ABSTRACT

MCNETT, JACQUELYN MCNETT. Proposing a New Method of Stormwater BMP Assessment and Evaluating the Toxicity of Forebay Sediments. (Under the direction of Dr. William F. Hunt).

Stormwater experts agree that the currently used Best Management Practice (BMP) percent removal methodology metric has many flaws, because it does not account for background water quality, eco-region differentiation, or irreducible concentrations. Some have suggested utilizing a BMP effluent concentration metric. Chapter 1 establishes an effluent target concentration for BMPs that relates to the health of macro-invertebrates in receiving water. 193 ambient water quality monitoring stations in North Carolina were paired with benthic macro-invertebrate health ratings collected in very close proximity. Water quality for the sites ranged from Excellent to Poor and was divided into three distinct eco-regions: Mountain, Piedmont, and Coastal. Statistically significant relationships were found in one or more eco-regions for DO, Fecal Coliform, NH3, NO_{2-3} - N, TKN, TN, and TP. BMPs can then be selected and designed to meet these target effluent concentrations. Based upon this research, a development, and therefore set of BMPs, in Piedmont North Carolina could be required to release TN and TP effluent concentrations of 0.99 mg/L and 0.11 mg/L, respectively. These concentrations are both associated with "Good" Benthos health. The new method was most effective in the Piedmont eco-region, however with more data collection, the Mountain and Coastal eco-regions may also benefit.

The removal efficiency metric inherently assumes a definite association between influent and effluent pollutant concentrations. Such a relationship has been minimally studied for bioretention, the most common stormwater control measure associated with Low Impact Development (LID). Chapter 2 analyzes influent and effluent TN and TP concentrations from

11 bioretention cells in the Mid-Atlantic United States. Pooled data showed only a slight association between influent and effluent TN. Essentially no relationship exists between influent and effluent TP concentrations. Both findings indicate that the percent removal metric is probably a faulty means of evaluating bioretention performance. Furthermore, as influent nutrient concentration in runoff increases, the removal efficiency increases for TN and TP. "Dirtier" influent TP concentrations were effectively reduced; conversely, "cleaner" TP influent concentrations increased, both tending toward an irreducible effluent concentration (0.10 to 0.18 mg/l). TN data also may have been tending toward a common concentration; however, the value was not as discernible.

After developing a new set of standards for water quality (Chapter 1) and verifying the need for a new method of BMP assessment (Chapter 2), the feasibility of these innovative standards was examined (Chapter 3). Which BMPs, if any, can meet effluent standards set by the WQABI metric established in Chapter 1? A compilation of Mid-Atlantic effluent total nitrogen (TN) and effluent total phosphorus (TP) concentrations were compiled from existing, published studies conducted on the nine most commonly installed BMPs, including: bioretention cells, dry detention, green roofs, level spreaders-vegetated filter strips (LS-VFS), permeable pavement, sand-filter, vegetated swales, wetlands, and wet ponds. Of all BMPs examined in this study, none met "E" WQABI standards for TN ($\leq$ 0.69 mg/L), 4 BMPs (bioretention, permeable pavement, wetlands and wet ponds) met "G" WQABI standards for TN ($\leq$ 0.99 mg/L), and 6 BMPs (bioretention, dry detention basins, LS-VFS, permeable pavement, wetlands and wet ponds) met "GF" WQABI standards for TN ($\leq$ 1.17 mg/L). Only green roofs and grassed swales were unable to achieve "F" TN water quality standards ($\leq$ 2.16 mg/L). Of all BMPs selected in this study, only permeable pavement met

“E” WQABI standards for TP ($\leq$ 0.06 mg/L) and 4 BMPs (bioretention, permeable pavement, wetlands and wet ponds) met “G” and “GF” WQABI standards for TP ($\leq$ 0.11 mg/L and $\leq$ 0.13 mg/L, respectively; Table 5).

A final, distinct study (Chapter 4) evaluated the toxicity of forebay sediments and suggested an appropriate method to dispose of excavated sediment. Forebays, small settling basins placed at the inlet of Stormwater Best Management Practices (BMPs), encourage sedimentation with the intention of pollutant consolidation and capture. To test for the potential toxicity of forebay spoils, 30 stormwater wetland and wetpond forebays, of varying age, size, and upstream landuse were sampled across North Carolina and analyzed for 7 metals: cadmium, chromium, copper, iron, lead, nickel, and zinc. The relative toxicity of all sampled sediment metal concentrations was evaluated using existing aquatic health sediment guidelines and U.S. Environmental Protection Agency (EPA) standards for the land application of biosolids (40 CFR503). Twenty-two of 30 forebays exceeded sediment guidelines for aquatic health with respect to one or more metals, emphasizing the need for routine forebay sediment removal. All samples were less than 40 CFR 503 limits with factors of safety ranging from 5 to 13, indicating that land application of forebay sediment is an acceptable means of disposal.

Proposing a New Method of Stormwater BMP Assessment and Evaluating the Toxicity of Forebay Sediments

by
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DEDICATION

To my Parents who Gave me Life and Unconditional Love.

Everybody needs beauty as well as bread, places to play in and pray in, where nature may heal and give strength to body and soul.

- *John Muir*

BIOGRAPHY

Jacquelyn K. McNett was born in Dearborn, Michigan on July 31, 1986 to Gerald and Kimberly McNett. She was named after her loving grandfather, Jack Penrod, who died much too young, at the age of 54, in 1988. Gerald and Kimberly, now married for 31 years, also have a son, William, who was born 4 years prior to Jacquelyn.

When Jackie was 1 year old, her parents moved the family to Commerce Twp., MI where they still reside. Jackie spent her childhood outside bike-riding and playing with other kids on Weston Ct., a one street neighborhood. She also spent a lot of time creating memories with her family on family camping trips to the Upper Peninsula and Northern Michigan, as well as Wyoming, and parts of Canada. Jackie learned to fish, swim (aka “dog paddle”), and water-ski on the many lakes of Michigan. She gained a deep love of nature through the Great Lakes, seashores, dunes, and virgin forests of Michigan.

Jackie graduated from Walled Lake Central High School in May 2004 and started an undergraduate degree program in pre-Medicine at the Lymann Briggs School of Science at Michigan State University. Jackie then decided that Medical school was not for her, and she found a different way to help others while also helping the environment that she loves so much. Jackie graduated from Michigan State University in May 2008 with a degree in Biosystems Engineering and with a mindset to change the world! After receiving the “offer of a lifetime” from Bill Hunt at North Carolina State University, Jackie joined the Stormwater Engineering Team and began working toward a Master’s degree in Biological and Agricultural Engineering, with a minor in Horticulture. Upon graduation in May, 2010, Jackie plans to work for a company committed to environmental restoration and preservation. She wants to educate the public on sustainability and develop artistic, yet feasible, design solutions for environmental problems. Simultaneously, Jackie wants to invest in land and create a garden/small-scale farm, using ideals of Permaculture and travel abroad in her free-time.

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CHAPTER 1. ESTABLISHING STORMWATER BMP EVALUATION METRICS BASED UPON AMBIENT WATER QUALITY ASSOCIATED WITH BENTHIC MACRO-INVERTEBRATE POPULATIONS

Introduction

The USEPA requires that stormwater pollution is controlled to the maximum extent practicable (MEP) through the use of best management practices (BMPs) (USEPA, 2004).When adopted by state and local governments, BMPs are often used to comply with maximum pollutant loading requirements, such as Total Maximum Daily Loads (TMDLs). For example, the North Carolina Department of Environment and Natural Resources (NCDENR) requires a post-development load not to exceed 0.45 kg/ha/yr (0.4 lb/ac/yr) and 4.5 kg/ha/yr (4.0 lb/ac/yr) for total phosphorus (TP) and total nitrogen (TN), respectively, in the Tar-Pamlico River Basin (NCDENR, 2007). State and local governments further provide guidance on BMP designs, including the assignment of load reductions, efficiency ratios, and/or percent removal rates for pollutants. An efficiency ratio (ER) is calculated by Equation 1.1, using either pollutant concentrations or loads as a basis:

$$\boldsymbol{ER = \frac{inflow\ conc. - outflowe\ conc}{inflow\ conc.}} \qquad \textbf{\textit{Eqn (1.1.)}}$$

A percent removal rate is simply the ER multiplied by 100%. In the Tar-Pamlico river basin example, the percent removal rates for a stormwater wetland are 45% and 40% for TP and TN, respectively (Table 1.1). The load reduction and percent removal methods do not typically account for variations in BMP designs, eco-region, or background contaminant levels in receiving waters. As summarized by the Center for Watershed Protection (2007), “Stormwater management criteria commonly assign the median removal efficiency, but this often masks the role of certain design factors in reducing or enhancing performance.” Studies show that BMP efficiency is heavily influenced by changes in design, including media composition and depth, ratio of watershed area to BMP surface area, and/or ponding and retention times (Dietz and Clausen, 2005; Dietz and Clausen, 2006; Hsieh and Davis, 2005; Li et al, 2009; Strecker et al., 2001; Clary et al., 2002; Hunt et al, 2006; Urbonas, 1995; Scholes et al., 2008). Similarly, local hydrology, climate and system age impact BMP performance (Scholes et al., 2008; Strecker et al., 2001; Urbonas, 1995; Line et al., 2008), creating variations in reported removal rates (Table 1.1).

Table 1.1. Assigned total nitrogen (TN) and total phosphorus (TP) BMP removal efficiencies by state

	BMP type							
	Bioretention		Wetpond		Wetland		Sand Filter	
State	TN	TP	TN	TP	TN	TP	TN	TP
New Jersey [1]	30%	60%	30%	50%	30%	50%	35%	50%
North Carolina [2]	35%	45%	25%	40%	40%	40%	35%	45%
Virginia [3]	46%	5%	31%	52%	24%	48%	32%	59%*

*Sand Filter was not specifically listed in the VA Stormwater Handbook. "Other Infiltration Practices" was used in lieu.

[1] NJDEP (2004.)

[2] NCDENR (2007.)

[3] VDCR (1999.)

Percent removal rates and load reductions can be misleading indicators of performance. Jones et al. (2008) neatly present several flaws with the percent removal metric, some of which are elaborated upon herein. The percent removal rate and load reduction metrics do not necessarily coincide with water quality. For example, Lenhart and Hunt

(2010) show that influent TN concentrations at a wetland in River Bend, North Carolina, appear to approach irreducible levels. The wetland demonstrated poor performance when evaluated through the percent removal metric, despite influent and effluent nutrient concentrations being lower than BMP effluent concentrations measured in other nearby studies (Line et al., 2008). Conversely, a wastewater wetland examined by Schaafsma et al. (1999) reduced TN, TP, and total suspended solids (TSS) influent loads by more than 90%, yet effluent concentrations remained above regulatory levels.

A BMP may produce higher effluent concentrations due to surrounding environmental conditions. For example, a bioretention cell receiving runoff from a well-fertilized landscape tends to have higher inflow nutrient concentrations than a cell receiving runoff from a lightly used parking lot (Passeport and Hunt, 2009; Sharkey, 2006; Cook et al., 1996). Runoff from greyfield developments, brownfield developments, and reclaimed lands are expected to have higher pollutant loads than runoff from greenfield developments (Stehouwer et al., 2006). Background and inflow concentrations impact nutrient removal performance and should be considered when setting regulatory standards.

The ultimate goal of stormwater management is to protect water quality and promote healthy aquatic ecosystems. The percent removal and load reduction methods do not take into account aquatic health when assessing BMPs and do not ensure high quality, healthy waters. The following study introduces a novel methodology of assessing aquatic health using ambient water quality, in order to select BMPs that are able to meet the desired level of ambient water quality.

POSSIBLE SOLUTION: ASSESSING WATER QUALITY USING BENTHIC MACROINVERTEBRATES. Biotic indicator species, such as benthic macro-invertebrates (benthos), periphyton, and certain fish populations, are used to assess water quality of streams and rivers across the world (Skoulikidis et al., 2004; Gresens et al., 2006, Armitage, 1982; Van Duinen, 2006; Gray, 2008). These organisms are used to predict the degree of impairment in streams and can be an evaluation metric to determine the effectiveness of BMPs (Barbour et al., 1999). Forty-three states use biotic protocols in water resource management (NCDENR, 2008) with benthos being the most commonly used indicator species by state agencies in the United States (Southerland and Stribling, 1995). Benthos reside in rivers, streams, and estuaries in the following 5 habitats: cobble (hard substrates common in the Mountain and Piedmont regions), snags (woody debris including sticks and branches), vegetated banks (submerged lower banks with the presence of plants and roots), submerged macrophytes (seasonal aquatic plants rooted to the stream bottom), and sand/fine sediment (Barbour et al., 1999). A different collection technique is recommended for each habitat (Barbour et al., 1999). The species of benthos used for water quality monitoring correspond to the native species of the particular ecoregion of interest. For example, common freshwater benthos include mussels (class *Bivalvia*), crayfish (order *Decpoda*), worms (order *Oligochaeta*), and stoneflies (order *Plecoptera*). Such freshwater benthos are applicable in the Mountain and Piedmont eco-regions where freshwater is predominant. Benthos range in tolerance and therefore may be predicative of varying water qualities (Table 1.2).

After collection, the benthos are preserved and identified by family and genus/species type. Depending on the abundance and diversity of benthos collected, a stream health rating is assigned, including: Excellent, (E); Good, (G); Good-Fair, (GF); Fair, (F); and Poor, (P) (Barbour et al., 1999).

