6.87	0.07-38.72
$NO_{2,3}$-N	110	6.44	3.08	9.95	0.06-83.07
Mass flow rate					
TP	113	0.41	0.10	1.16	0.003-10.92
OPO_4-P	72	0.10	0.02	0.27	0.001-1.67
TN	140	2.90	1.16	5.83	0.032-41.91
TKN	112	2.37	0.80	5.10	0.016-35.42
NH_4-N	139	0.54	0.19	1.20	0.006-8.78
$NO_{2,3}$-N	110	0.55	0.21	0.82	0.007-4.07

Concentrations and loads for some pollutants varied up to four different orders of magnitude. TKN and, consequently, TN concentrations and loads presented the widest range of values. The ammonia mean concentration was nearly 0.32 mg/L, and the mean load did not exceed 4.86 mg/m², accounting for only 21 % of TKN mean load. Nitrate concentrations ranged between 0.02 and 3.00 mg/L and loads varied from 0.06 to 83.07 mg/m². It appeared that organic nitrogen (ON) was the major form of nitrogen in asphalt parking lot runoff. No TP and OPO_4-P concentration was greater than 1.40 mg/L and medians were 0.12 and 0.04 mg/L, respectively. TP and OPO_4-P median loads were 1.48 mg/m² and 0.36 mg/m², respectively. Half of the nutrient concentrations, concerning TP, TN and TKN, were found to be log-normally distributed; whereas, loads and mass flow rates for all constituents presented a log-normal distribution. Van Buren et al. (1997) studied parking lot runoff and found that EMCs of TP, NH_4-N and TKN were log-normally distributed. Mass flow rates were calculated to provide values easily comparable to other studies. Ranges for phosphorus species varied from 0.001 to 10.92 mg/m²/h; whereas, nitrogen species ranged from 0.006 to 41.91 mg/m²/h.

The first two studies in Table 5 relate specifically to parking lot runoff (Hope et al. 2004; Rushton 2001). NH_4-N and $NO_{2,3}$-N concentration results presented herein are similar to Rushton's results (2001) but are much lower than those found by Hope et al. (2004). This might be explained by the fact that the latter study was conducted after a dramatically long dry period enabling pollutants to accumulate. Like Rushton, this study was conducted in the Eastern U.S. Overall, taking into account the results from highways, roadways and urban runoff, the concentrations from the eight sites monitored for this study were slightly lower than those found in the literature. This excludes a few scattered values for TP (Barrett et al. 1995, Wu et al. 1996, Irish et al. 1995) and TKN (Driscoll et al. 1990). Only Rushton's results, obtained from a parking lot study, were lower than those presented herein. Nevertheless, except for the nitrate results reported by Hope et al. (2004), the maximum differences did not exceed one order of magnitude. It therefore appears that highway runoff TN concentrations are higher than parking lot TN concentrations. Conclusions are more difficult to draw for TP.

Similar to concentrations, loads found by Hope et al. (2004) for ammonia and nitrate were higher by one or two orders of magnitudes than the loads calculated in this study (Table 7).

Table 7. The results of this study and previous studies average loads (mg/m²) in urban, highway, roadway or asphalt parking lot runoff.

Reference	TP	OPO_4-P	TN	TKN	NH_4-N	$NO_{2,3}$-N
These 8 parking lots	3.88	1.70	29.37	23.35	4.86	6.43
Hope et al. 2004					11.5	156.3
					84.3	127.1
					57.1	28.7
Barrett et al. 1995	48					142
	13					98
	1					4
Wu et al. 1998	3.05	12			6.65	
	7.85	4.56			8.81	
	14.06	5.35			14.04	
Brezonik and Stadelmann 2002	1.9		10.3	9.6		
Choe et al. 2002	15			51.7		

Overall, most of the nutrient loads determined by previous studies were higher than the nutrient loads presented herein with the exception of a few values. However, load comparisons are only based on five studies and load units are not time-normalized.

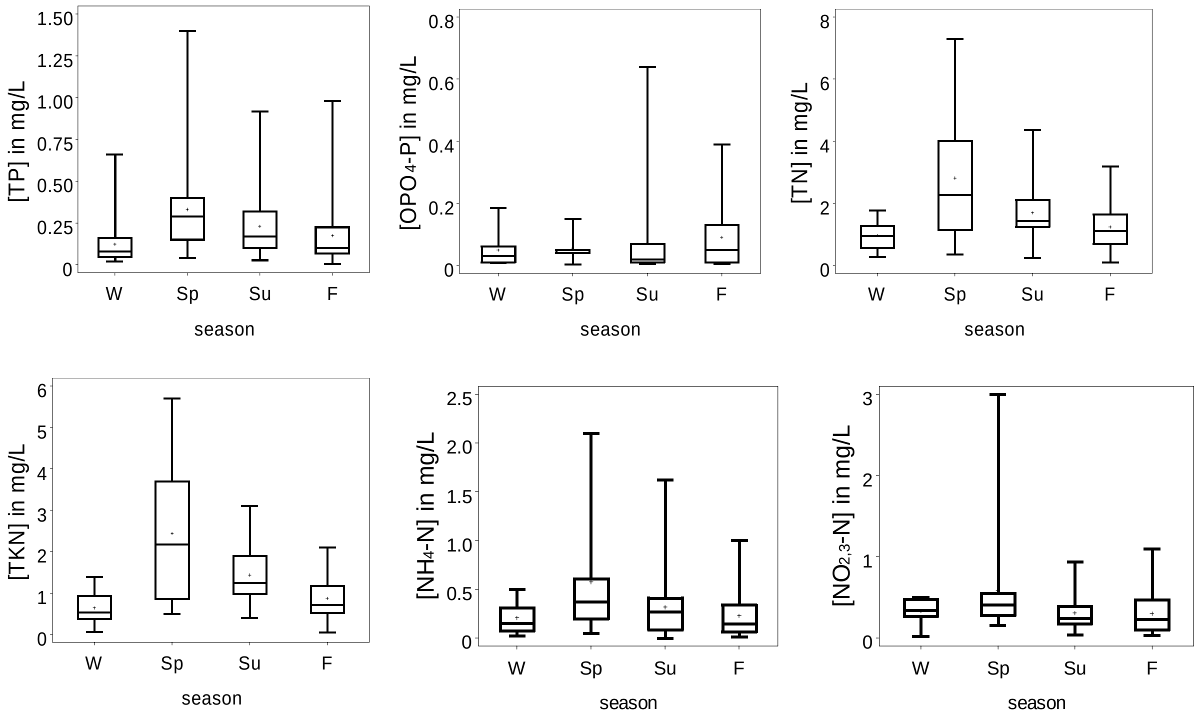
The major factor differentiating highway and parking lot sites is the average daily traffic (ADT). Comparing highway runoff quality results from the literature to asphalt parking lot results presented in this paper, there is a slightly difference between the concentrations and loads from these two types of sites, particularly for TN concentrations. This suggests that atmospheric deposition resulting from vehicle traffic may be rather localized. Higher concentrations of nitrogen in highway runoff than in parking lot runoff could be attributed to higher rates of oxide nitrogen gas emissions. Phosphorus concentration interpretation is more problematic, but as shown previously, the results presented herein are not systematically lower than those of highway values found in previous studies. It was previously shown that rainfall and atmospheric deposition accounted for a large part of runoff loads for nitrogen and phosphorus species (Irish et al. 1995; Wu et al. 1998; Hope et al. 2004; Rushton 2001; Barrett et al. 1995). Through a multiple linear regression analysis, Kayhanian et al. (2003) found that higher annual ADT produced higher $NO_{2,3}$-N, TKN and TP concentrations from highway runoff. In a recent study (Kayhanian et al. 2007), similar conclusions were drawn except that annual ADT had a negative effect on OPO_4-P concentrations. Barrett et al. (1995) reported the highest concentrations of $NO_{2,3}$-N and TP occurred at the highest traffic site of their study. Driscoll et al. (1990) showed that the effect of ADT was not significant.

Concentrations used in load prediction models are generally based on highway or roadway runoff quality data. It therefore seems that these models could overestimate TN, and in some cases, TP load predictions from asphalt parking lot runoff. The Tar-Pamlico load prediction model (NC DENR 2003), commonly used by engineers in North Carolina to choose and design BMPs, is based on 2.6 and 0.19 mg/L for TN and TP concentrations, respectively, for transportation catchments. A national model (STEPL model, Tetra Tech Inc. 2006) enables calculating loads for different kinds of watersheds by varying the annual rainfall amount but uses 3.0 and 0.5 mg/L for TN and TP concentrations, respectively, for all states. Other studies (NC DENR 2005) or models (PLOAD model, CH2M HILL 2006) recommend the use of similar values. These TN concentration values are greater than those found in this study for eight parking lot sites; whereas, the TP values are more similar (Table 4 and 6). A larger dataset for asphalt parking lot runoff nutrient quality should be studied to confirm these findings. However, all the eight sites, individually, showed that TN concentrations used in load prediction models are higher than those presented herein. Appendix I presents model estimation and measurement comparisons for concentrations and loads for each site and using the Tar-Pamlico river basin nutrient load exportation model. As the eight sites were not entirely impervious, average estimated concentrations used by the model, without considering land uses, were determined.

II.1.3. Seasonal variation

Figure 3 summarizes descriptive statistics for nutrient concentrations.

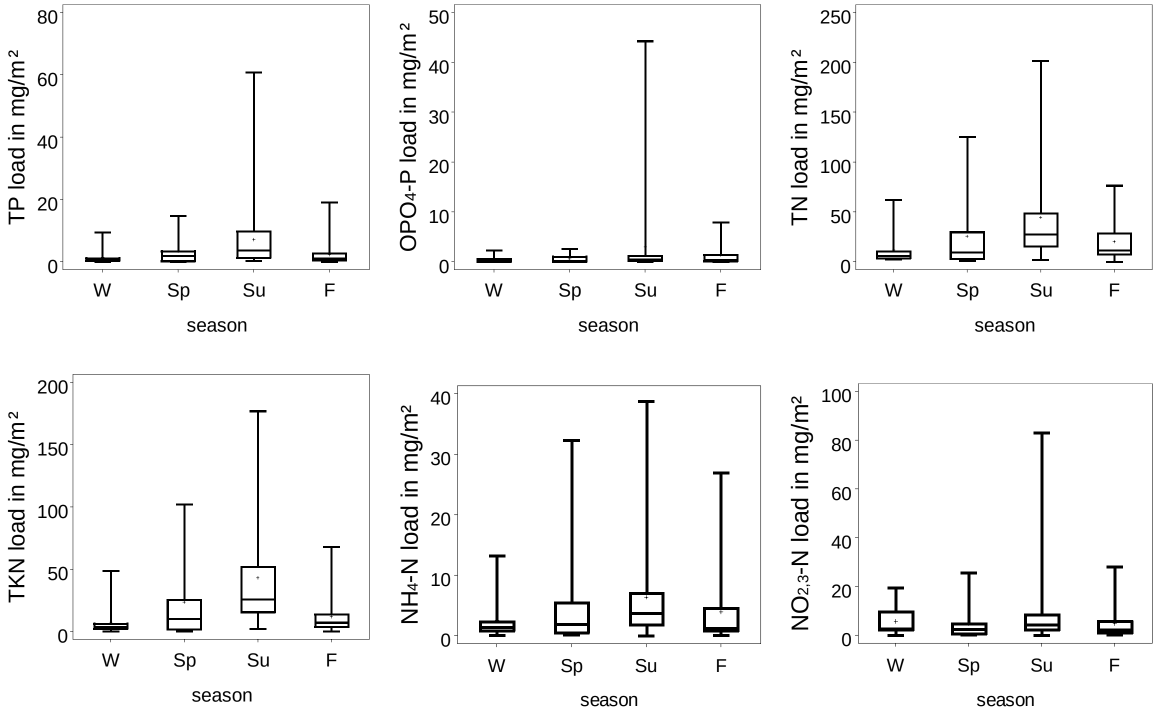
Fig. 3. Seasonal concentrations



Spring was the season during which the highest median concentrations and largest ranges were found for all nutrient forms except OPO$_4$-P. For that particular species, summer presented the widest range of values. Median concentrations of OPO$_4$-P were similar among seasons, ranging from 0.02 to 0.05 mg/L. Analyzing climate factors, it was found that the lowest values for rainfall depth, rainfall duration and antecedent dry period occurred in spring. Less frequent and smaller storm events of this season may explain that concentrations were less diluted, and therefore, high. The lowest median concentrations for OPO$_4$-P and NO$_{2,3}$-N occurred in summer; whereas, for all the other constituents, the lowest concentrations occurred in winter but they were not very different from fall median values. Summer, fall and winter were the wettest seasons with the highest values for rainfall duration and rainfall depth. These precipitation factors explain why concentrations were low for these seasons compared to spring values. Winter was the season showing the smallest range of concentrations. Overall, paired t-tests (α=0.05) showed significant differences between spring and both fall and winter concentrations for TN, TKN, NH$_4$-N and TP. Summer and winter concentrations were also significantly different for these four constituents. Summer concentrations were significantly different than those of fall for TN and TKN.