Table 1.2. Various tolerances of Benthos in terms of Water Quality (Barbour et al., 1999)

Rating	Description of Benthos	Sample Organisms by Scientific Name
Excellent	Very sensitive	*Ephemera Guttulata* (mayfly), *Litobrancha recurvata* (mayfly)
Good	Sensitive	*Drunella allegheniensis* (mayfly), *Rhyacophila fuscula* (caddisfly)
Good-Fair	Semi-tolerant	*Amnicola* (snail), *Elliptio complanata* (mussel)
Fair	Tolerant	*Cambarus* (crayfish), *Crangonyx* (crustacean)
Poor	Very tolerant	*Enchytraeidae* (worm), *Limnodrilus cervix* (worm)

Similar ratings and modified collection and assessment techniques are used in other regions of the world (Buffagni et al., 2004; Simboura and Reizopoulou, 2008). Benthic macro invertebrate health ratings (BMR) are commonly used in European countries to assess water quality and form the basis for the **A**ssessment System for the Ecological **Q**uality of Streams and Rivers throughout **E**urope using Benthic **M**acroinvertebrates (AQEM) project. AQEM uses BMRs to assign a stream health rating from 5 (best) to 1 (worst). Furthermore, AQEM aspires for all rivers and streams in 8 countries in Europe to reach at least a “4” quality rating by 2015 (Skoulikidis et al., 2004; Carsten von der Ohe et al., 2007).

Trends toward regional specificity have emerged with regards to BMR collection techniques and analysis. Using BMRs, Skoulikidis et al. (2004) examined the difference in water quality among three eco-regions in Greece with varying geographical and climatic features. A nutrient metric, reflecting ambient water quality (AWQ), exhibited satisfactory correlation with the biotic metric (BMR). Another study expands on the collection techniques of Barbour et al. (1999) by assessing benthos from 53 streams in Illinois. The study determined that high nutrient levels in streams corresponded to degraded aquatic habitat (Heatherly et al., 2007). In contrast to Skoulikidis et al. (2004), Heatherly et al. (2007), caution against using an eco-regional approach to stream classification. Heatherly et al. (2007) suggest that a certain ecoregion may not exhibit the same benthos throughout. However, like Skoulikidis et al. (2004), the study shows an overall correspondence between elevated nutrient levels and degraded habitats of benthos (Heatherly and Whiles, 2007). Efforts have been made to localize collection techniques. The Mid Atlantic Coastal Streams (MACS) workgroup (including North Carolina) and the Florida Department of Environmental Protection (FLDEP) have developed a scientifically validated sampling procedure for low-gradient coastal streams (Barbour et al., 1999). Numerous states have accepted the eco-region approach in benthos assessment (e.g. Iowa (IADNR, 2008), North Carolina (NCDENR, 2008), and Virginia (VDEQ, 2008)).

Other pollutants such as metals and pesticides are known to be detrimental to benthic health; however, recent studies indicate that nutrients may also be cause for concern. Wang and Dei (2001), for example, show that phytoplankton species in the coastal waters of China exhibited increased cellular metal uptake with elevated nitrogen concentrations. Furthermore, innovative nutrient biotic indices have been developed by pairing BMRs with TP and NO_{3-} concentrations and associated changes in trophic levels that prominently impact water quality (Smith et al., 2007).

Previous North Carolina studies have not directly compared AWQ nutrient concentrations to BMRs, but plentiful data collected by NCDENR provide potential to do so. The AWQ at 339 stream and river sites throughout North Carolina has been monitored on a monthly basis for the past 30 years (NCDWQ, 2008). AWQ data include concentrations of dissolved oxygen (DO), total suspended solids (TSS), nitrogen-ammonia (NH_3), Nitrate Nitrite ($NO_{2\text{-}3}$-N), Total Kjeldahl Nitrogen, (TKN), TP, and Fecal Coliforms. Data were collected in summer months only, from June to September. AWQ data are fairly conservative, as pollutants like DO may reach extreme lows during the hottest portions of summer. Simultaneously, the water quality of approximately 200 North Carolina streams has been rated using benthos or fish (NCDWQ, 2008).

PROPOSING THE WQABI METHOD. BMP performance (effluent pollutant concentrations) can be related to ambient water quality (AWQ) of receiving waters. The AWQ concentrations can further be related to benthos ratings (BMR). In this way, BMP effluent concentrations are linked to the aquatic health of receiving waters, which provides an innovative method for BMP performance evaluation. This study evaluates a method of assessing BMP performance using **W**ater **Q**uality **A**ssessed by **B**enthic macro **I**nvertebrate health ratings (WQABI), specific to one of three eco-regions, similar to Skoulikidis (2004). This study is different from others because all data are specific to North Carolina, including: BMRs, AWQ data, BMP efficiencies and BMP effluent concentrations. Moreover, the methodology proposed herein appears to be the first to link BMP efficiencies to BMRs.

Several application issues of the WQABI method are discussed herein. Are certain pollutants (e.g., TN and TP) more likely to negatively (or in the case of DO, positively) correlate to benthos health (BMR)? In North Carolina, do the Mountain, Piedmont, and Coastal eco-regions require different regulatory standards? It is possible that the WQABI method may be better applied in one eco-region versus another. Do greyfield developments, more likely located in degraded watersheds, need to meet the same standards as green field developments? These questions are important when determining criteria for development and whether or not a developed land can meet pre-developed, or target, water quality conditions. While these particular findings are regionally specific, the methodology presented has wide application.

Methods

AWQ, benthos, and fish sampling locations were obtained from the North Carolina Division of Water Quality Basin-wide Assessment Reports and imported into ArcGIS, version 9.2. (NCDENR, 2008; NCDWQ, 2008; NC OneMap, 2008) (Appendix A). Each BMR or fish sampling location was located in one of three eco-regions: Mountain, Piedmont, or Coastal (Figure 1.1). AWQ data were reported for DO, TSS, NH_3-N, $NO_{2\text{-}3}$-N, TKN, TP, and Fecal coliform. TN was included in the analysis by summing TKN and $NO_{2\text{-}3}$-N concentrations. The Mountain eco-region is the smallest of the three and is separated from the Piedmont region by the Blue Ridge Mountain range. The Piedmont eco-region is located between the Mountain region and the Coastal region. It is characterized by rolling topography with a wide variety of streams and higher levels of urbanization. Lastly, the Coastal region is located in the eastern one-third of North Carolina and primarily consists of relatively flat agricultural land, coastal wetlands, sand hills, and swamps (NCDENR, 2008). The species of benthos across one type of eco-region are fairly consistent.

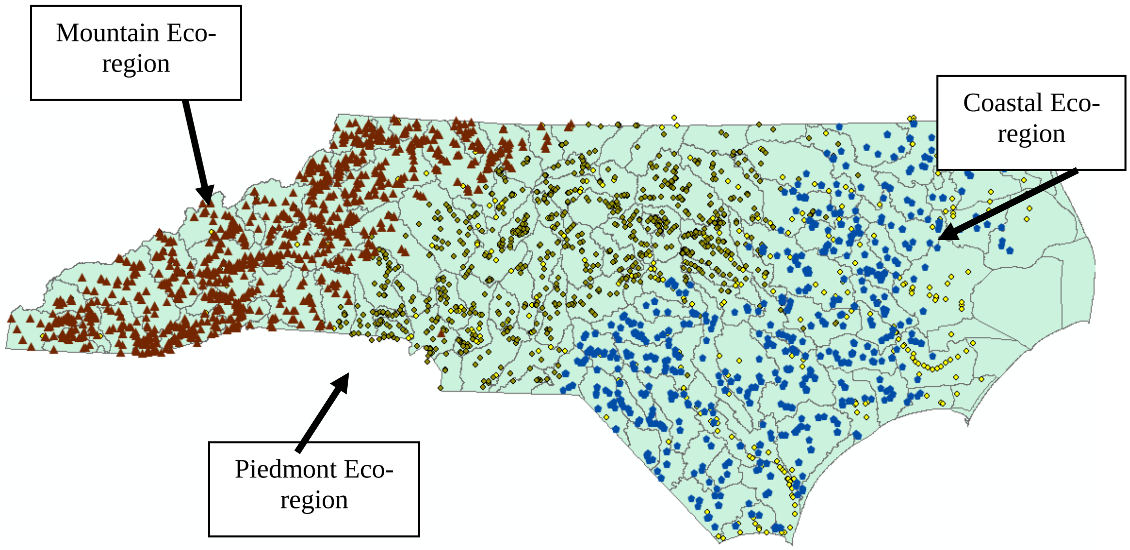

Figure 1.1. Benthic Monitoring sites in the mountain (triangles), piedmont (squares) and coastal (stars) eco-regions in North Carolina, as reported by the Division of Water Quality (NCDENR, 2008).

AWQ monitoring sites were paired with BMR and fish collection sites within a 402 m (¼-mile) range. All paired monitoring locations (AWQ with BMR, or AWQ with fish monitoring sites) were located in the same river or estuary (Appendix A). If the monitoring periods of AWQ and BMR/fish population data collection were not within two years of each other, the pair was omitted from the study due to possible temporal changes in water quality. If multiple benthic ratings were given within the time range of AWQ collection, the lower rating was used. If both fish community and BMRs were available, the BMR was used, as the vast majority of water quality ratings from NCDENR are based on BMR data. Each data pair received one of 5 ratings and was located in one of three eco-regions, yielding a total of 15 possible combinations (Figure 1.2.).

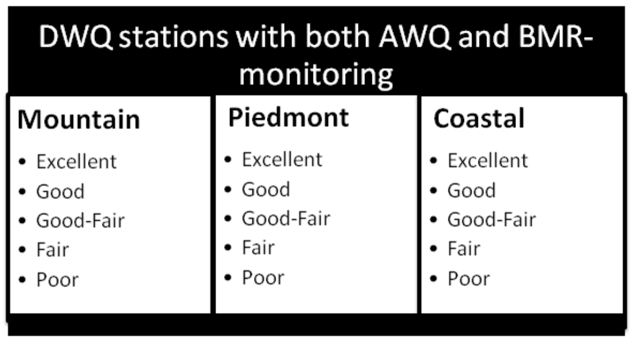

Figure 1.2. Possible combinations of water quality classifications

Results and Discussion

There were 193 DWQ stations in which both AWQ data were collected and BMRs were assigned. If a data pair contained a station where collected AWQ data were limited (a sample size of 10 or less), the pair was excluded. Also, if the number of pairs in a given eco-region was less than 8 (Table 1.3) that rating was excluded from the study. Namely, Fair (F) and Poor (P) ratings from both the Mountain and Coastal eco-regions were omitted (Table 1.3).

The Piedmont region had the most stations in which both AWQ and BMRs were monitored (106), followed by the Coastal (45), and the Mountain (42). The regions are discussed in that order.

Table 1.3. The number of AWQ and BMR sites found for each eco-region with paired collection locations

Number of Location Matches			
	Eco-Region		
BMR Rating*	Mountain	Piedmont	Coastal
E	8	8 (1)	10
G	19	20	14
GF	10 (1**)	31 (2)	16
F	4	39 (3)	3
P	1	8 (1)	2

* E = excellent, G = good, GF= good-fair, F = fair, and P = poor ratings

** Values in parentheses represent DWQ stations in which both AWQ and fish populations were monitored. The values outside of parenthesis represent stations in which BMR and AWQ were monitored.

PIEDMONT ECO-REGION. When more data matches exist, as in the Piedmont region, expected trends were more apparent. As expected, median DO concentrations in the Piedmont increased with increased BMR (Table 1.4). Median NO_{2-3}-N, TKN, TN, and TP concentrations all increased with decreased BMR. NH_3-N concentrations typically increased with decreased BMR. Only median TSS and fecal coliform concentrations did not exhibit a noticeable relationship with BMR. Application of the WQABI method appears to be practical for the North Carolina Piedmont eco-region.

Table 1.4. Median Concentrations of Pollutants in the Piedmont Eco-region

PIEDMONT								
	Constituent concentration (mg/L)							
Rating	DO	TSS	NH_3	$NO_{2\text{-}3}$-N	TKN	TN	TP	Fecal (#/100mL)
E	9.25	4.00	0.02	0.39	0.30	0.69	0.06	229.0
G	8.80	6.40	0.04	0.59	0.40	0.99	0.11	103.1
GF	8.40	5.00	0.06	0.67	0.50	1.17	0.13	93.0
F	7.70	7.00	0.06	1.60	0.56	2.16	0.22	169.2
P	6.80	5.00	0.13	6.34	1.25	7.59	0.63	316.4

COASTAL ECO-REGION. While expected relationships were not as apparent in the Coastal eco-region as they were in the Piedmont eco-region, data were promising. As anticipated, median concentrations of $NO_{2\text{-}3}$-N and TKN increased with decreased rating. More importantly, the target regulated nutrients, TN and TP, illustrated the expected results. Median DO concentrations decreased from E to G only. Median TSS concentrations remained the same from E to G and slightly increased from G to GF. Median NH_3 concentrations increased from E to G, yet did not change from G to GF. Finally, fecal coliform levels had no trend (Table 1.5). The WQABI method holds promise for select pollutants in the Coastal eco-region.

Table 1.5. Median Concentrations of Pollutants in the Coastal Eco-region

COASTAL								
	Constituent concentration (mg/L)							
Rating	DO	TSS	NH3	NO_{2-3}-N	TKN	TN	TP	Fecal (#/100mL)
E	7.55	3.00	0.03	0.21	0.40	0.61	0.05	58.1
G	6.15	3.00	0.06	0.23	0.50	0.73	0.09	61.7
GF	6.90	3.50	0.06	0.30	0.52	0.82	0.11	54.8

MOUNTAIN ECO-REGION. Overall, the Mountain eco-region showed lower pollutant levels when compared to the other two eco-regions, yet not all expected trends were evident. As expected, median TSS and TP concentrations increased with decreased ratings. However, median DO concentrations were nearly constant from the E to GF BMR-ratings (Table 1.6). Median NH_3 concentrations stayed the same between the E and G rating, but did increase from G to GF; median TKN concentrations did not change among E, G, and GF ratings. Median NO_{2-3}-N, TN, and fecal coliform concentrations had no obvious trend from E to G to GF (Table 1.4). These relationships suggest that the WQABI method may only work for 1-2 pollutants in assessing ambient water qualities in the Mountain eco-region. Perhaps this is due to a relatively limited data pool. Generally, the macro-invertebrates in the Piedmont eco-region were more tolerant to pollutants than macro-invertebrates in both the Mountain and Coastal ecoregions. For example, both median TN and TP concentrations were lowest in the Mountain region, followed by the Coastal region, with the highest concentrations in the Piedmont for respective BMRs (Tables 1.4, 1.5, & 1.6). Fecal coliforms appear to be lower in the Mountain eco-region than both the Coastal and Piedmont eco-regions. Overall, macro invertebrate tolerance to “dirtier” ambient conditions appears to increase with eco-region as follows: Mountain, Coastal, and Piedmont. This suggests that it is more difficult to design a stormwater treatment system to achieve comparable benthos protection in the Mountain than in the Piedmont. The WQABI method appears to be most applicable for TN and TP, which is fortuitous considering the number of states, including North Carolina, which contain nutrient sensitive waters and have regulations in place to reduce TN and/or TP.

Table 1.6. **Median Concentrations of Pollutants in the Mountain Eco-region**

MOUNTAIN								
	Constituent concentration (mg/L)							
Rating	DO	TSS	NH3	$NO_{2\text{-}3}$-N	TKN	TN	TP	Fecal (#/100mL)
E	10.25	3.50	0.02	0.41	0.20	0.61	0.02	32.6
G	10.30	4.00	0.02	0.21	0.20	0.41	0.03	25.3
GF	10.35	5.40	0.03	0.35	0.20	0.55	0.06	38.5

STATISTICAL ANALYSIS. All statistical analysis for this chapter was completed by a North Carolina State University statistical consultant, Dr. Jason A. Osborne. Dr. Osborne is a co-author on the accepted publication discussed in the abstract. A complete statistical write-up is found in Appendix B and a brief summary is provided below.

While many of the eco-regions had apparent AWQ-BMR relationships, statistical analyses were needed to examine if the trends were significantly correlated. To investigate the potential association between the concentration of various pollutants (DO, TSS, NH_3, $NO_{2\text{-}3}$-N, TKN, TP, and Fecal Coliforms) and benthic macroinvertebrate rating (BMR), cumulative logistic models were considered. In each model, the dependence of long-run (population) frequencies of five BMR values (E, G, GF, F, and P) on compound concentrations and possibly eco-regions were estimated using cumulative logit functions. To illustrate the possible associations, pollutant concentrations were made discrete and tabulated along with BMR ratings using SAS statistical software package (2004). Several positive associations were noted; for instance, the BMRs tended to be better with higher than average concentrations of DO. The output in Table 1.7 gives the p-values for all combinations of eco-regions and pollutants. All significant associations (p-value < 0.05) are in bold.

Most statistically significant associations between pollutant concentrations and BMR are evident in the Piedmont eco-region, specifically for DO, Fecal Coliform, NH_3, $NO_{2\text{-}3}$-N, TKN, TN, and TP. There are pollutants in the Coastal and Mountain regions that may be potential indicators of BMR as well. For instance, $NO_{2\text{-}3}$-N and TP concentrations were both significantly associated with BMRs in the Coastal region, and NH_3, TKN, and TP were significantly associated with BMR in the Mountain eco-region. Overall, the concentration of TP was a significant predictor of BMR in all three eco-regions, and may therefore prove to have the most comprehensive benefit in assessing water quality.

Table 1.7. P-values for the association between all eco-regions and pollutants

Pollutant	Eco-Region	P-value
DO	coastal	0.415
DO	mountain	0.647
DO	**piedmont**	**0.0051**
Fecal	coastal	0.7987
Fecal	mountain	0.1011
Fecal	**piedmont**	**0.0011**
NH3	coastal	0.1172
NH3	**mountain**	**0.0003**
NH3	**piedmont**	**<.0001**
NO2_3	**coastal**	**0.0483**
NO2_3	mountain	0.7822
NO2_3	**piedmont**	**0.0014**
TKN	coastal	0.6577
TKN	**mountain**	**0.0047**
TKN	**piedmont**	**<.0001**
TN	coastal	0.2464
TN	mountain	0.5395
TN	**piedmont**	**0.0026**
TP	**coastal**	**0.0041**
TP	**mountain**	**0.0039**
TP	**piedmont**	**0.0286**
TSS	coastal	0.6971
TSS	mountain	0.3659
TSS	piedmont	0.0668

* Bold values indicate a significant association between AWQ and BMR data.

Future Directions for WQABI

SETTING EFFLUENT STANDARDS FOR DEVELOPMENT. While it is not the intent of this article to develop regulations, the following suggestions are meant to stimulate thought among regulators and developers. If WQABI were implemented to properly design BMPs for a given watershed, target pollutant effluent concentrations need to be established. While excellent water quality conditions are encouraged, it is understood that background local ambient water quality may necessitate exceptions. For example, most Greenfield developments are to be located in areas with cleaner background water quality, so perhaps these developments should be required to release a clean effluent water quality such as that associated with an E or G. A greyfield or infill development, in an area of degraded background water quality, however, may have lower standards, such as GF or F. Even in degraded waters, water quality improvement from development could be required. For example, if local ambient water quality is in P (or poor) condition, a development could be required to release effluent concentrations associated with an F (or fair) condition. Excellent water qualities should be the *ultimate* goal, even if some situations require a longer duration of time to achieve such quality than others.

Effluent water quality standards in portions of the mountains with excellent or good effluent targets may be unattainable if substantially developed, as it is possible that no stormwater BMP can reliably release TP concentrations that are 0.02 or 0.03 mg/L. For example, monitored bioretention cells in North Carolina have been shown to release median effluent TN and TP concentrations of 0.95 mg/L and 0.09 mg/L, respectively (Table 1.8) and would not be able to meet any of the standards set for the Mountain eco-region. Effluent TN standards also would not be met in the Coastal eco-region. One obvious solution is to restrict development in these pristine locations. A bioretention cell in the Piedmont eco-region, unlike the Coastal and Mountain ecoregions, can obtain a G water quality standard for both TN and TP, suggesting that the Piedmont may be the best eco-region for development. It is important to note that all 3 bioretention studies in Table 1.8 occurred in the Piedmont, so the data presented are probably most appropriate for the Piedmont eco-region.

Table 1.8. Effluent concentrations in various bioretention field studies in Piedmont North Carolina.

	Effluent concentration	
Data Source [1]	*TN (mg/L)*	*TP (mg/L)*
Passeport and Hunt, 2009		
Graham, North	0.76	0.05
Graham, South	0.76	0.06
Hunt et al, 2008		
HMBC	1.14	0.13
Sharkey, 2006		
L1 (un-lined)	1.32	0.24
Median	**0.95**	**0.09**

[1] Studies selected represent those in which current design standards (NCDENR, 2009) for bioretention were monitored.

FUTURE RESEARCH NEEDS. Numerous flaws have been documented in using load reduction and percent removals for assessing BMPs (Lenhart and Hunt, 2010; Schaafsma et al, 1998; Strecker et al., 2001; Jones et al., 2008). The WQABI method accounts for some of the flaws associated with conventional BMP assessment methods by relating BMP performance to target ambient water quality concentrations. The purpose of this paper was to introduce this concept; however, many questions about using WQABI persist. For example, what is the appropriate course of action when no BMP can meet desired water quality effluent concentrations? How much variation lies among species collection techniques? Do the various types of indicator species (benthos, fish, periphyton) give equivalent water quality ratings? Further studies are encouraged to ensure consistency in the measurement of biotic health ratings. Detailed reporting of BMP design parameters, such as those required by the International BMP database (Clary et al., 2002), is encouraged to determine optimal BMP design criterion. Furthermore, it is important to recognize that the WQABI method only considers water quality and not quantity. Total Maximum Daily Loads (TMDLs) may be exceeded when runoff volume increases, negating potential benefits of cleaner effluent concentrations. Volume control must be considered *in addition to* the WQABI method to prevent flooding, maintain stable stream morphology, and meet target or pre-development hydrology. While the WQABI method can be completed independent of hydrologic data, it is strongly recommended that hydrologic design guidance be provided in addition to the WQABI method. Also, WQABI works when a substantial number of regional BMP studies reporting effluent concentrations have been conducted or if a reliable, deterministic way of predicting effluent concentrations is employed. It is recognized that for some BMPs in some parts of the U.S.A, a deficiency exists in data collection and BMP performance prediction.

Conclusions and Recommendations

Several conclusions were drawn from this case study:

(1) AWQ data and BMRs can be combined to set target nutrient concentrations that protect high quality waters. Target concentrations can be used as a baseline for BMP assessment. This study examined North Carolina-specific data, but similar analyses could be conducted elsewhere.

(2) Unlike the percent removal method, the WQABI method accounts for irreducible and background concentrations.

(3) Different eco-regions require different regulatory standards. This study showed that the streams in the Mountain eco-region of North Carolina held lower median nutrient levels across all BMRs when compared to streams in the Piedmont and Coastal eco-regions. The higher background water quality implies a need for more stringent regulations on development. Perhaps development in some pristine parts of the Mountain eco-region should be prevented altogether if excellent water quality is to be maintained. A similar trend is evident in the Coastal eco-region, where stream nutrient concentrations are generally lower than those in the Piedmont eco-region.

(4) The WQABI method allows different target conditions/standards to be chosen depending upon existing in-ground conditions. For example, a Greenfield development could be held to a more rigorous standard (an E or G BMR) than a greyfield, infill development (perhaps a GF or F BMR).

(5) The WQABI method is better applied in some eco-regions than others. In , the Piedmont eco-region had more correlated data between AWQ-data and BMRs; however, more AWQ data and BMRs were monitored in the Piedmont compared to the Mountain and Coastal regions. More paired data (AWQ and BMRs) from the Mountain and Coastal eco regions are recommended for further analysis.

(6) The WQABI method works for some, but not all, constituents. For example, NH_3-N, NO_{2-3}-N, TN and TP ambient water quality concentrations frequently were significantly correlated with BMR, whereas, TSS, DO, and fecal coliform concentrations showed little or no relation to BMR.

(7) The WQABI method needs further refinement. Efforts are needed to synchronize benthos collection and identification techniques and more data are needed to differentiate water quality among the five BMRs, particularly among the better water quality ratings. Additionally, hydrologic function, especially reductions in outflow volume *must* be considered and combined with the WQABI method to ensure optimal BMP effectiveness.

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CHAPTER 2. INFLUENT POLLUTANT CONCENTRATIONS AS PREDICTORS OF EFFLUENT POLLUTANT CONCENTRATIONS FOR MID-ATLANTIC BIORETENTION

Introduction

Best Management Practices (BMPs), such as bioretention, stormwater wetlands, wetponds, and sandfilters, are implemented to treat stormwater by mitigating runoff volumes, reducing peak flows, and sequestering or removing pollutants. Each BMP type is distinctive, yet even within BMP types there is much design variation. For example, bioretention designs may include different: in-situ soils, media characteristics, media depths, vegetated covers, under-drain configurations, and inlet features.

In many jurisdictions each BMP is simply credited with TN and TP removal efficiencies based on expected nutrient removal. An efficiency ratio (ER) is calculated by Equation 2.1, using either pollutant concentrations or loads as a basis:

$$\boldsymbol{ER = \frac{inflow\ conc. - outflowe\ conc}{inflow\ conc.}} \qquad \textbf{\textit{Eqn (2.1.)}}$$

A percent removal rate is simply the ER multiplied by 100%. Percent removals are used by designers to select BMPs to meet post-development standards. For example, the North Carolina Department of Environment and Natural Resources (NCDENR) requires all developments to release no more than 0.45 kg/ha/yr for TP and 4.5 kg/ha/yr for TN in the Tar

Pamlico River Basin of North Carolina (NCDENR, 2009). To meet these criteria, engineers select BMPs that remove percents of pollutant loads until the above loading rates are met, often necessitating multiple BMPs in series. In North Carolina, for example, a bioretention cell is credited with a 40% and 45% percent removal of TN and TP, respectively (NCDENR, 2009).

Yet, while percent removals are simple to work with and are now embedded in many regulations, there is no evidence that this simple metric represents the performance of a stormwater BMP. BMP performance percent removals have been reported to vary per geographical location, contributing watershed, and myriad design configurations (Center for Watershed Protection, 2007). Several studies discuss the flaws in using the percent removal metric as a primary metric for BMP evaluation (Barrett, 2008; McNett et al., 2010; Strecker et al., 2001). Using data from the international BMP database (www.bmpdatabase.org), Barrett (2008) examined whether effluent concentrations were independent of influent concentrations, for 4 different types of BMPs for TP, TN, and TSS, briefly discussed in the following paragraph.