When examining all loads, summer was the season presenting the highest median loads and the largest ranges (Fig. 4).

Fig. 4. Seasonal loads



In parallel, summer rainfall depth for the eight sites was the highest, further supporting the fact that rainfall could be a major source of nutrients. Winter had the smallest ranges of loads and showed the lowest load medians for all constituents apart from OPO_4-P and $NO_{2,3}$-N, for which the lowest median concentrations were in spring and in fall, respectively.

Finally, summer TP, TN and TKN loads were significantly different (α=0.05) from fall and winter loads. Ammonia loads were only significantly (α=0.05) different between summer and winter seasons. No significant difference was found for nitrate; whereas, fall ortho-phosphorus loads were significantly different from those of winter.

The fact that higher pollutant loads and concentrations were observed in summer and spring bodes well for the use of biological stormwater treatment systems. Stormwater wetlands and bioretention cells, which both rely on biological activity for nutrient removal, have been shown to remove more pollution during the growing season, which includes spring, summer and most of fall in North Carolina (Hunt et al. 2006, Jing et al. 2001).

II.1.4. Correlation analysis

Nine predictive factors were considered for correlation and multiple linear regression analyses. Among the nine factors, four were related to the climate and five made reference to watershed physical characteristics or the main characteristic of the surrounding neighborhood. Three factors were log-transformed. Climate factors were rainfall depth (lnRfdepth, mm), duration (lnRfdur, h), average rainfall intensity (lnRfint, mm/h) and antecedent dry period (ADP, h). Watershed physical characteristics were catchment area (CatArea, m^2) and the percentage of asphalt within the drainage area (PerAsphalt, %). Main surrounding activity corresponded to three different major land uses referred to as commercial (com), residential (res) or natural/park (nat). Correlations were assessed among the nine predictive factors.

II.1.4.1. Correlations among the predictive factors

The three surrounding land uses (SLUs), commercial, residential and natural, were strongly correlated among themselves along with the percentage of asphalt and the catchment area. Rainfall duration had a significantly strong correlation with two other rainfall variables, lnRfdepth (0.532) and lnRfint (-0.695). Rainfall depth was also correlated with antecedent dry period (0.233) and average rainfall intensity (0.238). The percentage of asphalt within the watershed was strongly correlated with the catchment area (-0.654) which reflects that the largest watersheds of the eight sites were the least impervious ones (Table 1).

II.1.4.2. Predictive factor correlations with EMCs and loads

Correlations among runoff nutrient concentrations and the predictive factors were assessed. Table 8 summarizes the results for EMC and load correlation coefficients with the predictive factors.

Table 8. Pearson correlation coefficients for EMCs and loads

Constituent	ln Rfdepth (mm)	ln Rfdur (h)	ADP (h)	ln Rfint (mm/h)	CatArea (m2)	PerAsphalt (%)	com	res	nat
Ln(EMCs)[a]									
TP	-0.131	-0.161	-0.148	0.063	0.015	0.147	-0.062	-0.191*	0.210*
OPO$_4$-P	-0.136	-0.164	-0.079	0.046	-0.100	0.517*	0.036	-0.277*	0.106
TN	-0.272*	-0.213*	-0.024	0.013	0.147	-0.038	-0.086	-0.011	0.121
TKN	-0.226*	-0.133	-0.066	-0.045	0.039	0.149	0.049	-0.167	0.068
NH$_4$-N	-0.393*	-0.190*	0.003	-0.123	0.002	0.144	0.196*	-0.049	-0.190*
NO$_{2,3}$-N	-0.350*	-0.1852	0.074	-0.093	0.098	0.041	0.054	0.118	-0.178
Ln(Loads)[a]									
TP	0.587*	0.190*	-0.043	0.274*	0.060	0.156*	0.010	-0.218*	0.150
OPO$_4$-P	0.663*	0.222	-0.049	0.329	-0.038	0.373*	0.020	-0.202	0.085
TN	0.665*	0.330*	0.123	0.183*	-0.010	0.147	0.077	-0.272*	0.180*
TKN	0.590*	0.241*	0.023	0.222	-0.090	0.145*	0.079	-0.193*	0.055
NH$_4$-N	0.466*	0.287*	0.148	0.069	-0.094	0.288*	0.291*	-0.273*	-0.087
NO$_{2,3}$-N	0.601*	0.377*	0.272*	0.084	-0.118	0.247*	0.219*	-0.193*	-0.066

*Coefficient are significantly different from zero (α=0.05).

[a] Ln(EMCs) and Ln(Loads) are, respectively, log-transformed concentrations (mg/L) and loads (mg/m^2)

The strongest correlations were negative and found between rainfall depth or duration and all concentrations. Only correlations between phosphorous forms and lnRfdepth were not significant (α=0.05). Similar results were found previously on a mixed land use watershed runoff study (Brezonik and Stadelmann 2002). LnRfint and ADP did not present any significant correlation with concentrations and the relations were both positive and negative. Positive signs were found between every constituent concentration and catchment area, except for TP. Few significant correlations were found between concentrations and SLU, and the corresponding sign coefficients were both positive and negative. TP concentrations were negatively correlated to residential and commercial land uses and positively correlated to natural land use. This could be associated with plant decomposition and the application of fertilizers in Louisburg surrounding parks. For TP concentrations, Brezonik and Stadelmann (2002) also found a negative correlation with commercial land use but their results showed a positive relationship with residential land use.

When examining loads, results showed that rainfall depth followed by rainfall duration had the strongest positive significant correlations with all loads. Overall, ADP, lnRfint and PerAsphalt also presented a positive correlation with all loads but they were not all significant. Because load is based upon runoff volume and concentration, that loads are positively correlated with rainfall depth is expected. Apart from TN load, catchment area was both positively and negatively correlated with loads but correlations were not significant. Positive correlations were obtained with commercial land use but less than half of the coefficients were significant. Results for residential land use showed significant negative correlations with loads as was previously found by Brezonik and Stadelmann (2002). Nevertheless, this is the result of a study that only includes one residential SLU site (Gra) and should consequently be confirmed by further research.

II.1.5. Multiple linear regression analysis

II.1.5.1. MLR with groups of variables

Three groups of variables were created and tested as predictors to evaluate their global influence on nutrient models for concentrations and loads. The group named as "climate" included lnRfdepth, lnRfdur, lnRfint and ADP. PerAsphalt and CatArea accounted for "physical characteristics" group, and res, com and nat belonged to the "SLU" group. It was shown that the collective contribution of climate variables was a significant predictor for most concentrations and all loads. Physical characteristics significantly predicted the concentrations of OPO_4-P, TN and TP and all loads except for $NO_{2,3}$-N. Finally, SLU appeared as a poor predictor being significant only for OPO_4-P concentration prediction and NH_4-N, OPO_4-P and TN load predictions.

The following general model was tested:

LnC_i or $LnL_i = b_1*\text{lnRfdepth} + b_2*\text{CatArea} + b_3*\text{ADP} + b_4*\text{PerAsphalt} + b_5*\text{lnRfdur} + b_6*\text{lnRfint} + b_7*\text{com} + b_8*\text{res} + b_9*\text{nat}$. (3)

Where:

LnC_i and LnL_i are, respectively, the log-transformed concentration and load of constituent i

b_i are the parameter coefficients (standardized values) computed by the SAS models.

II.1.5.2. MLR results for nutrient concentrations

Table 9 presents the b_i coefficients obtained from the MLR analysis. For each MLR model, between three (for TN, TKN) and six (for $NO_{2,3}$-N) factors showed statistically significant coefficients, indicating that some of the nine factors used in the models for EMC predictions are weak. R-squared values ranged between 0.195 and 0.499. The low R-squared values demonstrate that the models could probably be improved with the inclusion of other independent factors. Some of the factors were included despite not being significant. However, their inclusion slightly increased the R-squared value.

Table 9. MLR standardized coefficients for EMC and load predictions

Constituent	ln Rfdepth (mm)	ln Rfdur (h)	ADP (h)	CatArea (m2)	PerAsphalt (%)	com	res	nat	R^2
Ln(EMCs)[a]									
TP	-0.102	-0.209	-0.143	0.294	0.320	-0.425*	-0.444*		0.243*
OPO_4-P	-0.069	-0.158	-0.006	0.023	0.930*		0.292	0.605*	0.499*
TN	-0.284*	-0.167	-0.032	0.606*	-0.239	0.917*		0.734*	0.243*
TKN	-0.262*	-0.027	-0.072	0.560*	-0.016	0.918*		0.804*	0.212*
NH_4-N	-0.466*	0.046	0.052	0.369*	-0.008	0.534		0.271	0.235*
$NO_{2,3}$-N+0.02	-0.452*	0.125	0.152	0.030	0.303	-0.372		-0.288	0.195*
Ln(Loads)[a]									
TP	0.596*	-0.049	-	0.552*	0.095	0.845*	-	0.762*	0.511*
OPO_4-P	0.636*	0.008	-	0.241	0.674*	-	0.051	0.247	0.676*
TN	0.651*	-0.080	-	0.594*	-0.256	1.253*	-	0.829*	0.608*
TKN	0.563*	0.051	-	0.694*	-0.083	1.207*	-	0.861*	0.538*
NH_4-N	0.425*	0.068	-	0.358*	0.372*	-	-0.237*	-	0.356*
$NO_{2,3}$-N	0.510*	0.135	0.165*	0.056	0.319	0.142	0.144	-	0.478*

*Coefficient are significantly different from zero (α=0.05).

[a] Ln(EMCs) and Ln(Loads) are respectively log-transformed concentrations (mg/L) and loads (mg/m^2)

From the results, it appeared that lnRfdepth had negative coefficients for all constituent concentrations. These coefficients were significantly different from zero except for both phosphorus species. Therefore, a decrease in the concentration would be expected for every increase in the rainfall depth of a storm event sustaining the idea of a diluting effect of the pollution associated with big storms. A similar tendency was previously found for other urban watersheds (Brezonik and Stadelmann 2002; Kayhanian et al. 2003; Kayhanian et al. 2007). Generally, only one or two rainfall factors were included in the models and only one (lnRfdepth generally) was significant. Indeed, as shown previously, some factors were correlated, such as with lnRfdepth, lnRfint and lnRfdur that were not significant simultaneously. Moreover, it should be noted that substituting lnRfint to lnRfdur had no effect on the R-squared. LnRfdur always presented a higher standardized coefficient than lnRfint. The former was therefore introduced into the MLR models in lieu of lnRfint. Low standardized coefficients were found for antecedent dry period predictor. They were not significant and the relationships were both positive and negative. Some authors observed a positive relation between concentrations of $NO_{2,3}$-N, TP, TKN and antecedent dry period (Kayhanian et al. 2003; Kayhanian et al. 2007) whereas Brezonik and Stadelmann (2002) found a negative slope between dissolved phosphorus and the number of days since last event. Similar to these studies a negative, but non-significant, relationship was also found herein between OPO_4-P concentration and ADP; whereas, positive non-significant slopes were observed for NH_4-N and $NO_{2,3}$-N.