A primary criticism of using percent removal as a metric is that more polluted influent runoff may lead to higher removal efficiencies, yet still not provide for surface water quality protection. If effluent concentrations were independent of influent concentrations, removal efficiencies could be abandoned. Barrett (2008) showed a significant relationship between influent and effluent total suspended solid (TSS) concentrations in wet ponds ($p=0.008$) and ascertained a noticeable relationship between influent and effluent TN concentrations in detention basins and swales. Similar relationships were not apparent for any other BMPs nor between influent and effluent TP concentrations for any practice (Barrett, 2008). It is important to note that the BMPs examined by Barrett were located across the United States and Canada, therefore, greatly varying in geography and climate. Applying Barrett's findings to one particular region of the United States (which was not his intention) may be misleading. For instance, if BMPs in the Pacific Northwest displayed a significant relationship between influent and effluent pollutant concentrations, is it guaranteed that BMPs in the Mid-Atlantic United States will have similar results? Furthermore, Barrett (2008) found relationships for a few types of BMPs, but was unable to mine and evaluate sufficient data for bioretention.

Another concern is that total pollutant loads are based on the product of volume and (mean) concentration. Emphasizing only the concentration reduction does not fully account for the benefits of infiltration from a BMP such as bioretention.

Finally, as somewhat discussed above, while percent removals can be mechanistically supported by steady state unit treatment processes, this is not true in the highly unsteady state environment of a stormwater BMP (Wild and Davis, 2009). The great variability in influent flow and concentrations will provide a wide range of performance results that cannot be captured as a simple percent removal metric.

To address these issues, this study specifically addresses whether bioretention cells in the Mid-Atlantic United States have a significant linear, power, exponential or logarithmic correlation between influent and effluent nutrient concentrations; and if so, to what extent are effluent concentrations predicted by influent concentrations. Such a relationship may be mathematically represented as follows:

$$\boldsymbol{Effluent\ Concentration} = \boldsymbol{f}\ (\boldsymbol{Influent\ Concentration}) \qquad \boldsymbol{Eqn}\ \ \textbf{\textit{(2.2.)}}$$

Elevated nutrient concentrations are particularly important in the Mid-Atlantic United States, where excess TN and TP have resulted in many well-documented fish kills, including those in the Chesapeake Bay (USEPA, 2008) and North Carolina’s Albemarle and Pamlico Sounds (APNEP, 2009). The health and diversity of aquatic species is threatened, and the fishing and beach tourism in the region is greatly impacted by elevated nutrient levels.

BIORETENTION. Bioretention, also known as a rain garden, is a BMP frequently integrated into low-impact development (LID) for its ability to reduce stormwater pollutant loads and for its aesthetic appeal in landscaping. Bioretention is a mulch/soil/plant-based BMP that typically contains an engineered soil media ranging from 0.7 to 1 meter in depth (Davis et al., 2009). In North Carolina, the recommended media composition is 85-88% sand, 3-5% organic matter, and 8-12% fine particles (Hunt and Lord, 2006; NCDENR, 2009). Media composition is widely variable among jurisdictions outside NC (Davis et al., 2009). While the media provides the primary mechanism for pollutant removal and comprises the majority of a bioretention cell, there is also a surface mulch layer, assorted vegetations types, and an underdrain surrounded by a gravel envelope (Figure 2.1). Appropriate devices are installed for inflow, outflow, and overflow.

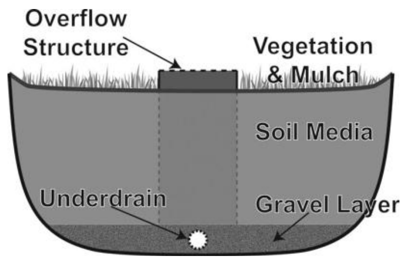

Figure 2.1. Diagram of a typical bioretention cell cross section.

Many studies have been conducted showing the high efficiency of pollutant removal exhibited by bioretention. Metals, nutrients, oil and grease, and thermal pollution can all be mitigated through the use of bioretention (Davis et al., 2003; Davis, 2007; Hsieh et al., 2007; Hsieh and Davis, 2005; Hunt et al., 2008; Jones and Hunt, 2009). Hydrologic benefits such as groundwater recharge and peak flow reduction are also evident (Davis, 2008; Li et al., 2009). Davis et al. (2009) gives an overview of the most current bioretention research findings, including specifics on design guidelines. They also conclude that while performance of bioretention is well researched, the specific design parameters responsible for optimal performance have not been studied as extensively.

Site Descriptions

Influent and effluent TN (TNin and TNout) concentrations and influent and effluent TP (TPin and TPout) concentrations are assessed using data collected from 11 bioretention cells at 7 field sites, monitored in the Mid-Atlantic United States. Brief descriptions for each BMP examined are provided in this section. Nomenclature from the original studies is used in this paper. All sites selected were of somewhat similar design and are summarized in Table 2.1. Data from published studies of bioretention cells with outdated and, thus, no longer recommended designs, such as those with poor (P-laden) fill media (Hunt et al., 2006), were not included in this study. One cell included in this analysis, L2, contained an impermeable plastic liner, which is not a typical design component, but such a feature may still be recommended at a brownfield site where hazardous leachate is of concern (Hunt and Lord, 2006).

COLLEGE PARK, CP. Located in College Park, Maryland, CP was monitored from April, 2006 to July, 2007. The cell received drainage from a 0.26 hectare watershed composed of parking lot and roadway surfaces. CP had a surface area of 181 m^2 and a media depth ranging from 50-80 cm (Li and Davis, 2009; Li et al., 2009).

GRAHAM, GRAHAM N AND GRAHAM S. Both Graham N and Graham S cells were located in Graham, North Carolina, and were monitored from September 2006 to August 2007. Runoff from a 0.36 ha catchment area of 33% asphalt parking lot and 60% lawn drained to a forebay and was equally diverted to the two cells. Both cells had surface areas of 227 m^2 with media depths of 75 cm and 105 cm (including 15 cm of gravel in each), for Graham N and Graham S, respectively (Passeport et al., 2009).

HAL MARSHAL BIORETENTION CELL, HMBC. The HMBC cell was located in Charlotte, North Carolina and was monitored from February, 2004 to March, 2006. The HMBC received drainage from a 0.37 ha asphalt parking lot and had a surface area of 229 m^2, which included a 3m diameter forebay at the cell inlet. The cell media depth was 120 cm (Hunt et al., 2008).

LOUISBURG, L1 AND L2. Located in Louisburg, North Carolina, cells L1 and L2 were continuously monitored from March, 2004, to December, 2004. L1 received drainage from a 0.36 ha parking lot and had a surface area of 162 m^2, while L2 received drainage from a 0.22 hectare catchment comprised of both a parking lot and ball field and had a smaller surface area of 99 m^2. L2 was lined with an impermeable liner to limit or eliminate ET as part of a prior study. The media depth for both cells ranged from 50-60 cm (Li et al., 2009; Sharkey, 2006).

NASHVILLE, NV_2 AND NV_3. Two cells located in Nashville, North Carolina, were monitored in an un-maintained/clogged state from April 3, 2008 to March 10, 2009 (Brown and Hunt, 2009). Media depths for NV_2 and NV_3 were 0.6 m (2 ft.) and 0.9 m (3 ft), respectively. NV_2 received runoff from 0.66 hectares of parking lot and NV_3 received runoff from 0.43 hectares of parking lot. Surface areas for NV_2 and NV_3 were 289 m^2 and 206 m^2, respectively. Both cells were monitored after a major maintenance overhaul, leaving them in an un-clogged state from March 11, 2009 to March, 2010. Only data collected *after* cell maintenance were included in this study.

ROCKY MOUNT, RM GRASS AND RM MULCH. Two cells located in Rocky Mount, North Carolina, were monitored from September 14, 2007 to December, 2009. RMgrass was a grassed cell, while RMmulch was covered by mulch, trees, perennials, and shrubs. RMgrass received drainage from 0.22 hectares of parking lot (75% impervious), and RMmulch received runoff from 0.25 hectares of parking lot (72% impervious). RMgrass and RMmulch had surface areas of 146 m^2 and 142 m^2, with media depths of 1.1 m and 0.96 m, respectively. Both cell media depths included 0.15 m of gravel and contained internal water storage zones (IWS) initially 1.02 m and 0.87 m deep that were reduced 0.3 m on January 12, 2009. No differentiation was made among the data from a water quality perspective.

SILVER SPRING, SS. Located in Silver Spring, Maryland, cell SS was monitored from April, 2006 to June, 2007. The cell received drainage from a 0.45 hectare watershed composed of a parking lot and a driveway, and had a surface area of 102 m^2. The media depth was 90 cm (Li and Davis, 2009; Li et al., 2009).

All cell configurations included a 0.25 mm tipping bucket rain gauge (ISCO 674 or Global water model), automated samplers with data loggers (ISCO 6712, ISCO 6712FR, or Sigma 900 max) and an outflow weir. Data from all sites were collected similarly. Installed automated samplers coupled with data loggers automatically collected flow-weighted and rainfall composite samples to determine event mean concentrations. All site samples were sealed immediately, placed on ice, and transported to a certified laboratory for analysis. All samples were analyzed within 24 hours of collection. More detailed descriptions of sample collection, analysis, and QA/QC are found in sources listed in Table 1; however, unpublished data are mentioned below.

Table 2.1. Cell and Watershed Characteristics for all cells included in Ch. 2 Study.

Site	CP	Graham, N	Graham, S	HMBC	L1	L2	Nashville 2 ft (fixed)	Nashville 3 ft (fixed)	RM, grass	RM, mulch	SS
Location	College Park, Md.	Graham, N.C.	Graham, N.C.	Charlotte, N.C.	Louisburg, N.C.	Louisburg, N.C.	Nashville, NC	Nashville, NC	Rocky Mount, NC	Rocky Mount, NC	Silver Spring, Md.
Reference	Li and Davis, 2009; Li et al., 2009	Passeport et al., 2009	Passeport et al., 2009	Hunt et al., 2008	Li et al., 2009	Li et al., 2009	Brown and Hunt, 2010	Brown and Hunt, 2010	North Carolina State University, field data.	North Carolina State University, field data.	Li and Davis, 2009; Li et al., 2009
Watershed	Anacostia	Cape Fear	Cape Fear	Catawba	Tar-Pamlico	Tar-Pamlico	Tar-Pamlico	Tar-Pamlico	Tar-Pamlico	Tar-Pamlico	Anacostia
Watershed (ha)	0.28	0.36	0.36	0.37	0.36	0.22	0.68	0.43	0.22	0.25	0.45
Surface Area (m^2)	181	227	227	229	162	99	290	206	146	142	102
SA/DA	0.06	0.03	0.03	0.06	0.05	0.05	0.04	0.05	0.07	0.06	0.02
% Impervious	90%	40%	40%	95%	95%	45%	83%	97%	76%	72%	90%
Watershed Composition	parking lot, roadway	33% asphalt parking lot, 60% lawn	33% asphalt parking lot, 60% lawn	asphalt parking lot	parking lot	parking lot, ballfield	parking lot	parking lot	parking lot	parking lot	parking lot, driveway
General cell shape	trapezoid	oval	oval	rectangle	oval	rectangle	rectangle	rectangle	rectangle	oval	triangle
media depth (m)	0.5-0.8	0.75	1.05	1.2	0.5-0.6	0.5-0.6	0.6	0.9	1.25	1.11	0.9
Year Built	2004	2005	2005	2003	2004	2004	2008	2008	2005	2005	2006
In-Situ Soil	-	loamy clay	sandy loam	-	clay	clay	-	-	-	-	-
Soil Media Type	sandy loam	expanded slate	expanded slate	loamy sand	sandy loam	sandy loam	loamy sand	loamy sand	sand	sand	sandy clay loam
Vegetation	trees/shrubs/mulch	bermuda grass	bermuda grass	trees/shrubs/mulch	trees/shrubs/mulch	trees/shrubs/ mulch	trees/shrubs/perennials/mulch	trees/shrubs/perennials/mulch	centipede sod	shrubs/ perennials/ mulch	trees/shrubs/mulch
Ponding Depth (cm)	10-34	25	25	23	15	15	13	15	16	13	30
Underdrains	2 @ 15 cm Perforated PVC	2 @ 15 cm Perforated PVC	2 @ 15 cm Perforated PVC	1 @ 0.15 m (6 in) corrugated plastic	2 @ 15 cm corrugated Plastic	3 @ 15 cm corrugated Plastic	4 @ 15cm PVC	4 @ 15cm PVC	N/A	N/A	2 @ 15 cm Perforated PVC
Distinguising Features	none	IWS	IWS	none	none	Lined with impermeable plastic	3 cells connected through 1 outlet	2 cells connected through 1 outlet	IWS	IWS	none
Internal Water Storage Zone Depth (cm)	No ISZ	45 cm (30 cm media, 15 cm gravel)	75 cm (60 cm media, 15 cm gravel)	No IWS	No IWS	No IWS	No IWS	No IWS	60 cm (9/07-1/09), then 30 cm (1/09-Present	60 cm (9/07-1/09), then 30 cm (1/09-Present	No IWS
P-Index	N/A	5	8	6 (low)	1-2(low)	1-2(low)	3-6 (low)	3-6 (low)	9-11 (low)	6-15 (low)	N/A

Rocky Mount cells contained an ISCO 674 rain gauge and a collection trough to temporarily pond a representative sample of runoff. If runoff depths reached 0.51 mm in the trough within 15 minutes, as determined by a 730 model bubbler module, a sample was automatically taken using an ISCO 6712 automated sampler. As with all sites included in this study, immediately following collection, samples were sealed, placed on ice, and transported to a certified lab. All Rocky Mount samples were analyzed by North Carolina State University's Center for Applied Aquatic Ecology (CAAE) laboratory within 24 hours of collection. Lastly, all runoff volumes at the Rocky Mount cells were estimated using the initial abstraction method.

Statistical Methodology

SAS Statistical software package (2004) was used for the analysis with a 95% level of confidence ($\alpha = 0.05$). Scatter-plots were created to visually inspect the relationship between TNin versus TNout and TPin versus TPout, and histograms were constructed to investigate the distribution of data. The Cramer-von Mises test was used to test the null hypothesis of normally distributed data for 4 different variables (TNnormal or TPnormal) and log transformed data set (logTNnormal or logTPnormal) (Appendix C). Variables represent differences between respective influent and effluent TN and TP concentrations. Representative equations are illustrated in equations 2.3 and 2.4:

$$\boldsymbol{TNnormal = TNin - TNout} \qquad \textit{\textbf{Eqn. (2.3.)}}$$

$$\log(TNnormal) = log_{10}(TNin) - log_{10}(TNout) \qquad \textit{Eqn. (2.4.)}$$

A general linear model (GLM) was constructed using influent concentrations as predictors for effluent concentrations (Equation 2), for both normal and log-normal TN and TP data (Cramer-von Mises, $p > 0.05$). A t-test was also performed on normal data to discern which cells, if any, significantly reduced or increased nutrient concentrations. Non-normal data were analyzed using Kendall's tau coefficient for non-parametric statistics which tested two null hypotheses: (1) an association exists between TNin and TNout and (2) an association exists between TPin and TPout. High tau values and lower p-values ($p < 0.05$) indicate an association between the two variables being tested. Both the pooled set of data and individual sites of sample sizes greater than 8 ($n>8$) were analyzed separately in the manner described above. An example of statistical code and respective output from SAS is provided in Appendix C.

A second analysis was conducted where composited data were ranked in order of increasing magnitude and separated into 2 groups using average parking lot runoff concentrations of TN and TP (defined by Passeport and Hunt, (2009)) as points of separation. TNin and TNout data pairs were separated into those with TNin concentrations greater than or equal to 1.57 mg/L (median TN parking lot concentration found by Passeport and Hunt) and those with TNin concentrations less than 1.57 mg/L. Similarly, TP data were segregated using Passeport and Hunt's (2009) median concentration of 0.19 mg/L. Concentrations from each group (above median and below median) were then averaged (mean), and mean differences (TNin-TNout and TPin-TPout) and associated removal efficiencies were calculated to assess the relative relationship between influent and effluent concentrations when influent nutrient concentrations were either greater than or less than “normal.”

Results and Discussion

POOLED STATISTICS. One hundred and fifty-four (n=154) TNin concentrations ranged from 0.1 mg/L to 7.3 mg/L and had a median and mean concentration of 1.47 mg/L and 1.74 mg/L, respectively. One hundred and forty-six (n=146) TNout concentrations ranged from below the detectable limit (BDL) of 0.1 mg/L to 7.20 mg/L and had a 1.00 mg/L median and 1.16 mg/L mean concentration. For statistical evaluation, Clausen and Spooner (1993) suggest that all data below the detection limit have a value of one-half the BDL, in this case, 0.05. One hundred and thirty-four (n=134) TPin concentrations ranged from 0.005 mg/L to 0.92 mg/L with median and mean concentrations of 0.14 mg/L and 0.19 mg/L, respectively. One hundred and twenty-three (n=123) TPout concentrations ranged from 0.03 mg/L to 1.30 mg/L with median and mean concentrations of 0.11 mg/L and 0.17 mg/L, respectively. Individual site statistics are listed in Tables 2.2. and 2.3.

Table 2.2. TN site statistics including: range, median and mean concentrations in mg/L of influent and effluent and respective sample size (n).

	TNin				TNout			
Site	n	range (mg/L)	median (mg/L)	mean (mg/L)	n	range (mg/L)	median (mg/L)	mean (mg/L)
CP	12	0.1 - 7.3	1.55	1.88	12	0.74 - 7.2	2.50	2.81
Graham, N	14	0.381 - 4.082	1.33	1.77	13	0.05 - 1.478	0.70	0.73
Graham, S	24	0.381 - 4.083	1.17	1.54	19	0.12 - 1.683	0.79	1.54
HMBC	27	0.63 - 7.0	1.46	1.86	23	0.59 - 2.82	1.06	1.14
L1	30	0.31 - 4.01	2.11	1.96	30	0.37 - 2.51	1.55	1.38
L2	12	0.1 - 7.3	1.75	2.19	13	0.77 - 1.92	1.15	1.20
Nashville*	2	-	-	-	2	-	-	-
RMgrass	8	0.401 - 1.201	0.73	0.79	8	0.198 - 0.595	0.37	0.37
RMmulch*	3	-	-	-	3	-	-	-
SS	9	0.1 - 1.6	1.00	0.83	9	0.3 - 1.1	0.60	0.66
Pooled Data	**154**	**0.1 -7.3**	**1.47**	**1.74**	**146**	**0.05 - 7.2**	**1.0**	**1.16**

* Nashville and RMmulch were not analyzed individually due to low sample sizes, n=2 and n=3, respectively.

Table 2.3. TP site statistics including: range, median and mean concentrations in mg/L of influent and effluent and respective sample size (n).

	TPin				TPout			
Site	n	range (mg/L)	median (mg/L)	mean (mg/L)	n	range (mg/L)	median (mg/L)	mean (mg/L)
CP	6	0.005 - 0.7	0.10	0.20	6	0.04 - 1.3	0.35	0.31
Graham, N	14	0.039 - 0.399	0.08	0.15	13	0.03 - 0.111	0.05	0.05
Graham, S	17	0.039 - 0.399	0.08	0.14	11	0.041 - 0.1	0.06	0.06
HMBC	27	0.07 - 0.74	0.13	0.20	22	0.05 - 0.45	0.09	0.13
L1	30	0.07 - 0.92	0.33	0.33	30	0.12 - 0.8	0.18	0.23
L2	12	0.005 - 0.37	0.15	0.15	13	0.04 - 1.3	0.12	0.25
Nashville*	2	-	-	-	2	-	-	-
RMgrass	8	0.035 - 0.082	0.05	0.05	8	0.06 - 0.1	0.08	0.08
RMmulch*	3	-	-	-	3	-	-	-
SS	9	0.05 - 0.2	0.05	0.08	9	0.05 - 0.1	0.05	0.07
Pooled Data	**128**	**0.005 - 0.92**	**0.14**	**0.19**	**117**	**0.03 - 1.3**	**0.11**	**0.17**

* Nashville and RMmulch were not analyzed individually due to low sample sizes, n=2 and n=3, respectively.

Basic scatter-plots of influent versus effluent TN and TP concentrations showed various outliers and minimal correlation between influent and effluent concentrations. Each graph was curve-fitted using a power function, an exponential function, and a polynomial with a maximum order of 2 in order to achieve the best fit (Appendix F). TN data were best correlated using an exponential function (R^2 = 0.11), while TP data were best predicted using a power function (R^2= 0.024); however, even the “best” models in both cases were poor (Figure 2.2).

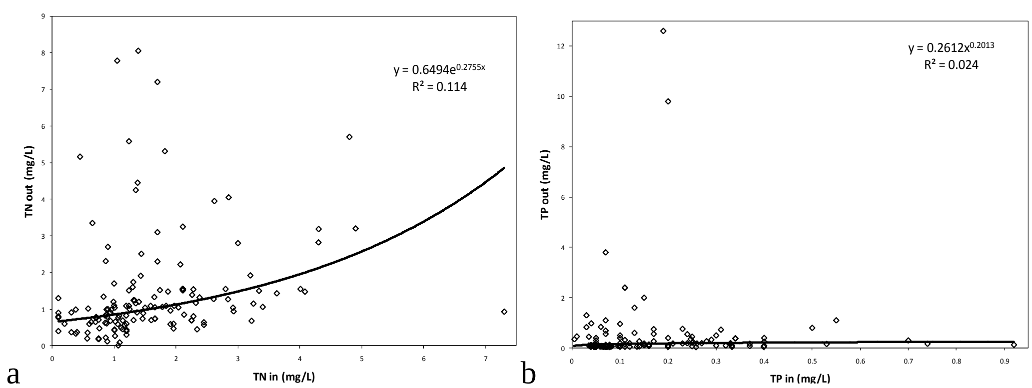

Figure 2.2. (a) Influent TN concentrations (mg/L) versus respective effluent TN concentrations (mg/L). (b) Influent TP concentration (mg/L) versus respective effluent TP concentrations (mg/L).

The Cramer-von Mises test revealed that neither raw nor log-transformed TN and TP data were normally distributed; therefore, non-parametric statistics were used for assessment. Kendall’s test showed a “fair” association between TNin and TNout (τ = 0.35) and a “poor” association between TPin and TPout (τ = 0.26). The abnormality of data distribution did not allow for the creation of a general linear model to predict effluent TN and TP concentrations. These data confirm weak, if any, association between influent and effluent concentrations.

GENERAL LINEAR MODELS AND ASSOCIATION. Twelve General Linear Models (GLMs) were created using influent concentrations as predictors of effluent concentrations. Stronger relationships, that is, those where influent concentrations were predictive of effluent concentrations, resulted in higher R^2 values. Models were arbitrarily characterized according to R^2 values, as follows: none (< 0.1), poor (0.1 - 0.33), fair (0.33 - 0.67), good (0.67 - 0.9), and very good (> 0.9). An example of each model strength is provided in Appendix D. Similarly, four non-normal datasets were ranked according to association using tau values in place of R^2; however, due to the ambiguity of associations, when compared to GLMs, tau values will not be discussed in detail. All model parameters are listed in Table 2.4.

Table 2.4. Statistical Analysis of Individual Sites included in Study.

	Composite		CP		Graham, N		Graham, S		HMBC		L1		L2		RMgrass		SS	
	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP
Primary [1] Assesment Method	[2] Kendall's tau	Kendall's tau	[3] GLM	GLM	GLM	GLM	GLM	GLM	GLM	GLM	Kendall's tau	Kendall's tau	GLM	Kendall's tau	GLM	GLM	GLM	Kendall's tau
Slope	-	-	logTNout = 0.34 (logTNin) + 0.35	TPout = 0.098 (TPin) + 0.29	TNout = 0.22 (TNin) - 0.075	TPout = -20.57 (TPin) + 3.1	logTNout = -0.07 (logTNin) - 0.17	logTPout = 0.296 (logTNin) - 0.89	TNout = 0.38 (TNin) + 0.497	logTPout = 0.537 (logTPin) - 0.53	-	-	logTNout = 0.11 (logTNin) + 0.048	-	TNout = 0.04 (TNin) + 0.34	TPout = -0.92 (TPin) + 0.13	TNout = 0.0013 (TNin) + 0.654	-
Intercept	-	-									-	-						-
Result	**$\tau = 0.35$**	$\tau = 0.26$	**$R^2 = 0.49$**	$R^2 = 0.017$	$R^2 = 0.17$	$R^2 = 0.19$	$R^2 = 0.003$	$R^2 = 0.31$	**$R^2 = 0.52$**	$R^2 = 0.32$	**$\tau = 0.4$**	$\tau = -0.019$	$R^2 = 0.2$	**$\tau = 0.52$**	$R^2 = 0.007$	**$R^2 = 0.72$**	$R^2 = 0.000007$	**$\tau = 0.76$**
Strength of Association/ Model	**fair**	poor	**fair**	very poor	poor	poor	very poor	poor	**fair**	poor	**fair**	very poor	poor	**fair**	very poor	**good**	very poor	**good**
Secondary [1] Assesment	N/A	N/A	[3] t-test	t-test	t-test	t-test	t-test	t-test	t-test	t-test	N/A	N/A	t-test	t-test	t-test	t-test	t-test	N/A
Result	-	-	Pooled p=0.097	Satterthwaite p=0.12	Satterthwaite p=0.2	Pooled p=0.007	Satterthwaite p=0.002	Pooled p=0.03	Pooled p= 0.02	Satterthwaite p=0.03	-	-	Pooled p= 0.5	-	Satterthwaite p=0.0043	Satterthwaite p=0.0026	Satterthwaite p=0.41	-
Significant Differences	-	-	none	none	none	TPin >> TPout	TNin >> TNout	TPin >> TPout	TNin >> TNout	TPin >> TPout	-	-	none	none	TNin >> TNout	TPin >> TPout	none	-

[1] General Linear Models and Kendall's tau test for non-parametric statistics was used as a primary assessment. T-tests were used a secondary assessment.

[2] Kendall's tau test for non-parametric statistics is used when the assumption of normality is violated.

[3] Simple t-tests and general linear models (GLM) can only be used with normally distributed data sets.