Half of the catchment area coefficients were significantly different from zero for TN, TKN and NH_4-N concentrations. Positive signs were found for all constituents. This could seemingly indicate that a greater drainage area involves a higher concentration of these nutrients in the asphalt runoff. However, this needs to be linked to the fact that among the eight sites, large watersheds were associated with more grassed surfaces where fertilizers are suspected to have been applied. Brezonik and Stadelmann (2002) analyzed data from mixed land use watersheds and only found a significant negative coefficient between TP concentrations and drainage area from a MLR analysis. From highway runoff studies, Kayhanian et al. (2003) found both positive (for $NO_{2,3}$-N and TP) and negative (for ortho-phosphorus) coefficients between concentrations and drainage area. In a recent study on highway stormwater runoff quality, Kayhanian et al. (2007) showed that catchment area was a poor factor to predict nutrient EMCs. Positive coefficients were found for the percentage of asphalt within the watershed except for TN, TKN and NH_4-N concentrations. It indicates that concentrations increase when increasing the percentage of impervious surface. Larger percentages of impermeable surfaces are assumed to be able to accumulate more pollutants but also to generate more runoff as infiltration is extremely reduced. With a higher percentage of asphalt, on the one hand, higher runoff volumes are produced and, on the other hand, more pollutants are accumulated. With both runoff volume and pollutant accumulation increasing, concentration evolution is not intuitive. The results of this study tend to show that the rate of pollutant build-up outweighs the rate of runoff production. However, only the coefficient for OPO_4-P was significant and three negative signs were also found. In addition, coefficient values were low. Therefore, the percentage of asphalt was not a major predictor of nutrient concentrations.

Surrounding land uses were found to be poor predictors of concentrations. Both positive and negative signs were found for the three SLU for most of the constituents. Furthermore, only half of the coefficients were significant. For residential land use, only the negative coefficient for TP was significant. A negative and significantly different from zero coefficient was found for commercial land use to predict TP concentrations and a positive and significant coefficient was found for TN.

Natural SLU presented four positive and one negative coefficients. Three of the positive coefficients were statistically significant. It seems to be the strongest SLU predictor. However, only two sites (L1 and L2) had a natural (park) SLU to which fertilizers had been applied. It could explain the results found herein. SLU predictors were previously shown to affect runoff pollution (Brezonik and Stadelmann 2002). Nevertheless, a general trend is still difficult to determine from these results.

II.1.5.3. MLR results for nutrient loads

Load predictions by MLR analysis were also conducted. Generally, between one and five factors were significant in the MLR models and R-squared values were higher than in EMCs models, ranging from 0.356 to 0.676.

MLR analysis showed that coefficients for rainfall depth were significantly different from zero and positive for all loads. Rfdur coefficients were low and not significant. TP and TN coefficients presented negative relationships; whereas, the other four constituents had a positive relation with Rfdur. This is probably due to Rfdur correlation with rainfall depth which prevents both factors from being significant and strong predictors simultaneously. Rfdepth positive signs suggest that in addition to the diluting effect showed previously, higher rainfall amounts were associated with higher nutrient loads. This is probably partly due to a pollutant build-up and wash-off effect but could also be attributed to the fact that rainfall is generally a major source of nutrients in urban areas (Rushton 2001; Wu et al. 1998). Driscoll et al. (1990) showed that precipitation volume was the climate factor having the strongest effect on pollutant loading; whereas, rainfall intensity or duration were not significant. The nitrate load prediction model included ADP with a significant and positive coefficient. This suggests that a longer dry period before a storm event could enable these pollutants to be accumulated on asphalt impervious surfaces prior to being washed-off during the next storm. Brezonik and Stadelmann (2002) also found a positive correlation between days since last event factor and both TKN and TN loads for mixed land use watersheds.

Generally, catchment area and the percentage of asphalt predictive factors generated positive and significantly different from zero coefficients. Brezonik and Stadelmann (2002) found similar results for rainfall depth and drainage area coefficients to predict the loads of TP, TKN and TN. For the eight sites included in this study, the sites with the largest catchment area were associated with natural and residential land uses and had relatively lower fractions of impervious area. The loads are normalized per catchment areas; therefore, the effect of this predictor could reflect either the SLU or the percentage of impervious area. In addition, commercial and natural land uses showed positive coefficients, most of them being significant, to predict most of the constituent loads.

Coefficients for commercial land use were stronger than for natural surrounding land use. The use of fertilizers on surrounding parks or onto grass drainage areas, and plant material decomposition could partly explain that constituent loads increase with natural surrounding land uses and higher catchment areas. One negative and two positive coefficients were found with residential land use but only one was significant making an interpretation of residential land use effect on nutrient loads difficult. The lack of heterogeneity for SLU predictor among the eight sites probably influenced these results and prevented consistent conclusions from being drawn.

II.2. Field grassed bioretention cell efficiencies to remove urban stormwater runoff pollutants

II.2.1. Hydrology

Inflow volumes, the sum of direct rainfall and runoff volumes, were not significantly lower than the sum of N and S outflow volumes. However, for most of the events, the bioretention cells reduced the volumes between the inlet and both outlets. This difference is attributed to water storage in the fill media, exfiltration below the underdrains and evapotranspiration. Surprisingly, in some cases, a volume increase occurred. For the first 13 storm events, volume increases through the bioretention system were probably due to the inaccuracy of inflow volume estimations rather than the contribution of the water table which was found to be low. On April 17, 2007, a 60.2 mm rainfall event may have overwhelmed the inflow monitoring system causing an under-estimating of total inflow volumes.

Overall, 2-minute peak inflows were reduced by the bioretention cells probably due to water storage in the fill media, which act to dampen flow rates. However, outflow rates were calculated for only 10 (N) and 13 (S) storm events. For three storm events which occurred in March and April 2007, outflow peaks were higher than inflow peaks. For short antecedent dry period, some water still remaining in the fill media could have been flushed out of the system. For big storm events (e.g. the 60.2 mm of April 17, 2007), an under-estimation of inflow peak could have occurred. Therefore, low (3 (N) and 15 (S) %) average peak reductions were found (Appendix VI).

II.2.2. Water Quality

Table 10 summarizes water quality and statistical results for the inlet and the two outlets of the bioretention cells. Appendix VII presents all the results for each storm event.

Table 10. Influent and effluent EMCs (mg/L) and loads (mg/m^2) and significant differences

	Average EMCs or Loads[a]				Inlet	Significant differences			%reduc	
Constituent		IN	N	S	Log-normality	Inlet/N	Inlet/S	N/S	N	S
TKN	EMCs	0.683+/-0.33	0.368+/-0.08	0.452+/-0.05	Log	Yes (p=0.0057)	No	No	46	34
	Loads	8.400+/-8.59	5.211+/-6.38	3.397+/-6.18	Log	Yes (p=0.0027)	Yes (p<0.0001)	No	38	60
NH_4-N	EMCs	0.268+/-0.25	0.079+/-0.06	0.055+/-0.05	Log	Yes (p=0.0001)	Yes (p=0.0039)	Yes (p=0.0448)	71	80
	Loads	2.102+/-3.02	0.463+/-0.53	0.325+/-0.61	Log	Yes (p<0.0001)	Yes (p=0.0002)	Yes (p=0.0192)	78	85
$NO_{2,3}$-N	EMCs	0.363+/-0.17	0.252+/-0.17	0.384+/-0.19	Not log	Yes (p=0.0074)	No	No	31	-6
	Loads	3.507+/-3.25	2.029+/-3.68	4.195+/-8.88	Log	Yes (p=0.0088)	No	No	42	-20
TN	EMCs	1.429+/-0.82	0.634+/-0.25	0.757+/-0.29	Log	Yes (p<0.0001)	Yes (p=0.0265)	No	56	47
	Loads	11.690+/-11.97	4.592+/-8.18	7.607+/-14.87	Log	Yes (p=0.0007)	Yes (p=0.0057)	No	61	35
TP	EMCs	0.083+/-0.05	0.055+/-0.03	0.058+/-0.02	Log	No	No	No	34	31
	Loads	0.912+/-0.84	0.710+/-0.81	0.420+/-0.68	Log	No	Yes (p=0.0014)	Yes (p=0.0442)	22	54
OPO_4-P	EMCs	0.010+/-0.004	0.014+/-0.003	0.015+/-0.001	Log	No	No	No	-35	-45
	Loads	0.188+/-0.20	0.260+/-0.37	0.114+/-0.22	Log	No	No	No	-38	39
FC	Concentration	4172+/-7141	125+/-171	646+/-849	-	-	-	-	97	85

[a] EMCs are in mg/L, load units are mg/m^2 and FC units are colonies/100mL. Results present mean+/- standard deviation.

II.2.2.1. Nutrients

NH_4-N, $NO_{2,3}$-N and TN concentrations and loads were analyzed for 20 and 18 storm events, for the N and S cells, respectively; whereas, TKN and TP results were both based on 7 (N) and 10 (S) storms and only 5 storms were analyzed for OPO_4-P. Apart from $NO_{2,3}$-N EMCs, all nutrient EMCs and loads were log-normally distributed (α=0.05). The higher load removal rates were found in fall and winter for the south cell, except for NH_4-N. Generally, load exports corresponded to relatively higher outflow volumes.

Ammonia EMC removal rates were 71 and 80 % and load reductions were 78 and 85% for the N and S cells, respectively. Mean effluent EMCs and loads were significantly lower (0.079 (N) and 0.055 (S) mg/L) than those measured in the runoff (0.268 mg/L). If the fill media had had a high organic matter content, the media could have been a source of ammonia created by OM decomposition. This was not noticed in this study which could indicate that the soil had a low OM content. This bodes well for fill media mainly comprised of stalite. NH_4-N was probably removed by nitrification suggesting that there was room for aerobic biological processes to take place into the fill media. This probably occurred in the top portion of the fill media which had less frequent and shorter saturation periods, because the top was not impacted by the ISZ. These results were higher than those found by Hsieh and Davis (2005) who reported a maximum of 49% ammonia EMC reduction in a field bioretention cell, but they are similar to those found studying large pilot bioretention systems (Davis et al. 2001) (79 %) and a field raingarden monitored by Dietz and Clausen (2005) (84.6 %).

Effluent TKN EMCs (0.368 (N) and 0.452 (S) mg/L) and loads were significantly lower than runoff TKN EMCs (0.683 mg/L). Average TKN concentration reductions were nearly 46 (N) and 34 (S) %, and TKN load removal rates were 38 (N) and 60 (S) %. Moreover, median TKN EMCs (0.385 (N) and 0.459 (S) mg/L) were one order of magnitude lower than those presented in the international BMP database (GeoSyntec Consultants and Wright Water Egineers, Inc. 2006). These results were based on storm events occurring in fall, winter and spring. It has been shown that the highest asphalt parking lot nutrient concentrations occurred in spring and summer in North Carolina. However, summer is a warm and humid growing season during which higher removal rates are expected thanks to increased microbial degradation and plant up-take (Hunt et al. 2006). Annual TKN removal rates could therefore be higher than those presented herein. Previous studies found moderate TKN EMC and load removal rates ranging from 50 to 65% (Davis et al. 2006, Birch et al. 2005); whereas, other studies' results were lower (Hunt et al. 2006). TKN reduction is partially due to the high NH_4-N conversion to $NO_{2,3}$-N. In addition, the results showed that organic nitrogen concentrations slightly decreased by 30 (N) and 4 (S) %. It indicates that little ammonification, generating additional NH_4-N, took place into the soil.

Average removal rates of $NO_{2,3}$-N concentrations and loads were approximately 31 and 42 %, respectively, for the N cell, leading to significantly lower effluent EMCs (0.252 mg/L) and loads than those of the influent (0.363 mg/L). However, on average, the S cell exported 6% (EMCs) and 20 % (loads) more $NO_{2,3}$-N than was present in the runoff. Higher removal rates were found for the

N cell. This may be partly attributed to N cell in-situ soil (clay) which can provide a wetter fill media than the S cell (sandy loam in-situ soil) thus being more likely to retain anaerobic conditions for longer periods of time. In fall and winter, $NO_{2,3}$-N outflow concentrations were higher (0.43 mg/L) than those of the inflow (0.27 mg/L) for the S cell and little reduction (4.2%) occurred in the N cell. These two seasons were drier and cooler compared to spring and summer, thus having slower rates of microbial activity and expected denitrification processes. Moreover, nitrate reduction by denitrification requires organic matter as an energy source for microorganisms and an anaerobic environment. It was previously suggested that little organic matter was expected to be present into the fill media. The presence of induced ISZs did not appear to provide a sufficiently long contact time under fall and winter climatic conditions for denitrification to occur at a higher rate than OM decomposition and ammonia nitrification that produce additional $NO_{2,3}$-N. Hunt et al. (2006) found 75 and 13 % nitrate concentration reduction in two conventionally drained bioretention cells with high and low OM content, respectively. Kim et al. (2000) studied nitrate reduction through column experiments including a submerged zone. These experiments were conducted at 22°C with different types of electron donors and showed the importance of the presence of an easily metabolizable carbon source for nitrate removal. Several laboratory (Davis et al. 2001 and 2006) or field studies (Dietz and Clausen 2005, Hsieh and Davis 2005, Birch et al. 2005 and 2006) also found little nitrate removal and in some cases nitrate exportations by bioretention systems. During spring and summer, higher removal rates were found with averages of 41 and 18 % for the N and S cells, respectively. Median warm season effluent $NO_{2,3}$-N EMCs (0.250 (N) and (0.401 (S) mg/L) were lower than those of the international BMP database (GeoSyntec Consultants and Wright Water Engineers, Inc. 2006). This indicates that the bioretention cells, probably partly due to the presence of ISZs, were able to provide acceptable effluent quality.

TN concentration average removal rates were 56 (N) and 47 (S) % and mean effluent EMCs were 0.63 (N) and 0.76 (S) mg/L. Median effluent concentrations (0.68 (N) and 0.78 (S) mg/L) were in the range of TN median EMCs presented in the international stormwater BMP database (GeoSyntec Consultants and Wright Water Egineers, Inc. 2006). EMCs and loads from the underdrains were significantly lower than those of TN influent. As expected, similar to $NO_{2,3}$-N, spring and summer TN EMC removal rates (nearly 64% for both cells) were greater than fall and winter values (28 (N) and -3 (S) %). TN reduction is mainly explained by TKN capture and $NO_{2,3}$-N conversion in the N cell.

No samples were analyzed for phosphorous species during the summer. Average TP EMC reductions were close to 30 % for all seasons and mean effluent concentrations were 0.055 (N) and 0.058 (S) mg/L. Phosphorous removal occurs in the top soil layer which was essentially the same in the N and S cells. This is the reason why effluent concentrations were so similar. Median TP concentrations (0.049 (N) and 0.056 (S) mg/L) were one order of magnitude lower than those presented in the international stormwater BMP database (GeoSyntec Consultants and Wright Water Engineers, Inc. 2006). It therefore shows that even with relatively clean influent (0.083 mg/L), the bioretention cells were able to sequester TP. OPO_4-P concentrations increased nearly systematically between the inlet and both outlets with mean removal rates of -35 and -45% for the N and S cells, respectively. However, effluent mean OPO_4-P EMCs remained relatively low (0.014 (N) and 0.015 (S) mg/L). Therefore, little phosphorous desorption from the fill media occurred which is mainly due to its probable low P-Index. Particulate bounded phosphorous therefore appeared to be moderately retained into the filtration media.

Fall and winter correspond to the decay period for vegetation. Grass mowing and decomposition could have released previously up-taken nutrients thus contributing to the relatively low $NO_{2,3}$-N and OPO_4-P reductions.

Table 11 summarizes previous studies outflow concentrations for TN and TP found for commonly vegetated (trees, shrubs and mulch) bioretention cells in North Carolina. Comparing to these studies, Graham cell outflow concentrations were low suggesting that grass seems to be at least as able as mulch and vegetation to reduce nutrient species.

Table 11. Comparison of effluent concentrations from select studies in North Carolina[1]

	Graham N	Graham S	Charlotte[2]	Louisburg 1[3]	Louisburg 2[3]	Greensboro 1[4]	Greensboro 2[4]
TN	0.634	0.757	1.14	1.32	1.16	4.38	5.23
TP	0.055	0.058	0.13	0.24	0.25	0.56	3.00

[1] All cell examined, except those in Graham, were vegetated by trees and shrubs and covered in hardwood mulch.

[2] Hunt et al. 2007, submitted

[3] Sharkey 2006

[4] Hunt et al. 2006

Overall, apart from NH_4-N, the results showed that the N cell was slightly more efficient than the S cell to reduce concentrations of nitrogen (14 storm events) and phosphorous (6 storm events) species. This is counter-intuitive with the fact that the N cell fill media depth is lower than that of the S cell. Bioretention depth was demonstrated to be an important parameter of phosphorous removal (Davis et al. 2001) under approximately 60 to 80 cm (Davis et al. 2006). However, overall, N outflow concentrations presented herein were not significantly different (α=0.05) from S outflow concentrations. Moreover, it should be noted that fewer samples were collected from the N cell during the fall 2006 due to the absence of outflow or because of equipment breakdowns. When considering S cell results, fall and winter effluent nutrient concentrations were higher than those of the influent. It is expected that similar results of nutrient exportations could have occurred for the N cell in fall 2006 thus reducing the efficiencies reported herein for this cell. Nevertheless, when only considering the storms producing water quality results for both cells, it appeared that S cell efficiencies were generally lower than those found for the N cell. Therefore, it seems that a 0.75 m depth above a clayey soil, could provide more nutrient removal than a 1.05 m depth placed on a sandy loam soil.

II.2.2.2. Fecal coliforms

Fecal coliform grab samples were collected during 7 and 4 storm events for the N and the S cells, respectively. Inflow concentrations ranged from 220 to more than 20000 col/100 mL and effluent concentrations ranged from 2 to more than 1890 col/100mL. Removal rates were high ranging from 13 to nearly 100% for both cells and the average fecal coliform concentration reductions were 95 (N) and 85 (S) %. However, outflow concentrations exceeded the state limit of 200 col/100 mL for 3 of the 4 samples for the S cell and for one of the 6 samples of the N cell. Fecal coliform die-off is expected with sunlight and dry soils. The grassed cells, contrary to commonly vegetated bioretention systems, were nearly never shaded. Moreover, the cells rapidly drained but in spite of the presence of an ISZ that maintained humid conditions, effluent fecal coliform concentrations were quite low. Few studies assessed bioretention cell efficiencies to remove FC. Some published results also indicated moderate (Birch et al. 2006) to high (Birch et al. 2005, Hunt et al. 2007) removal rates of FC. Due to the limited number of samples, there were no statistically significant findings of fecal coliform reduction. This shows promising results for the ability of bioretention cells in fecal coliform reduction, and pathogenic bacteria as a whole.

Conclusions

Nitrogen species concentrations and loads determined from asphalt parking lot runoff from eight sites in North Carolina were lower than those found in previous studies on urban or highway runoff; whereas, phosphorus EMCs and loads were similar between parking lot and highway runoff. This indicates that traffic may be a key factor in nitrogen runoff input, but probably is not for phosphorous species. As a result, load prediction models, commonly based on highway or roadway nutrient concentrations, can therefore be expected to over-estimate nitrogen loads. Overall, seasonal differences were significant between spring and fall or winter for concentrations and between summer and fall or winter for loads. Spring and summer presented the highest values of concentrations and loads, respectively, for nearly all nutrient species.

Correlation and multiple linear regression analyses showed that some of the nine factors, lnRfdepth, lnRfdur, lnRfint, ADP, PerAsphalt, CatArea, res, com and nat, were not independent. Hence, they were not all included in concentration or load prediction MLR models. In addition, they were not all significant simultaneously. It was highlighted that two groups of factors (climate and physical characteristics) best predicted concentrations and loads; whereas, SLU variable effects were more difficult to determine. Overall, rainfall depth, catchment area, the percentage of asphalt and natural land use were good predictors for nutrient EMCs and loads. Adding or removing other of the predictive factors presented herein generally did not noticeably improve the models. From these eight sites, runoff quality results supported the idea of a diluting effect and a pollutant build-up and wash-off tendency on impervious surfaces for nutrients. This phenomenon is partly attributed to higher rainfall depths, which could be a non-negligible source of nutrients, and higher percentages of imperviousness. A longer antecedent dry period effect was not very clear, but it was found to positively affect some EMCs and nitrate loads. Moreover, it appears with these parking lots that the rate of pollutant accumulation onto an asphalt parking lot was greater than the rate of runoff generation. The impacts of residential and commercial surrounding land uses on parking lot runoff quality were unclear. A wider assortment of surrounding land uses among the sites tested would probably have improved the assessment of this factor.

These preliminary results showed that, despite the likely source of nutrients due to grass decay and mowing, both N and S grassed bioretention cells effectively removed nutrient species when considering either EMC reductions or effluent quality. However, it was found that OPO_4-P and $NO_{2,3}$-N effluent EMCs and loads were higher than those of the influent, particularly during fall and winter. Fecal coliform showed promising removal rates but some effluent concentrations were higher than the maximum permissible state limit. Overall, volumes and peak flows were reduced through the bioretention systems. However, this was not systematic, and it should be confirmed by the addition of new data. The ISZ seemed to improve denitrification particularly during the warm and humid seasons. However, under fall and winter climatic conditions, this effect is less evident and a higher contact time could probably provide better $NO_{2,3}$-N removal. The importance of the underlying, in situ soil needs to be further explored. Shallower cell (N, 0.75 m fill media depth) overlaying a tighter clay soil provided better pollution abatement than a 1.05 m deep (S cell) overlaying a sandy loam in situ soil. It is probable that the ISZ in the former zone stayed saturated, and therefore anaerobic, longer explaining why $NO_{2,3}$-N concentrations were substantially lower leaving the N cell. The probable lack of an anaerobic zone may explain why influent and effluent $NO_{2,3}$-N concentrations were roughly the same for the S cell. Increasing the media depth from 0.75 to 1.05 m did not improve water quality. The system of a grass vegetated bioretention cell with an expanded slate fill media containing ISZs performed favorably to any other conventionally vegetated (trees, shrubs and mulch) bioretention cell studies in North Carolina.

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APPENDICES

Table of appendices

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Appendix I: Simple Method and Tar-Pamlico river basin model (North Carolina, USA)

I.1. Simple Method

The Simple Method (Schueler 1987) is an empirical way to estimate annual stormwater loadings exported by small urban watersheds. Little information is required for this method: the precipitation amount (P), the percentage of imperviousness (I) of the drainage area and the average event mean concentration (C) of a pollutant during a storm event.

The formula is as follows:

L = P*Pi*Rv*C*0.227
where:
L is the nutrient load (lbs/ac/yr)
P is the average annual rainfall (45 in/yr for the Piedmont region of North Carolina, 50 in/yr for the Coastal Plain region of North Carolina).
Pi is a correction factor for storms with no runoff (0.9). This coefficient takes into account the fact that the water from some small storm events will only be captured by soil depressions and evaporated without producing any runoff.
Rv is the runoff coefficient equal to 0.05 + 0.9*I’ (where I’ is the fraction of imperviousness from 0 to 1). This equation was determined by Schueler (1987) after plotting the runoff coefficient (the ratio between the runoff depth and the rainfall depth) against the percentage of imperviousness in 44 small urban catchments monitored during the national NURP study.
C is the flow-weighted EMC in lbs/ac/yr.
0.227 is a unit conversion factor.

I.2. Tar-Pamlico river basin nutrient export model (North Carolina, Piedmont region)

The Tar-Pamlico river basin nutrient export model applies the Simple Method for land uses categorized as “transportation impervious”, “roof impervious”, “managed pervious” or “wooded pervious” within a development site.