* All models where influent concentrations account for 33%, or more, of the variation in effluent concentrations are highlighted in bold.

** All non-parametric models where tau is greater than 0.33 are highlighted in bold.

INDIVIDUAL SITE STATISTICS. In keeping with results from pooled data statistics, individual sites, taken as a whole, weakly support a relationship between influent and effluent concentrations. Of 7 GLMs created for TN, 3 models were “very poor” (Graham S, RMgrass, and SS) with a median R^2 of 0.3%, 2 models were “poor” (L2 and Graham, N) with an R^2 of 19%, and 2 models were “fair” (HMBC and CP) with a median R^2 of 51%.

Of 5 GLMs created for TP, 1 model was “very poor” (CP) with an R^2 of 1.7%, 3 were “poor” (Graham N, Graham S, and HMBC) with a median R^2 of 31%, and 1 was “good” with an R^2 of 72%.

Of 5 GLMs created for TP, 1 model was “very poor” (CP) with an R^2 of 1.7%, 3 were “poor” (Graham N, Graham S, and HMBC) with a median R^2 of 31%, and 1 was “good” with an R^2 of 72%.

Four non-parametric tests were also conducted: 1 for TN (L1) and 3 for TP (L1, L2 and SS). L1 exhibited a “fair” association between TNin and TNout and a “very poor” association between TPin and TPout. L2 showed a “fair” association between TPin and TPout and SS showed a “good” association between TPin and TPout (Table 2.4). The reasons why some influent-effluent relationships were stronger than others are examined in the following section. Furthermore, exceedance probability plots are provided in Appendix E for each site.

Model Strength and Bioretention Cell Parameters

This study further examines whether various design parameters enable influent concentrations to have more influence on effluent concentrations. Three parameters potentially impacting an influent concentrations' ability to predict effluent concentrations were examined. Two of which are cell dependent (media depth, SA/DA) and one of which is watershed dependent (percent imperviousness). Again, data were curve-fitted using a power function, an exponential function, and a polynomial with a maximum order of 2 to achieve the best fit. If such parameters do influence the relationship between influent and effluent nutrient concentrations, then perhaps the removal efficiency metric is more applicable for assessing the performance of bioretention cells than an effluent concentration metric.

1. ***DOES MEDIA DEPTH OF A CELL INFLUENCE THE DEGREE TO WHICH EFFLUENT NUTRIENT CONCENTRATIONS ARE PREDICATED UPON INFLUENT NUTRIENT CONCENTRATIONS?***

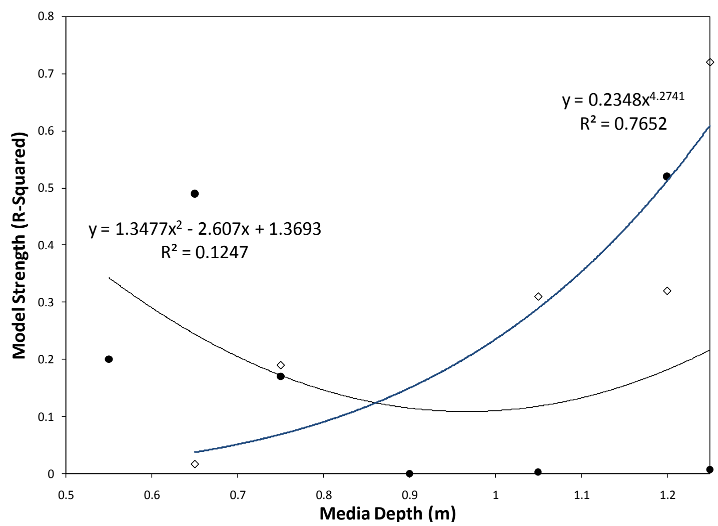

Figure 2.3. TN (solid circle) and TP (unfilled circle) GLM model strengths (R^2), versus media depth (m).

The relationship between influent TN concentrations' ability to predict effluent TN concentrations and cell media depth was best represented through a second order polynomial function (Figure 2.3, R^2 = 0.13) as shown by Equation 2.5:

$$\boldsymbol{Model\,Strength = 1.345\,(Media\,Depth)^2 - 2.607\,(Media\,Depth) + 1.3693} \qquad \textbf{\textit{Eqn. (2.5.)}}$$

Where, "Model Strength" is the ability of influent TN concentrations to predict effluent TN concentrations. According to the best fit model, the ability of influent concentrations to predict effluent concentrations weakens with increasing media depth, yet begins to strengthen with increasing media depth beyond a tipping point. THE model's R2 is minimal and furthermore, the shape of the model and what it projects make no sense physically. The model illustrates that media depth probably has no impact on the influent concentration's ability to predict effluent concentrations. Conversely, TP showed that with an increase in media depth, influent concentrations were more able to predict effluent concentrations. Such a relationship is best represented through a power function (Equation 2.6):

$$\boldsymbol{Model\,Strength = 0.235\,(Media\,Depth)^{4.27}} \qquad \textbf{\textit{Eqn (2.6.)}}$$

It might be expected that deeper media cells would provide more "buffering capacity" to effluent concentrations than shallow media cells, and, therefore, have effluent concentrations less dependent on influent concentrations. This phenomenon was not observed when examining the best fit curves. Li et al. (2009) discussed how cells with deeper media depths had increased media volumes and were able to perform better in nearly all hydrologic metrics considered, when compared to shallow-depth cells. Nitrate and TKN, however, have been shown to be most effectively removed in the upper-most layers of bioretention cells (Davis et al., 2006) and in areas where nitrification occurs, as in the IWS zones (Davis et al., 2009). TP concentrations were shown to decrease from the upper to lower portion of a cell (Davis et al., 2006); however, the authors postulate that TP concentrations will reach a system specific irreducible concentration at a given depth. Thus, an excessive cell media depth will not further reduce nutrient concentrations, nor will it result in a stronger relationship between influent and effluent nutrient concentrations. Perhaps this minimum depth needed for concentration reduction, for example 0.6 m, was reached by all the sites studied.

2. ***DO SA/DA VALUES DICTATE THE DEGREE TO WHICH INFLUENT NUTRIENT CONCENTRATIONS PREDICT EFFLUENT NUTRIENT CONCENTRATIONS?***

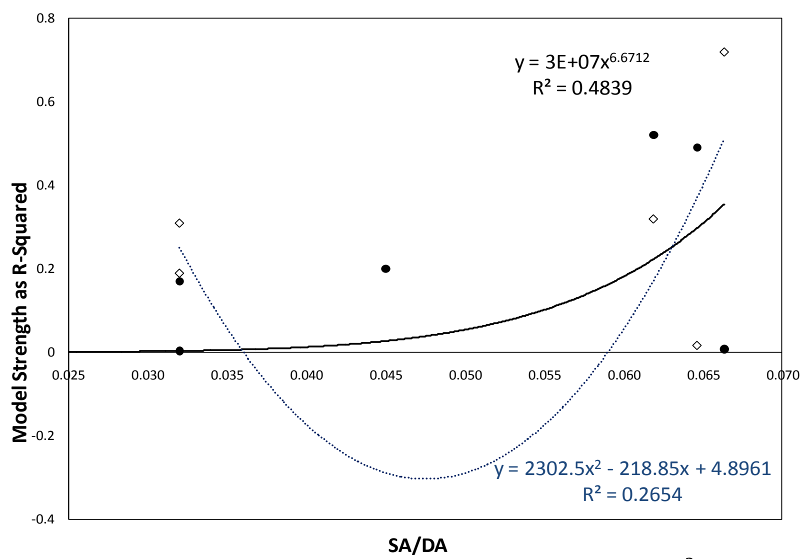

Figure 2.4. TN (solid circle) and TP (unfilled circle) model strengths (R^2), versus SA/DA.

Higher SA/DA ratios could be expected to provide more contact time for influent runoff treatment, which in turn should lead to more consistent effluent concentrations, thereby decreasing influent nutrient concentration's ability to predict effluent nutrient concentrations. However, this trend is absent with respect to TP, and reversed with respect to TN (Figure 2.4). Cells with larger SA/DA ratios were more likely to have effluent TN concentrations influenced by influent TN concentrations, as evidenced by the best fit power function (Equation 2.7, $R^2 = 0.48$):

$$\boldsymbol{Model\,Strength = 3^{7} \times \left(\frac{SA}{DA}\right)^{6.67}} \qquad \textbf{\textit{Eqn (2.7).}}$$

This reverse relationship is somewhat weak, as evidenced by cells CP and HMBC, both with similar SA/DA ratios, 0.06 and 0.07, respectively, yet each had different predictive capabilities. Influent TN concentrations at CP predict effluent TN concentrations to a "fair" degree, yet influent TN concentrations at HMBC have no prediction capability for effluent TN concentrations. Overall, the reverse relationship exhibited by TN is probably the result of confounding factors.

3. *DOES PERCENT IMPERVIOUSNESS INFLUENCE HOW WELL INFLUENT CONCENTRATIONS PREDICT EFFLUENT CONCENTRATIONS?*

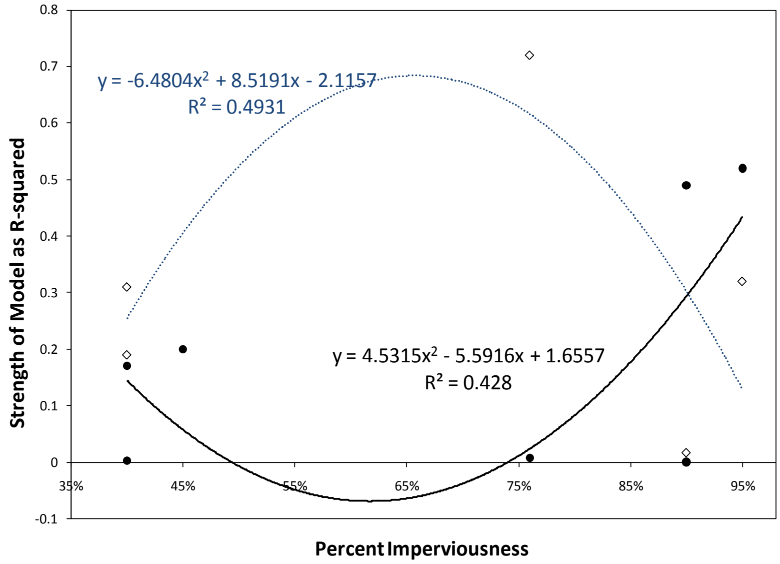

Figure 2.5. TN (solid circle) and TP (unfilled circle) model strength (R^2) versus percent imperviousness of a bioretention cell's watershed.

As percent imperviousness increases, pollutant loads would be expected to also increase, which would affect an influent concentration's ability to predict effluent concentrations. Ceteris paribus, increased percent imperviousness may lead a bioretention cell to be overwhelmed with influent runoff, thereby diminishing contact time of water-borne pollutants inside the cell. Less treatment occurs and effluent nutrient concentrations are more likely predicated upon influent concentrations. This expected relationship was observed, albeit slightly, with TN (R^2 = 0.43, Figure 2.5) and evidenced by cells Graham N and Graham S, both with the smallest percent of imperviousness and the weakest relationship between influent and effluent TN concentrations. As with the other parameters considered, TP data exhibited a trend reverse those of TN. Again, notwithstanding an R^2 of 0.49, the predictive curve does not make rational sense. Clearly, percent imperviousness had little influence on influent TP concentrations' ability to predict TP effluent concentrations (Figure 2.5).

Influent nutrient concentration magnitude and its impact on effluent nutrient concentrations

When TN concentrations in runoff were lower than, or equal to, those from an average parking lot (≤1.57 mg/L) (Passeport and Hunt, 2009), the median removal efficiency was 7%, however, when TN concentrations in runoff were higher than an average parking lot (> 1.57 mg/L), the median removal efficiency was 39% (Table 2.5). The median system performance, as measured by the percent removal metric, increased dramatically with increased TNin concentrations. The same phenomena was true for TP, where TP concentrations lower than, or equal to, average (≤ 0.19 mg/L) resulted in a 39% median *increase* from inlet to outlet, and TP concentrations higher than average (>0.19 mg/L) resulted in a 45% median removal efficiency (Table 2.6).

Table 2.5. Percent Removal of TN as a function of influent TN concentration

	TN in		TN out		Difference		Percent Removal	
	Mean	**Median**	**Mean**	**Median**	**Mean**	**Median**	**Mean**	**Median**
(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	%	%
TN in ≤ 1.57	0.94	1.00	1.23	0.81	-0.29	0.19	-78	7
TN in > 1.57	2.57	2.11	1.65	1.48	0.92	0.63	29	39

Table 2.6. Percent Removal of TP as a function of influent TP concentration

	TP in		TP out		Difference		Percent Removal	
	Mean	**Median**	**Mean**	**Median**	**Mean**	**Median**	**Mean**	**Median**
(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	%	%
TP in ≤ 0.19	0.08	0.07	0.49	0.10	-0.40	-0.03	-567	-39
TP in > 0.19	0.34	0.33	0.41	0.18	-0.08	0.15	-64	45

Removal efficiencies for the "cleaner" mean TP (TPin ≤ 0.19 mg/L) increased; whereas, those for the "dirtier" mean TP (TPin > 0.019 mg/L) decreased, or "got worse." (Table 2.6). This provides evidence that TP effluent concentrations from bioretention cells will tend toward an "irreducible" effluent concentration. Moreover, influent runoff, if cleaner than the irreducible effluent concentration, may be made dirtier due to the soil and biological properties within the bioretention cell. Lenhart and Hunt (2010) provide evidence for this phenomenon through a field study conducted at a stormwater wetland in River Bend, North Carolina. The wetland increased TP concentrations, yet had *lower* effluent TP concentrations than those found in surrounding receiving waters. Lenhart and Hunt (2010) hypothesized that biological activity in the wetland may produce TP effluent concentrations greater than those of influent water.