The only “variable” that needs to be determined is C. Proposed values were based on highway and roadway EMC analysis for urban watersheds. For transportation impervious, TN and

TP average EMCs are 2.60 and 0.19 mg/L, respectively.

The Excel spreadsheet of the Tar-Pamlico Piedmont region model is reproduced below (table I.1) using input values (areas) for the Graham site.

Table I.1. Excel spreadsheet for the Tar-Pamlico river basin (applied to Gra site)

(1)	*(2)*	*(3)*	*(4)*	*(5)*	*(6)*	*(7)*
Type of Land Cover	**Area** (acres)	**S.M. Formula** (0.46 + 8.3I)	**Average EMC of TN** (mg/L)	Column (2) * (3) * (4)	**Average EMC of TP** (mg/L)	Column (2) * (3) * (6)
Transportation impervious	0.63	3.64	2.60	5.98	0.19	0.44
Roof impervious	0.04	3.64	1.95	0.31	0.11	0.02
Managed pervious	1.09	3.64	1.42	5.63	0.28	1.11
Wooded pervious	0.00	3.64	0.95	0.00	0.14	0.00
Fraction Impervious (I) =	0.38		**TN Loading (lb/yr) =**	11.92 *(8)*	**TP Loading (lb/yr) =**	1.56 *(8)*
Total Area of Development =	1.77 *(10)*		**TN Exp. Coeff. (lb/ac/yr) =**	6.75 *(9)*	**TP Exp. Coeff. (lb/ac/yr) =**	0.89 *(9)*

Column (3) calculates the runoff amount using the Simple Method (S.M.) as follows:

Runoff = P*Pi*Rv*0.227

= 45*0.9*(0.05+0.9*I)*0.227

= 0.46+8.3*I

Columns (4) and (6) are input average EMCs used by this model.

Columns (5) and (7) are the calculations of load exportations for each land use:

L = runoff*area*C.

Cells (8) are the sums of the previous cells for the same column and finally give the total load exportations in pound/year. The last two cells (9) are load exportations normalized by the total area of the watershed (cell 10).

I.3. Model application: comparisons between model estimations and measured values for EMCs and loads

I.3.1. Concentration comparisons

EMC measurements, given by the laboratory, take into account the whole watershed, with its different land uses. In the load prediction models, there is an EMC input value for each type of land use within the watershed. To compare EMC measurements to EMC inputs of the Tar-Pamlico model, it has been chosen to calculate an average EMC used by the model, without considering differences between land types. For sites where there is only one land use, then this average input EMC would be the one of this particular land use input EMC.

To compare both measured and estimated concentrations, a mean concentration of TN and TP is determined using the model ("EMCe"). This is based on the concentrations used in the model for each of 4 land uses. For this analysis, runoff volumes must be calculated for each land cover.

The model calculates the loads for each land cover. Therefore, it indirectly includes a runoff volume calculation for each land cover. The loads are given for one year because the calculation involves the use of an average annual precipitation depth.

The calculations were performed as follows:

1) Annual runoff volume (Vroi) determination for land use i:

$$Vroi = \frac{Lei}{EMCei} * 453592 \qquad \text{(VII.1)}$$

where:

Lei is the estimated load for land use i given by the model in lb/yr

EMCei is the estimated EMC (for TN and TP) for land use i given by the model in mg/L

453592 is a unit conversion factor from milligrams to pounds.

2) Total runoff volume calculation (Vtot, in L/yr):

$$Vtot = \Sigma \left(Vroi \right) \qquad \text{(VII.2)}$$

3) Average estimated EMC ($AEMC_e$) by the model (mg/L) for the whole site:

$$AEMC_e = \frac{L_e}{Vtot} * 453592 \quad \text{(VII.3)}$$

Where

$AEMC_e$ is the calculated average EMC estimated by the model for the whole watershed, without considering land use specificity (mg/L)

L_e is the total load given by the model in lb/yr

The $AEMC_e$ is determined for the site based on annual precipitation depth. Hence, it should be compared to the monitored site for which an adequate number of events has been collected such that the mean of the EMCm (EMC measured) is a good approximation of the annual mean concentration.

A number of 11 events should be enough provided that they were got for all the four seasons.

The results are presented in table I.2.

Table I.2. Concentration comparisons between the estimations by the Tar-Pamlico model and the average EMCs measured on the eight sites

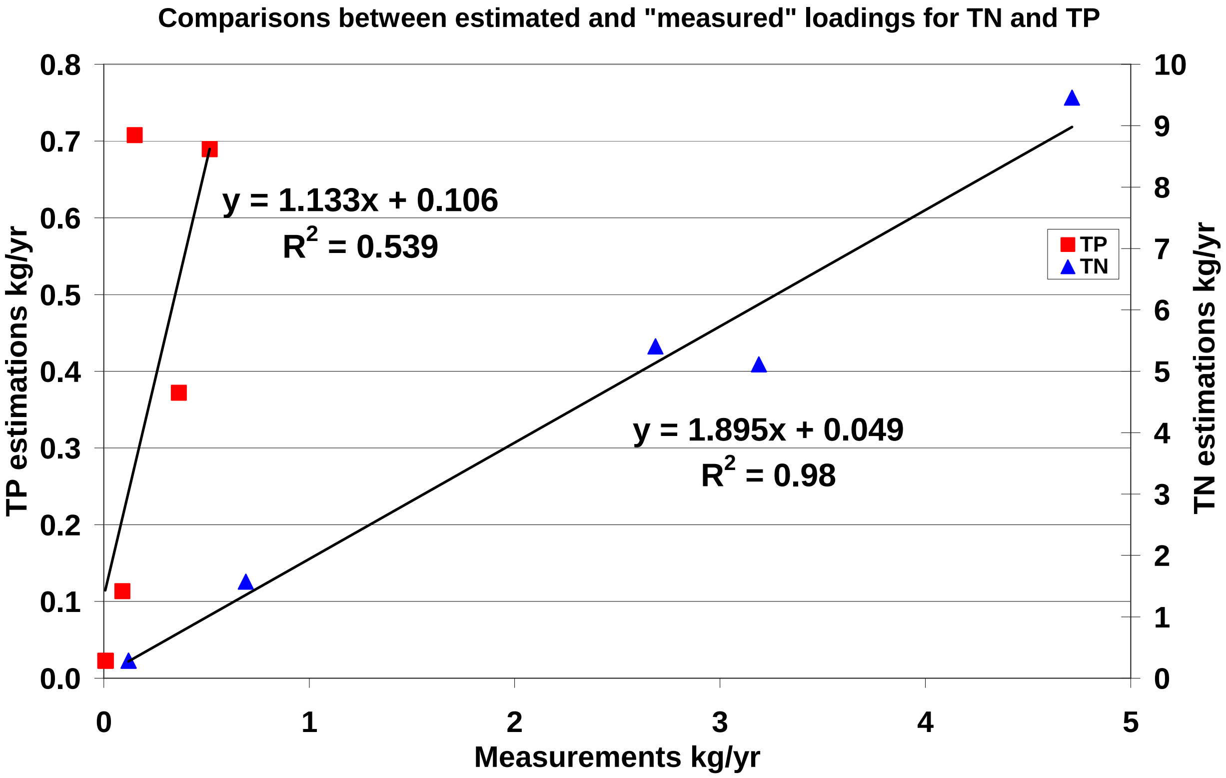
	Concentrations mg/L				
	Measurements		**Model estimations**		
	TNm	**TPm**	**TNe**	**TPe**	**Nb events**
Gra	1.43	0.08	1.86	0.24	25 (TN) and 13 (TP)
C	1.83	0.20	2.60	0.19	27
K1	1.13	0.10	2.60	0.19	11
K2	1.14	0.07	2.60	0.19	11
L1	1.84	0.33	2.52	0.20	12
L2	2.19	0.23	1.46	0.19	12
Go	1.52	0.20	2.60	0.19	14
Gre	1.57	0.18	2.60	0.19	27

Apart from L2 site, model concentrations are systematically higher than measured average EMCs for TN which confirms what was found previously. For TP, average EMCs are more similar and in some cases, measurements were higher than estimations.

I.3.2. Load comparisons

For a subset of the eight sites evaluated in this study, annual load exportations were determined. For that, the total annual runoff depth was needed. However, this was not an available parameter for all sites and was therefore estimated by the NRCS CN (λ=0.05) methodology for each storm event. Rainfall depths were either measured on site or taken from USGS or SCO NC stations. Furthermore, there was not an EMC measurement for every storm event. Therefore, it was assumed that the average of the EMCs monitored from a given site (at least 11 storms over a year) was a good approximation of the "real" annual average EMC. Figure VII.3.2.a presents the results of estimations by the Tar-Pamlico model and "real" loadings determined from measured EMCs for six sites (Gra, C, K1, K2 (points are overlaid), Go, Gre).

Fig. I.3.2.a. Comparisons between estimated (by Tar-Pamlico model) and "measured" loadings for TN and TP.



As suggested in the analysis of asphalt parking lot runoff quality, TN estimations by the Tar-Pamlico model are higher than "real" loadings measured at the research sites. This is less evident for TP where one point is slightly offset (Gra site), thus reducing the R-squared value. Both slopes are greater than one and the TN regression line R-squared value is close to one. However, it should be stressed that “real, measured” loadings come from calculations made using estimated runoff volumes because total annual runoff volumes were not available. In spite of this estimation, when considering EMCs and loads, it is clear that the model over-estimates TN and potentially also over-estimates TP loadings.

Appendix II: NRCS CN methodology

II.1. Method presentation

The National Resources Conservation Service Curve Number (NRCS CN) methodology (SCS 1986; Mishra and Singh 2003) is a conceptual model calculating direct runoff depth from rainfall depths and initial abstraction losses using a one-parameter equation. This method is widely used for hydrological design due to its simplicity, stability, limited input requirements, and dependence on the main factors influencing runoff generation in a given watershed.

The NRCS CN methodology was employed for two aspects in this project. First, it was used to calculate the runoff volumes generated by each land cover in the watersheds, thus enabling an estimate of the proportion of runoff volume that could be attributed to the asphalt portion of the watersheds. Secondly, it was used to substitute missing runoff volume values for the sites and the storm events which were not measured.

The method is based on the water balance equation (1) and two hypotheses (2 and 3).

$$P = Ia + F + Q \qquad (1)$$

$$\frac{Q}{(P - Ia)} = \frac{F}{S} \qquad (2)$$

$$Ia = \lambda * S \qquad (3)$$

Combining equations gives equation (4) :

$$Q = \frac{(P - Ia)^2}{(P - Ia + S)} \qquad (4)$$

Where:

P = rainfall depth (mm)
Ia = initial abstraction depth (mm)
F = infiltration depth (mm)
Q = surface runoff (mm)
S = potential maximum retention or infiltration (mm): it depends on soil type, land use, hydrologic conditions and antecedent dry moisture conditions. It is mapped into a dimensionless curve number

that can be found in tables:

$$S = \frac{1000}{CN} - 10$$

λ = initial abstraction coefficient: regional parameter linked to S and CN.

Despite the wide use of the SCS CN method, it was suggested that both λ and CN were too large in the current tables (Woodward et al. 2003). Thus, some studies provide new guidelines to implement a new CN table (Schneider and McCuen 2005). The use of an initial abstraction coefficient of 0.05 was proposed instead of the $\lambda = 0.2$ that is commonly used (Woodward et al. 2003; Lim et al 2006). Modified CNs have to be taken into account when using 0.05 value.

With $\lambda=0.05$, the runoff equation becomes:

$$Q = \frac{(P - 0.05 * S_{0.05})^2}{(P + 0.95 * S_{0.05})} \qquad \text{for } P>0.05*S_{0.05}$$

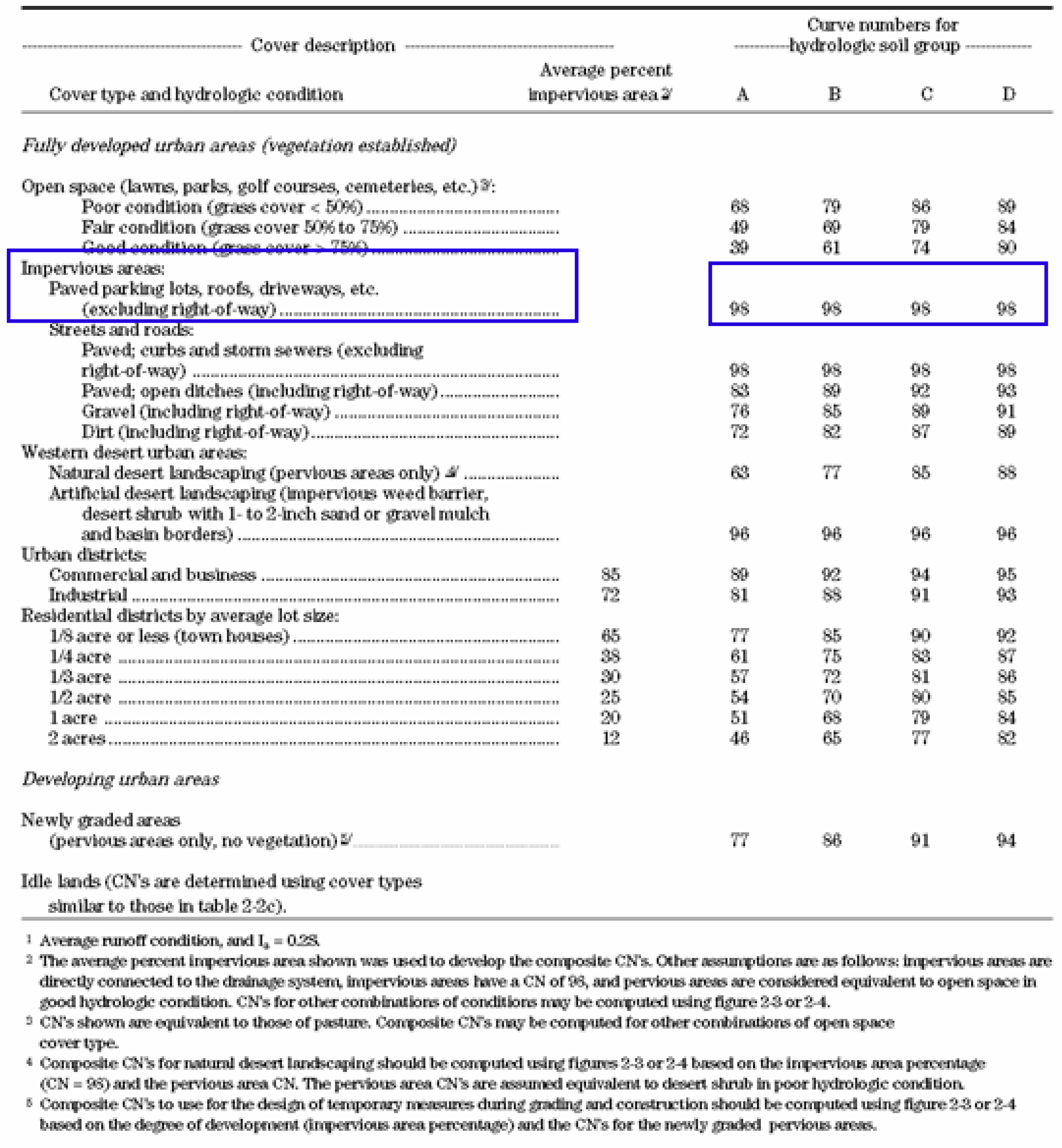
$$Q = 0 \qquad \text{for } P<0.05*S_{0.05}$$

Where:

$$S_{0.05} = 1.33*S_{0.20}^{1.15}$$

$$CN_{0.05} = \frac{100}{1.879*(\frac{100}{CN_{0.20}} - 1)^{1.15} + 1}$$

The following table is a part of a common CN table (from SCS 1986).



Cover description		Curve numbers for hydrologic soil group			
Cover type and hydrologic condition	Average percent impervious area [2]	A	B	C	D
Fully developed urban areas (vegetation established)					
Open space (lawns, parks, golf courses, cemeteries, etc.) [3]:					
Poor condition (grass cover < 50%)		68	79	86	89
Fair condition (grass cover 50% to 75%)		49	69	79	84
Good condition (grass cover > 75%)		39	61	74	80
Impervious areas:					
Paved parking lots, roofs, driveways, etc. (excluding right-of-way)		98	98	98	98
Streets and roads:					
Paved; curbs and storm sewers (excluding right-of-way)		98	98	98	98
Paved; open ditches (including right-of-way)		83	89	92	93
Gravel (including right-of-way)		76	85	89	91
Dirt (including right-of-way)		72	82	87	89
Western desert urban areas:					
Natural desert landscaping (pervious areas only) [4]		63	77	85	88
Artificial desert landscaping (impervious weed barrier, desert shrub with 1- to 2-inch sand or gravel mulch and basin borders)		96	96	96	96
Urban districts:					
Commercial and business	85	89	92	94	95
Industrial	72	81	88	91	93
Residential districts by average lot size:					
1/8 acre or less (town houses)	65	77	85	90	92
1/4 acre	38	61	75	83	87
1/3 acre	30	57	72	81	86
1/2 acre	25	54	70	80	85
1 acre	20	51	68	79	84
2 acres	12	46	65	77	82
Developing urban areas					
Newly graded areas (pervious areas only, no vegetation) [5]		77	86	91	94
Idle lands (CN's are determined using cover types similar to those in table 2-2c).					

[1] Average runoff condition, and $I_a = 0.2S$.

[2] The average percent impervious area shown was used to develop the composite CN's. Other assumptions are as follows: impervious areas are directly connected to the drainage system, impervious areas have a CN of 98, and pervious areas are considered equivalent to open space in good hydrologic condition. CN's for other combinations of conditions may be computed using figure 2-3 or 2-4.

[3] CN's shown are equivalent to those of pasture. Composite CN's may be computed for other combinations of open space cover type.

[4] Composite CN's for natural desert landscaping should be computed using figures 2-3 or 2-4 based on the impervious area percentage (CN = 98) and the pervious area CN. The pervious area CN's are assumed equivalent to desert shrub in poor hydrologic condition.

[5] Composite CN's to use for the design of temporary measures during grading and construction should be computed using figure 2-3 or 2-4 based on the degree of development (impervious area percentage) and the CN's for the newly graded pervious areas.

II.2. Example of applicability: comparisons between estimations and measurements

Runoff volume was measured at Kinston and Charlotte sites. The following graphs present both estimations by the NRCS CN method and the measurements of runoff volumes for site K1 and C.

Fig. II.a: Comparison of volume estimations and measurements for Kinston K1 site

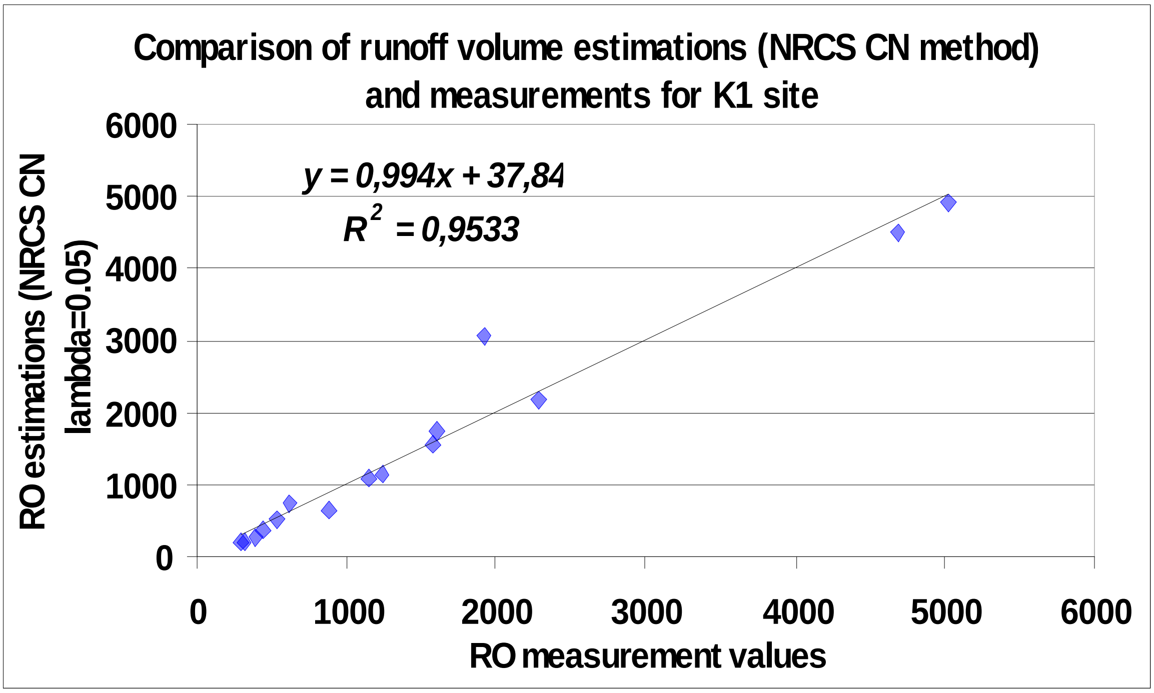
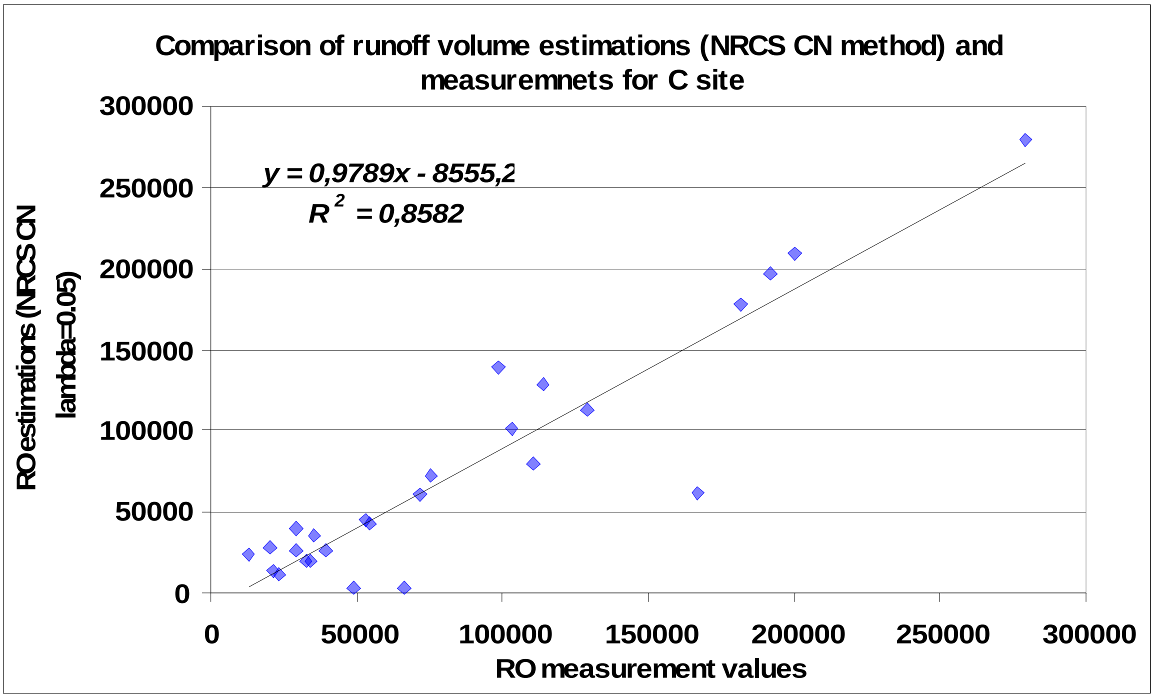


Fig. II.b: Comparison of volume estimations and measurements for Charlotte site



It can be shown from these graphs that the slopes are very close to one and a small bias (compared to total volume values) is present. Moreover, the R-squared value is high, showing that runoff volume estimations calculated by this method are considered to be reasonable approximations of runoff volume measurements.