The existence of a "standard" TN effluent concentration was not as pronounced as that for TP. However, the bioretention cell data presented herein suggest that influent concentrations lower than 1.0 mg/L are unlikely to see substantial improvement. When higher TN concentrations entered, bioretention cells were unable to provide treatment of TN to an apparent irreducible concentration.

Conclusions

1. Effluent TP concentrations appear to be independent of influent TP concentrations using every analysis considered. Authors suggest this is because there was an adequate media volume used in all sites analyzed. Consequently, using an effluent concentration metric to describe bioretention performance for TP treatment is preferred to using the removal efficiency metric.
2. Bioretention cells' effluent TN concentrations are moderately influenced by influent TN concentrations. Of 7 GLMs created for TN, 3 were considered "fair," 2 were "poor," and 3 were "none." TN removal is sensitive to contact time within bioretention cells. So cells that provide minimal contact time may explain why influent TN concentrations can be partially predictive of effluent TN concentrations. The *sole* use of a TN effluent concentration metric for bioretention performance assessment is not recommended; however, much like the percent removal metric, the effluent concentration metric may be a *component* of useful bioretention performance assessment.
3. Relatively "dirtier" influent nutrient concentrations resulted in somewhat "dirtier" effluent nutrient concentrations, but yielded substantially higher removal efficiencies, particularly for TN. When influent TN concentrations were less than that of average runoff observed from parking lots in North Carolina (Passeport and Hunt, 2009), effluent TN concentrations appeared to be tending toward an irreducible effluent concentration. Relatively dirty TN runoff was not released near an irreducible effluent concentration, however. TP concentrations, regardless of initial magnitude, noticeably approached an

“irreducible effluent concentration.” “Cleaner” influent TP concentrations increased, while “dirtier” TP concentrations decreased.

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CHAPTER 3: ASSESSING THE FEASIBILITY OF WQABI

Introduction

Stormwater runoff is a leading cause of water impairments in the United States (USEPA, 2009) and is particularly detrimental to regions economically dependent on fishing and beach tourism, such as the Mid-Atlantic region. Numerous fish kills, resulting from elevated nutrient concentrations, have been well-documented in the Chesapeake Bay (USEPA, 2008) and North Carolina's Albemarle and Pamlico Sounds (APNEP, 2009). Consequently, local and state-wide jurisdictions now incorporate stormwater runoff regulations. For example, the North Carolina Department of Environment and Natural Resources (NCDENR) utilizes a post-development standard of 0.45 kg/ha/yr (0.4 lb/ac/yr) and 4.5 kg/ha/yr (4.0 lb/ac/yr) for total phosphorus (TP) and total nitrogen (TN), respectively, in North Carolina's Tar-Pamlico River Basin (NCDENR, 2009a). To achieve regulatory standards, various Best Management Practices (BMPs) are employed to treat stormwater runoff before it enters natural water bodies.

BMPs are given credit based on percent nutrient removal for that particular BMP type. For instance, a stormwater wetland, in North Carolina, is credited with a 35% and 40% pollutant reduction of TP and TN, respectively. The percent removal metric that is commonly used to assess BMP performance is calculated using Equation 3.1:

$$\boldsymbol{Percent\ Removal} = \frac{\boldsymbol{inflow\ concentration - outflow\ concentration}}{\boldsymbol{inflow\ concentration}} \times \mathbf{100}\% \qquad \textbf{\textit{Eqn. (3.1.)}}$$

Increasingly, however, flaws of the percent removal methodology are being documented (Lenhart and Hunt, 2010; Schaafsma et al, 1998; Strecker et al., 2001; Jones et al., 2008). The percent removal metric does not account for BMP design variation, background water quality based on geographic location, background contaminant levels of receiving waters or land-use of surrounding watersheds. Furthermore, the percent or load removal metric promotes the false notion that high concentration and load percent removals directly correspond to "excellent" BMP performance. Such defects in the pollutant removal metric have led to a growing appeal for a new metric of assessing BMP function (Lenhart and Hunt, 2010; Shaafsma, 1998; Winer, 2000).

In 2008-2009, NCDENR developed the innovative Jordan Lake Nutrient Rules to manage excess nutrients, particularly TN and TP in Jordan Lake, a 303(d) listed, impaired water body in the Cape Fear River Basin of North Carolina (NCDENR, 2009b). The rules still utilize the percent reduction metric; however, nutrient reductions are only applied to existing developments and encourage the use of retrofits for nutrient reduction goals. Nutrient load requirements and nutrient discharge concentration standards are used as well. Additionally, the rules are adaptive and based on baseline or background water quality conditions (NCDENR, 2009b). Nutrient concentration standards, per varying background conditions, have not yet been established as part of the Jordan Lake Rules. McNett et al. (2010) propose an innovative metric of assessment based on effluent nutrient concentrations, known as the "water quality assessed by benthic macro-invertebrate (WQABI) metric". WQABI accounts for errors associated with the conventional BMP assessment methods by relating performance to target ambient water quality concentrations. The study paired data from 193 existing ambient water quality monitoring stations (AMS) with benthic macro-invertebrate rating (BMR) stations across 3 eco-regions in North Carolina. Each BMR rating (Excellent (E), Good (G), Good-Fair (GF), Fair (F), and Poor (P)) was correlated with an average pollutant constituent concentration (Table 3.1). Results indicated that dissimilar eco-regions may require distinct water quality standards. For instance, streams in the mountain eco-region of North Carolina had lower overall median nutrient concentrations than both Piedmont and Coastal streams; therefore, mountainous regions may require more stringent development regulations. However, the study determined that WQABI was best utilized in the Piedmont eco-region where a larger data pool existed and where the highest correlation between AMS and BMR data was observed (McNett et al., 2010).

Table 3. 1. Concentrations (mg/L) of pollutants associated with each water quality rating in Piedmont North Carolina.

PIEDMONT								
	Constituent concentration (mg/L)							
Rating	DO	TSS	NH3	NO2 + NO3	TKN	TN	TP	Fecal (#/100mL)
E	9.25	4.00	0.02	0.39	0.30	0.69	0.06	229.0
G	8.80	6.40	0.04	0.59	0.40	0.99	0.11	103.1
GF	8.40	5.00	0.06	0.67	0.50	1.17	0.13	93.0
F	7.70	7.00	0.06	1.60	0.56	2.16	0.22	169.2
P	6.80	5.00	0.13	6.34	1.25	7.59	0.63	316.4

Some researchers (Strecker et al. 2001, McNett et al. 2010) suggest that using an effluent concentration approach for BMP performance assessment is more accurate than the percent removal metric, in most cases, as evidenced in Chapter 2. Influent TN concentrations only slightly predicted effluent TN concentrations; however, influent TP concentrations were not able to predict influent TP concentrations with any significance. TN effluent concentrations may be predicted using a removal efficiency methodology; however, even in the very best general linear model created in Chapter 2, only 51 % of the variability in effluent concentrations was accounted for when using influent concentrations as predictors of effluent concentrations. Not a single "excellent" model was produced when using influent concentrations as predictors of effluent concentrations, thus caution should be exercised when using removal efficiencies as a sole water quality assessment metric.

The feasibility of an effluent assessment metric, such as WQABI, is unknown. Moreover, will current BMP performance allow for developed land to meet pre-developed, or target, water quality conditions, and, if so, will meeting these standards economically compromise development? Following are two case studies examining these questions. First, a compilation of Mid-Atlantic field data is used to determine which BMPs, if any, can meet effluent standards set by the WQABI metric established by McNett et al. (2010) and, second, a case study using TN and TP data, compares the WQABI method to the currently employed percent removal method to determine economic feasibility.

The Feasibility of Effluent Water Quality Guidelines

While effluent water quality guidelines have been successfully constructed for Piedmont North Carolina (McNett et al., 2010), the feasibility of such guidelines needs to be examined, based on actual BMP field data. Effluent total nitrogen (TN) and effluent total phosphorus (TP) concentrations were compiled from existing, published studies conducted on the nine most commonly installed BMPs, including: bioretention, dry detention, green roofs, level spreader-vegetated filter strips (LS-VFS), permeable pavement, sand-filters, vegetated swales, wetlands, and wet ponds. These data were then compared to standards set by WQABI for an overall performance assessment.

It is important to note that the ability to compile such data for this assessment is only feasible due to North Carolina's prior investment in research studies. This study is very unique in time and place, and many other states are not capable of such an assessment due to lacking research and correspondent data collection. Even if such an assessment was attempted 5 or more years ago in North Carolina, data would have been lacking. Overall, the increasing importance placed on research by North Carolina's local and state governments has made this analysis possible.

STUDY SELECTION. Studies were primarily selected based on proximity to North Carolina. Studies conducted at North Carolina State University (NCSU) were selected first, followed by any published and peer-reviewed studies conducted by other researchers in North Carolina. If data quantity was insufficient for a particular BMP, published, peer-reviewed studies from neighboring states were considered, including: South Carolina, Georgia, Virginia, and Tennessee. If data were still lacking for a particular BMP, published, peer-reviewed studies from familiar researchers in somewhat similar climates and geographies (to North Carolina) were used, including: Florida, Maryland, and Austin, Texas. In addition to proximity to North Carolina, selected studies were required to have TN and/or TP concentration data reported with $n > 8$. Sources for all data used are provided in Table 3.2, by BMP type. The various pollutant removal mechanism and configuration of each BMP type is briefly described in Hunt et al. (2009). Once assembled, effluent TN and TP concentrations were compiled by BMP type and averaged then compared to WQABI standards.

Table 3.2. Sources and locations of all sites included in this study.

BMP	Reference (Site name)	Location
Bioretention	Passeport et al., 2009b (Graham N)	Graham, NC
Bioretention	Passeport et al., 2009b (Graham S)	Graham, NC
Bioretention	Hunt et al., 2008 (HMBC)	Charlotte, NC
Bioretention	Line and Hunt, 2009 (I-40)	Catawba County, NC
Bioretention	Li et al., 2009; Sharkey, 2006 (L1)	Louisburg, NC
Bioretention	Li et al., 2009; Sharkey, 2006 (L2)	Louisburg, NC
Bioretention	NCSU Field Data (RM Grass)	Rocky Mount, NC
Bioretention	Hunt et al., 2006 (C1)	Chapill Hill, NC
Dry Detention	Hathaway et al., 2007 (Morehead)	Charlotte, NC
Dry Detention	Hathaway et al., 2007 (University Exec)	Charlotte, NC
Dry Detention	IBMPDB, 2009 (Greenville Pond)	Greenville, NC
Dry Detention	IBMPDB, 2009 (Mountain Park)	Lilburn, GA
Dry Detention	IBMPDB, 2009 (Twin Towers)	Tallahassee, FL
Green Roof	Hathaway et al., 2008	Goldsboro, NC. & Kinston, NC
LS-VFS	Line and Hunt, 2009	Johnston County, NC
LS-VFS	Winston, 2009 (1a)	Apex, NC
LS-VFS	Winston, 2009 (1b)	Apex, NC
LS-VFS	Winston, 2009 (2a)	Louisburg, NC
LS-VFS	Winston, 2009 (2b)	Louisburg, NC
LS-VFS	Yu et al., 1993 (FS 1)	Virginia
LS-VFS	Yu et al., 1993 (FS 2)	Virginia
LS-VFS	Hathaway and Hunt, 2008	Charlotte, NC
Permeable Pavement	Bean et al., 2007 (Swansboro)	Swansboro, NC
Permeable Pavement	Bean et al., 2007 (Goldsboro)	Goldsboro, NC
Permeable Pavement	Collins et al., 2010 (PC)	Kinston, NC
Permeable Pavement	Collins et al., 2010 (PICP1)	Kinston, NC
Permeable Pavement	Collins et al., 2010 (CGP)	Kinston, NC
Permeable Pavement	Collins et al., 2010 (PICP2)	Kinston, NC
Vegetated Swale	IBMPD, 2009 (Walnut Creek)	Austin, TX
Vegetated Swale	IBMPD, 2009 (US-183)	Austin, TX
Vegetated Swale	NCSU Field Data, 2009 (NV-A)	Johnston County, NC
Vegetated Swale	NCSU Field Data, 2009 (NV-B)	Johnston County, NC
Vegetated Swale	Stagge, 2006 (SHA)	Savage, MD
Vegetated Swale	Stagge, 2006 (MDE)	Savage, MD
Wet Pond	Hathaway et al., 2007 (Shade Valley)	Charlotte, NC
Wet Pond	Hathaway et al., 2007 (Pierson)	Charlotte, NC
Wet Pond	IBMPDB, 2009 (Shawnee Ridge)	Suwanee, GA
Wet Pond	IBMPDB, 2009 (Lake Munson)	Tallahassee, FL
Wet Pond	Mallin et al., 2002 (Ann McCrary)	New Hanover County, NC
Wet Pond	Mallin et al., 2002 (Silver Stream)	New Hanover County, NC
Wet Pond	Mallin et al., 2002 (Echo Farms)	New Hanover County, NC
Wetland	Bass, 2000 (Edenton)	Edenton, NC
Wetland	Burchell and Hunt (Laney HS)	Wilmington, NC
Wetland	Carleton et al., 2001; Carleton, 1997 (Crestwood)	Manassas, VA
Wetland	Hathaway and Hunt, 2010 (Dye Branch - cell 3)	Mooresville, NC
Wetland	Hathaway et al., 2007 (Edwards Branch)	Charlotte, NC
Wetland	Hathaway et al., 2007; Johnson, 2006 (BES)	Charlotte, NC
Wetland	Johnson, 2006 (SSHS)	Charlotte, NC
Wetland	Lenhart and Hunt, 2010 (River Bend)	Riverbend, NC
Wetland	Line et al., 2008 (CMS)	35′ 45.4″ N; 78′ 41.2″ W
Wetland	Line et al., 2008 (UNC)	Asheville, NC
Wetland	Maristany and Bartel, 1989 (Lake Munson)	Tallahasee, FL
Wetland	NCSU Field Data, 2009 (VA)	Johnston County, NC
Wetland	NCSU Field Data, 2009 (VB)	Sampson, NC
Wetland	Yu et al., 1998; IBMPDB, 2009 (Covington)	Rappahannock County, VA
Wetland	Yu et al., 1998; IBMPDB, 2009 (Rt. 288)	Chesterfield, VA