Appendix III: Photos of typical asphalt parking lot site drainage areas

The following pictures are examples of the drainage areas of some of the eight sites evaluated as part of the study. The red crosses indicate where the asphalt parking lot runoff was measured.

Fig. III.a. Louisburg L2 site

Fig. III.b. Kinston K1 site

Fig. III.c. Kinston K2 site

Fig. III.d. Charlotte site

Fig. III.e. Graham site

Appendix IV: Data transformation

As a result of the surprisingly high values for nutrient species in the asphalt runoff at the Kinston sites, duplicate (and then triplicate) of flow-weighted composite samples of the runoff were sent to a new laboratory (and then a third laboratory).

For the duplicated samples, the values of the concentrations for the first laboratory (Y) are plotted against the values determined for the same sample by the second laboratory (X). These results are not yet published and only the approximate linear equations and associated R-squared values are presented below.

Table IV.1. Comparisons of the first and the second laboratory EMC values.

Constituent	**First order linear regression equation**	**R-squared value**
TP	Y = -0.0098*X + 0.632	0.260
OPO_4-P	Y = -0.0011*X + 0.134	0.025
TN	Y = 0.956*X + 0.240	0.689
TKN	Y = 0.775*X + 0.320	0.491
NH_4-N	Y = 0.826*X + 0.008	0.681
$NO_{2,3}$-N	Y = 1.075*X – 0.061	0.964

These inconsistencies were confirmed by the analysis of 4 samples by a third lab whose results matched those of the second lab. From these results, it clearly appears that phosphorous species (TP and OPO_4-P) and TKN concentrations determined by the first laboratory are probably very different from the real concentration values. Furthermore, the slope of the TN curve was close to one and the R-squared value was not substantially low (0.689). However, a bias of 0.24 was noticed. Therefore, for that particular variable, TN concentrations determined by the first lab were transformed removing a 0.24 bias (for K1 and K2; 0.20 for Graham) and were then included in the analyses.

Three sites were affected by this laboratory issue, Graham, Kinston K1 and Kinston K2 for 13, 12 and 8 storm events, respectively.

Appendix V: Graham High School bioretention cells

V.1. Plan and general characteristics of the Graham H.S. Bioretention cells and associated drainage area

Fig. V.a. Plan of Graham H.S. Site (*by Ryan Smith, Extension Associate*)

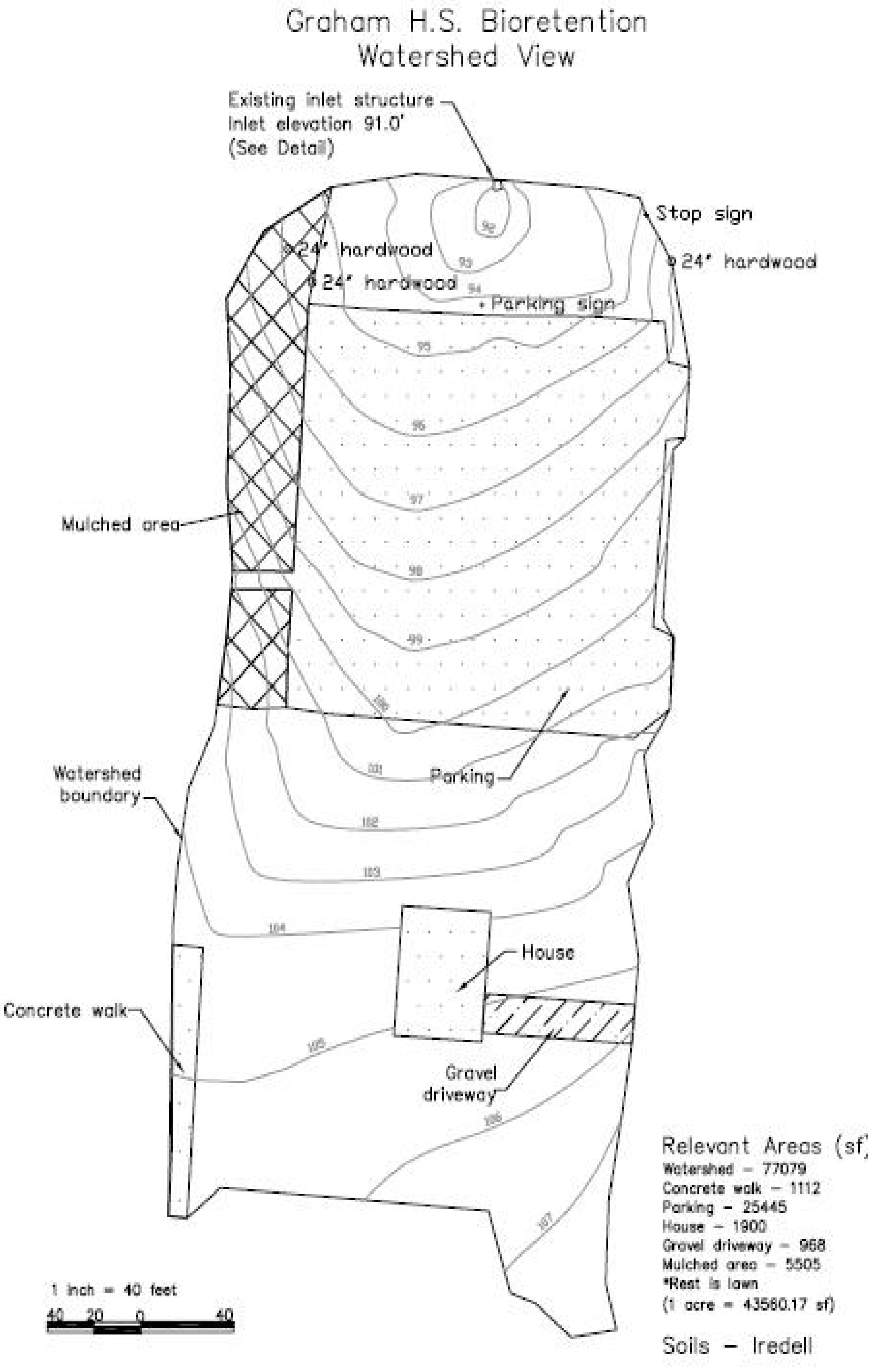
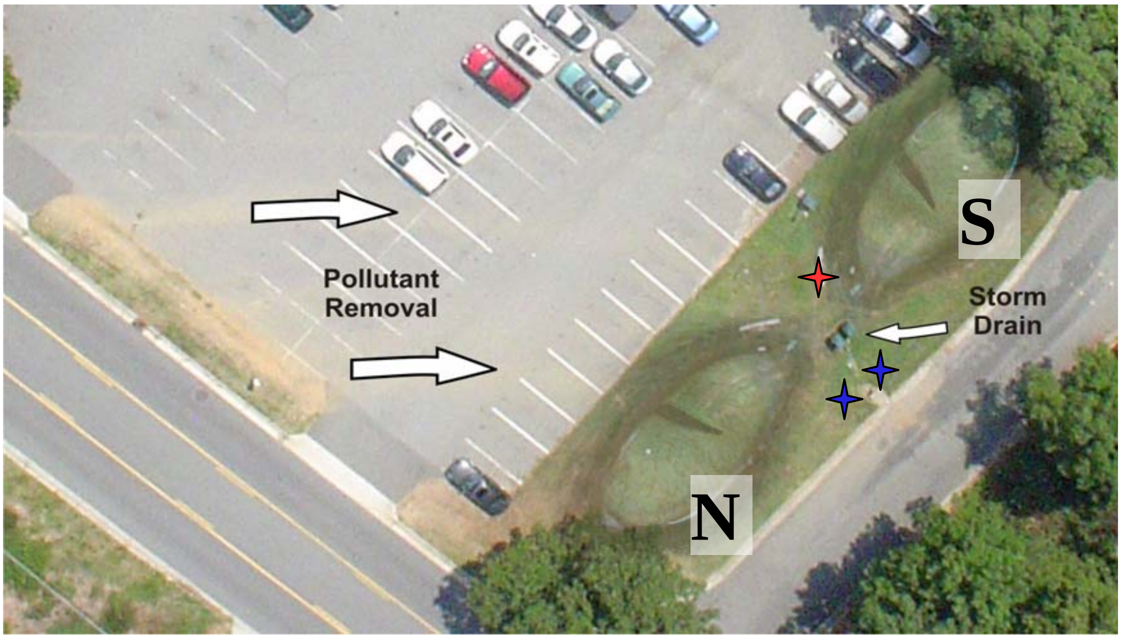


Table V.1. Drainage area components

	Area (m²)
Concrete walk	103
Parking lot	2364
House	177
Gravel driveway	90
Mulched area	511
Lawn	3916
Total Watershed	7161

V.2. Photos of Graham H.S. site

Fig. V.b. Aerial photo of the cells (*photo by Seth Nagy, NC Cooperative Extension*)



The asphalt parking lot portion of the drainage area is shown in this photo. The red cross indicates runoff (inflow volumes) measurement location and the blue crosses show outflow volume measurements for the Northern (N) and the Southern (S) cells.

Fig. V.c. Drainage area

The drainage area land uses mainly consist of this house (on the left), the yard and the asphalt parking lot (a better view is shown in Fig. IV.b.)

Fig. V.d. Inlet structure

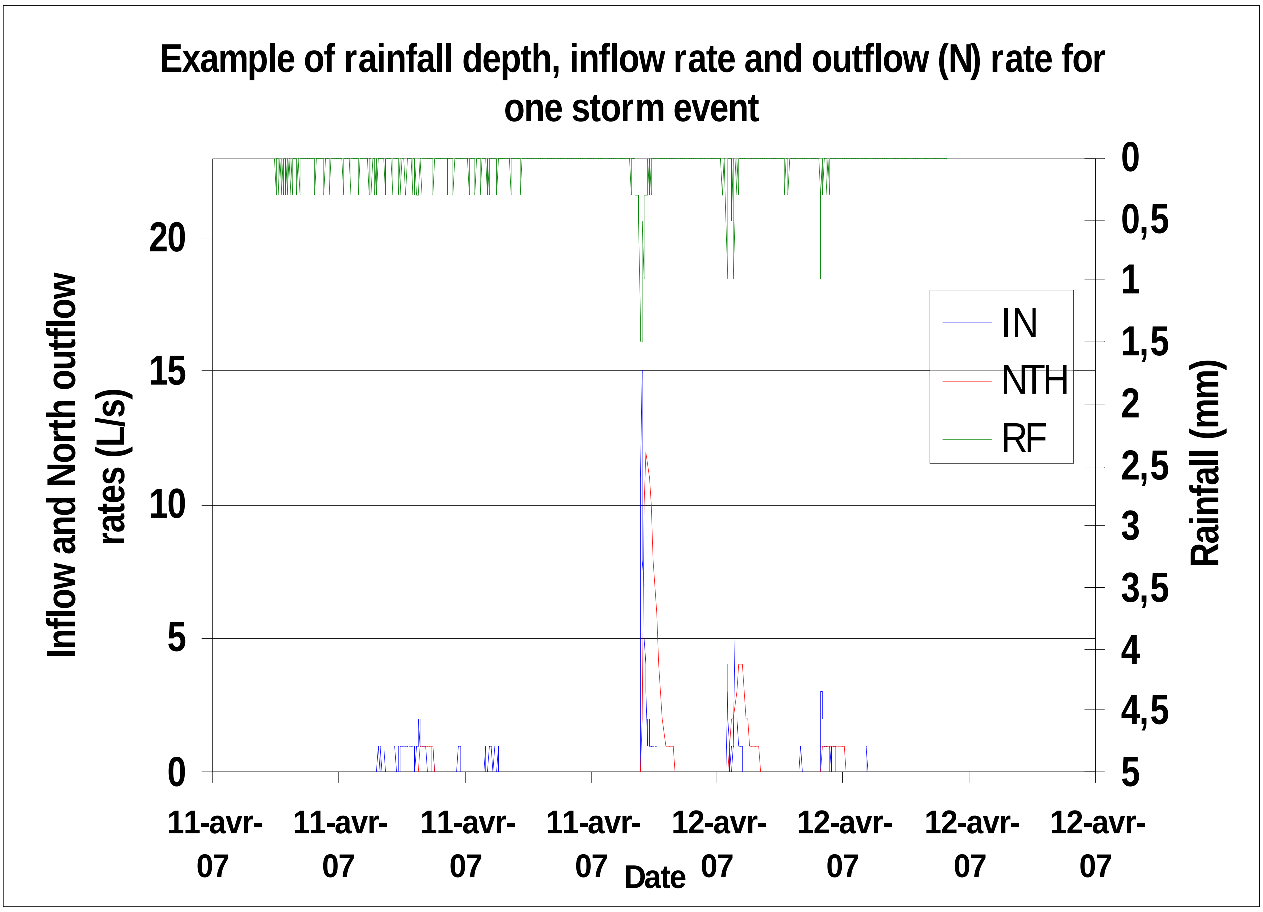
Fig. V.e. Outflow measurement

Fig. V.f. Automatic sampler

Appendix VI: Graham hydrology: typical inflow and outflow hydrographs

On the graph presented below for one particular storm event (April 12, 2007), it appears that the peak flow was reduced and delayed after passing through the North cell. This expected tendency was not found for all the monitored storm events.

Fig. VI.1. Example of rainfall depth, inflow rate and outflow rate (N) for one particular storm event



Appendix VII: All the results for Graham water quality and hydrology data (IN=inlet, N=North cell, S=south cell)

VII.1.All storm event EMCs

		EMCs (mg/L)					
CODE	DATE	TKN	NH$_4$-N	NO$_{2,3}$-N	TN	TP	OPO$_4$-P
IN	03/04/2006		0.24	0.43	0.89		
N	03/04/2006		0.06	0.64	0.79		
S	03/04/2006		0	0.01	0.12		
IN	04/04/2006		0.36	0.41	2.92		
N	04/04/2006		0.13	0.53	1.04		
S	04/04/2006	not enough outflow					
IN	10/04/2006		0.63	0.78	3.22		
N	10/04/2006		0	0.28	0.68		
S	10/04/2006	not enough outflow					
IN	28/04/2006		0.25	0.28	0.94		
N	28/04/2006		0.04	0.27	0.68		
S	28/04/2006		0	0.42	0.91		
IN	08/05/2006		0.21	0.38	1.96		
N	08/05/2006		0.11	0.29	0.6		
S	08/05/2006		0.06	0.2	0.47		
IN	19/05/2006		1.02	0.55	2.93		
N	19/05/2006		0.29	0.29	0.94		
IN	26/06/2006		0.64	0.4	1.91		
N	26/06/2006		0.08	0.21	0.96		
S	26/06/2006		0.05	0.43	0.59		
IN	28/06/2006		0.17	0.04	1.09		
N	28/06/2006		0.04	0.02	0.09		

IN	07/07/2006		0.05	0.38	1.14	
N	07/07/2006		0.07	0.04	0.45	
S	07/07/2006		0	0.28	0.69	
IN	14/07/2006		0.31	0.44	2.45	
N	14/07/2006		0.08	0.05	0.57	
S	14/07/2006		0	0.33	0.64	
IN	24/07/2006		0.52	0.58	2.25	
N	24/07/2006		0.09	0.17	0.69	
S	24/07/2006		0.11	0.34	0.7	
IN	31/08/2006		0.8	0.64	2.34	
N	31/08/2006		0.08	0.23	0.45	
IN	14/09/2006		0.27	0.24	0.75	
N	14/09/2006		0.07	0.02	0.18	
S	14/09/2006		0.02	0.13	0.2	
IN	24/09/2006	1.036	0.1733	0.323	1.359	0.1544
N	24/09/2006	No sample				
S	24/09/2006	No outflow				
IN	28/09/2006	1.459	0.147	.		0.215
N	28/09/2006	No sample				
S	28/09/2006	No outflow				
IN	08/10/2006	0.462	0.1925	0.3	0.762	0.0636
N	08/10/2006	0.3048	0.0605	0.17127	0.47605	0.0469
S	08/10/2006	No outflow				
IN	17/10/2006	0.279	0.0697	0.102	0.381	0.0707
N	17/10/2006	0.2137	0.0362	0.11806	0.33171	0.0395
S	17/10/2006	0.387	0.089	0.602	0.989	0.055
IN	28/10/2006	0.293	0.0195	0.288	0.581	0.0429
N	28/10/2006	No sample				
S	28/10/2006	0.532	0.1	0.484	1.016	0.094

IN	07/11/2006	0.523	0.0647	0.431	0.954	0.0777	
N	07/11/2006						
S	07/11/2006	0.461	0.024	0.523	0.984	0.058	
IN	13/11/2006	0.825	0.0302	0.243	1.068	0.0819	
N	13/11/2006				0		
S	13/11/2006	0.497	0.037	0.273	0.77	0.061	
					0		
IN	16/11/2006	0.689	0.0658	0.19	0.879	0.0961	
N	16/11/2006						
S	16/11/2006	0.405	0.047	0.204	0.609	0.077	
IN	26/02/2007	0.443	0.153	0.267	0.71	0.039	0.012
N	26/02/2007	0.382	0.071	0.376	0.758	0.111	0.012
S	26/02/2007	0.364	0.07	0.421	0.785	0.044	0.015
IN	03/02/2007	0.699	0.111	0.158	0.857	0.059	0.009
N	03/02/2007	0.406	0.065	0.221	0.627	0.049	0.012
S	03/02/2007	0.442	0.06	0.383	0.825	0.056	0.015
IN	03/19/2007	0.526	0.122	0.352	0.878	0.046	0.009
N	03/19/2007	0.46	0.074	0.385	0.845	0.039	0.011
S	03/19/2007	0.467	0.1	0.524	0.991	0.045	.
IN	04/13/2007	0.722	0.179	0.473	1.195	0.066	0.005
N	04/13/2007	0.423	0.081	0.404	0.827	0.051	0.016
S	04/13/2007	0.513	0.162	0.578	1.091	0.041	0.015
IN	04/17/2007	0.921	0.172	0.393	1.314	0.073	0.016
N	04/17/2007	0.385	0.045	0.315	0.7	0.05	0.018
S	04/17/2007	0.456	0.054	0.787	1.243	0.047	0.013

VII.2. All storm event inflow and outflow volumes. rainfall depths and loads

		Volumes	RF	Loads (mg/m²)					
CODE	DATE	L	mm	TKN	NH_4-N	$NO_{2.3}$-N	TN	TP	OPO_4-P
IN	03/04/2006	4930	5.334		0.166	0.298	0.616		
N	03/04/2006	4548			0.038	0.409	0.504		
S	03/04/2006	85			0.000	0.000	0.001		
IN	04/04/2006	4033	4.826		0.204	0.232	1.654		
N	04/04/2006				0.000	0.000	0.000		
S	04/04/2006	31							
IN	10/04/2006	5872	5.842		0.519	0.643	2.655		
N	10/04/2006	5379			0.000	0.211	0.514		
S	10/04/2006	60							
IN	28/04/2006	40746	20.574		1.430	1.602	5.378		
N	28/04/2006	83936			0.471	3.182	8.014		
S	28/04/2006	110851			0.000	6.537	14.164		
IN	08/05/2006	39401	20.066		1.162	2.102	10.843		
N	08/05/2006	9174			0.142	0.374	0.773		
S	08/05/2006	1934			0.016	0.054	0.128		
IN	19/05/2006	2788	4.064		0.399	0.215	1.147		
N	19/05/2006	1599			0.065	0.065	0.211		
IN	26/06/2006	125476	44.196		11.276	7.047	33.651		
N	26/06/2006	76715			0.862	2.262	10.341		
S	26/06/2006	98108			0.689	5.923	8.127		
IN	28/06/2006	10520	8.128		0.251	0.059	1.610		
N	28/06/2006	10166			0.057	0.029	0.128		
IN	07/07/2006	141078	47.752		0.990	7.527	22.582		
N	07/07/2006	78221			0.769	0.439	4.942		
S	07/07/2006	326778			0.000	12.847	31.659		

IN	14/07/2006	48587	23.368		2.115	3.002	16.714	
N	14/07/2006	33823			0.380	0.237	2.707	
S	14/07/2006	34946			0.000	1.619	3.140	
IN	24/07/2006	53166	24.892		3.882	4.330	16.796	
N	24/07/2006	25517			0.322	0.609	2.472	
S	24/07/2006	12910			0.199	0.616	1.269	
IN	31/08/2006	101193	38.354		11.367	9.093	33.248	
N	31/08/2006	not recorded						
IN	14/09/2006	94170	36.576		3.570	3.173	9.917	
N	14/09/2006	not recorded						
S	14/09/2006	not recorded						
IN	24/09/2006	16656	9.652	2.423	0.405	0.755	3.178	0.361
N	24/09/2006	5103						
S	24/09/2006	0						
IN	28/09/2006	5762	4.572	1.180	0.119			0.174
N	28/09/2006	0						
S	28/09/2006	0						
IN	08/10/2006	27634	18.034	1.793	0.747	1.164	2.957	0.247
N	08/10/2006	46182		1.976	0.392	1.111	3.087	0.304
S	08/10/2006	0						
IN	17/10/2006	57826	23.622	2.265	0.566	0.828	3.093	0.574
N	17/10/2006	62153		1.865	0.316	1.030	2.895	0.345
S	17/10/2006	5030		0.273	0.063	0.425	0.698	0.039
IN	28/10/2006	71378	22.606	2.937	0.195	2.886	5.823	0.430
N	28/10/2006	55051						
S	28/10/2006	21345		1.594	0.300	1.451	3.045	0.282
IN	07/11/2006	111836	33.02	8.213	1.016	6.768	14.981	1.220
N	07/11/2006	not recorded						
S	07/11/2006	30060		1.946	0.101	2.207	4.153	0.245

IN	13/11/2006	108816	22.86	12.605	0.461	3.713	16.318	1.251	
N	13/11/2006	not recorded							
S	13/11/2006	40530		2.828	0.211	1.554	4.382	0.347	
IN	16/11/2006	210000	29.718	20.316	1.940	5.602	25.918	2.834	
N	16/11/2006	not recorded	0						
S	16/11/2006	124569	0	7.084	0.822	3.568	10.652	1.347	
			0						
IN	26/02/2007	69950	22.86	4.351	1.503	2.622	6.973	0.383	0.118
N	26/02/2007	41270	0	2.214	0.411	2.179	4.392	0.643	0.070
S	26/02/2007	19514	0	0.997	0.192	1.154	2.151	0.121	0.041
			0						
IN	03/02/2007	91336	20.828	8.964	1.424	2.026	10.991	0.757	0.115
N	03/02/2007	61782	0	3.522	0.564	1.917	5.439	0.425	0.104
S	03/02/2007	57175	0	3.548	0.482	3.075	6.623	0.450	0.120
			0						
IN	03/19/2007	104764	34.798	7.737	1.795	5.178	12.915	0.677	0.132
N	03/19/2007	73725	0	4.762	0.766	3.985	8.747	0.404	0.114
S	03/19/2007	34584	0	2.268	0.486	2.545	4.812	0.219	0.078
			0						
			0						
IN	04/13/2007	55222	28.956	5.598	1.388	3.668	9.266	0.512	0.039
N	04/13/2007	44456	0	2.640	0.506	2.522	5.162	0.318	0.100
S	04/13/2007	9919	0	0.714	0.226	0.805	1.519	0.057	0.021
			0						
IN	04/17/2007	238326	60.198	30.820	5.756	13.151	43.971	2.443	0.535
N	04/17/2007	360757	0	19.502	2.279	15.956	35.458	2.533	0.912
S	04/17/2007	357705	0	22.903	2.712	39.527	62.430	2.361	0.653