Results

The most documented BMP types were bioretention cells, LS-VFS, wetlands, and wetponds. There were no peer-reviewed, published studies found on sand-filters. The performance of each BMP type varied widely: effluent TN concentrations ranged from 0.88 mg/L to 4.34 mg/L, while effluent TP concentrations ranged from 0.05 mg/L to 1.01 mg/L (Table 3.3). Overall, stormwater wetponds performed best with respect to TN, and permeable pavement performed best with respect to TP. The highest effluent concentrations for both TN and TP were observed in green roofs; however, data from only one local green roof study could be found. Likewise, the lowest TP concentration was observed in permeable pavement; however, only one study was found on permeable pavement. Thus, more research is needed before any concrete assumptions are made on the performance of these particular BMPs.

Table 3.3. Median Effluent TN and TP concentrations (mg/L) by BMP type

BMP Type	**n*** (TN)	Median TN out (mg/L)	**n*** (TP)	Median TP out (mg/L)
Bioretention	*6*	0.95	*6*	0.11
Dry Detention	*4*	1.15	*5*	0.19
Green Roof	*1*	4.34	*1*	1.01
Level Spreader-VFS	*5*	1.06	*5*	0.16
Permeable Pavement	*1*	0.99	*2*	0.05
Vegetated Swale	*4*	3.25	*6*	0.22
Wetland	*12*	0.97	*11*	0.10
Wet Pond	*7*	0.88	*7*	0.10

*** "n" indicates the number of sites analyzed in Chapter 3. "n" is NOT the number of samples taken per study. Each study had n>8 water quality samples for both TN and TP.**

Of all BMPs examined in this study, none met “E” WQABI standards for TN (= 0.69 mg/L), 4 BMPs (bioretention, permeable pavement, wetlands and wet ponds) met “G” WQABI standards for TN (= 0.99 mg/L), and 6 BMPs (bioretention, dry detention basins, LS-VFS, permeable pavement, wetlands and wet ponds) met “GF” WQABI standards for TN (= 1.17 mg/L). Only green roofs and grassed swales were unable to achieve “F” TN water quality standards (= 2.16 mg/L) (Table 3.4).

Table 3.4. Achievable total nitrogen (TN) WQABI Water Quality Standards by BMP type

		Achievable WQABI Rating in the Piedmont Eco-region				
BMP Type	Median TN out (mg/L)	Excellent (0.69 mg/L)	Good (0.99 mg/L)	Good Fair (1.17 mg/L)	Fair (2.16 mg/L)	Poor (7.59 mg/L)
Bioretention	0.95	-	X	X	X	X
Dry Detention	1.15	-	-	X	X	X
Green Roof *	4.34	-	-	-	-	X
Level Spreader- VFS	1.06	-	-	X	X	X
Permeable* Pavement	0.99	-	X	X	X	X
Vegetated Swale	3.25	-	-	-	-	X
Wetland	0.97	-	X	X	X	X
Wet Pond	0.88	-	X	X	X	X

* only one study was used for greenroof and permeable pavement TN data

Of all BMPs selected in this study, only permeable pavement met “E” WQABI standards for TP (= 0.06 mg/L) and 4 BMPs (bioretention, permeable pavement, wetlands and wet ponds) met “G” and “GF” WQABI standards for TP (= 0.11 mg/L and = 0.13 mg/L, respectively;

Table 5). Only green roofs were unable to achieve “F” and “P” water quality standards for TP (= 0.22 mg/L and = 0.63 mg/L, respectively; Table3.5).

Table 3.5. Achievable total phosphorus (TP) WQABI Water Quality Standards by BMP type

		Achievable WQABI Rating in the Piedmont Eco-region				
BMP Type	Median TP out (mg/L)	Excellent (0.06 mg/L)	Good (0.11 mg/L)	Good Fair (0.13 mg/L)	Fair (0.22 mg/L)	Poor (0.63 mg/L)
Bioretention	0.11	-	X	X	X	X
Dry Detention	0.19	-	-	-	X	X
Green Roof*	1.01	-	-	-	-	-
Level Spreader- VFS	0.16	-	-	-	X	X
Permeable Pavement	0.05	X	X	X	X	X
Vegetated Swale	0.22	-	-	-	X	X
Wet Pond	0.10	-	X	X	X	X
Wetland	0.10	-	X	X	X	X

* only one study was used for greenroof data

Discussion

Similar to a study by Barrett et al. (2009), BMPs with the highest runoff resident times appeared to be the best performers. Even so, no difference exists in the overall effluent water quality among these 4 BMPs, according to WQABI standards (all 4 are capable of producing “Good” water quality). Both studies showed that wet ponds perform superior to filters (sand filters and bioretention), and filters superior to grassed swales (Table 3.6). Furthermore, in both studies, grassed swales produced higher effluent nutrient concentrations than respective influent concentrations. Overall most of the BMPs in this study tended to produce lower nutrient concentrations than respective BMPs examined in Barrett et al. (2009). This is likely because the study by Barrett et al. (2009) included sites with outdated design configurations, or designs that would no longer be recommended. Furthermore, the methods of quality control for sites included in the Barrett et al. (2009) study are unknown, whereas all of the QA/QC for sites in this study (Chapter 3) is publically accessible. Unlike this study, Barrett et al. (2009) did not use data inclusive to one geographical location. Further, it is important to note in Table 3.6 that “filters” in Barrett et al. (2009) were just “sand filters” and in this study “filters” includes “bioretention.” The vegetation and engineered media used in bioretention cells likely cause bioretention to be a better performer than sand filters.

Table 3.6. Mean effluent TN and TP concentrations from 4 BMP types documented in this study and in Barrett et al. (2009)

	NC-Centric	Barrett et al. (2009)*	NC-Centric	Barrett et al. (2009)*
	Mean TN out (mg/L)	Mean TN out (mg/L)	Mean TP out (mg/L)	Mean TP out (mg/L)
wetponds	1.07	1.50	0.11	0.20
filters	0.92	2.25	0.14	0.25
detention	1.14	2.50	0.17	0.30
swales	3.81	2.50	0.26	0.35

* Values are estimated off of graphs provided in Barrett et al. (2009)

Four BMPs in this study (bioretention, permeable pavement, wetponds, and wetlands) very noticeably produced lower TN and TP effluent concentrations than the other BMPs examined. Using this evaluation tool, these BMPs would be recommended in pristine areas, such as North Carolina’s Coastal and Mountain eco-regions, or other sensitive areas where

TN and TP are of large concern; however, in areas where space or budget restraints apply, dry detention or level spreader – vegetated filter strips may be used in lieu. Obviously dry detention and LS-VFSs produce “dirtier” effluent than other BMPs examined (Table 3.3); however, even these BMPs may improve water quality in urban areas or from land uses such as greyfields and brownfields. Based on elevated effluent nutrient concentrations, vegetated swales and green roofs are not recommended for water quality improvements from a concentration perspective. Again, please note that more data gain is necessary before any concrete assertions be made on the performance of green roofs and permeable pavement. Finally, this evaluation tool does not account for volume reduction; thus any recommendations must be tempered with that in mind. Overall, the desired water quality rating of an area, in combination with a BMP’s expected effluent concentrations, should be an important factor in BMP selection; however, site constraints such as retrofit restrictions and/or community resistance may inhibit the use of one BMP over another, and an alternate BMP may be chosen accordingly. For instance, a wetland may not be feasible in an urban setting, yet may be ideal for a rural setting and vice versa for permeable pavement.

WQABI versus Percent Removal: a Theoretical Case Study

A probable flaw in the percent removal methodology for meeting regulatory standards is the required use of BMPs in series when unnecessary (Hathaway and Hunt, 2010; Van Der Wiele, 2007). The WQABI method has potential to prevent excessive BMP installations, while maintaining healthy rivers and streams, as shown in the following case study.

The quantity and types of BMPs needed to treat a typical 0.4 hectare (1 acre) parking lot in the piedmont eco-region of North Carolina were considered. Both percent removal and effluent event mean concentration (EMC) from published research were used. First, using the percent removal methodology the quantity of BMPs needed to meet current mass load effluent standards, 0.4 lb/ac/yr for TP and 4.0 lb/acre/yr for TN for the Tar-Pamlico Basin (NCDENR, 2009a) were determined. Then, the median TN and TP effluent concentrations required to achieve E to G water quality in the piedmont region, 0.06 to 0.11 mg/L for TP and 0.69 to 0.99 mg/L for TN (Table 3.1), were used to select the type and quantity of BMPs.

Pollutant loads for small urban watersheds are commonly calculated using the Simple Method (Schueler, 1987), despite documented shortcomings. For example, the Simple Method only accounts for pollutants found in storm-flow, neglecting baseflow-borne pollution; however, the probability of baseflow containing a non-negligible amount of pollutants is un-likely (NYDEC, 2008). Other issues with the Simple Method are that it does not account for development age nor does it account for watershed characteristics such as landuse. Despite such concerns many East Coast States (Georgia, New Jersey, New York, North Carolina, Pennsylvania, and Virginia) all use the Simple Method to predict stormwater pollutant loads (Atlanta Regional Commission, 2001; NJDEP, 2004; NYDEC, 2008; NCDENR, 2007; PADEP, 2006; VDCR, 1999).

Using the Simple Method, Eqn. 2 (Schueler, 1987), nutrient mass loads were calculated using typical parking lot runoff event mean concentrations, as measured in North

Carolina, of 0.19 mg/L and 1.57 mg/L for TP and TN, respectively (Passeport and Hunt, 2009). Explanations for each model parameter are listed in Table 3.7.

$$L = P * Pi * Rv * C * 0.227 \qquad \textit{Eqn. (3.2.)}$$

Table 3. 7. The Variable used in the Simple Method (Schueler, 1987)

Variable	Meaning and Units
L	nutrient load (lb/ac/yr)
P	average annual rainfall (45 inch/yr in the Piedmont region of North Carolina)
Pi	correction factor for storms with no runoff, (0.9)
Rv	runoff coefficient, 0.05 + 0.9*I', where I' = fraction imperviousness from 0 to 1
C	flow weighted EMC (mg/L), 0.19 for TP and 1.57 for TN (Passeport and Hunt, 2009)
0.227	unit conversion factor (Passeport, 2007)

A TP load of 1.86 kg/ha/yr (1.66 lb/ac/yr) was calculated, using the following values: P = 1.14 m (45 in/yr), Pi = 0.9, Rv = 0.14, and C = 0.19 (Table 3.7). Similarly, a mass load of 15.4 kg/ha/yr (13.7 lb/ac/yr) was calculated for TN. A typical bioretention cell in North Carolina, with 45% and 35% removal rates for TP and TN, respectively (NCDENR, 2009a), would reduce the runoff loads for TP and TN to 1.02 kg/ha/yr (0.91 lb/ac/yr) and 10.0 kg/ha/yr (8.9 lb/ac/yr), respectively, per this calculation. The regulatory standards of 4.5 kg/ha/yr (4.0 lb/ac/yr) for TN and 0.45 kg/ha/yr (0.4 lb/ac/yr) for TP would not be met. In order to meet TP and TN regulatory load standards, <u>three</u> bioretention cells in series would be necessary per the percent removal methodology. Other BMPs like stormwater wetland, with removal rates of 40% for both TP and TN, respectively (NCDENR, 2009), would also require at least 3 BMPs in series to comply with state TP and TN standards.

Using the WQABI method, Excellent (E) to Good (G) rated water quality concentrations range from 0.06 to 0.11 mg/L for TP and 0.69 to 0.99 mg/L for TN (Table 3.1). Based on the median TP and TN bioretention effluent values mentioned previously of 0.11 mg/L and 0.95 mg/L, respectively (Tables 3.3 and 3.4), G quality ratings are met for both TP and TN using *one* bioretention cell. Stormwater wetlands with median effluent values of 0.10 mg/L and 0.97 mg/L, for TP and TN respectively (Tables 3.3 and 3.4), also meet a G rating, with respect to TN and TP, using *one* practice. The WQABI method shows that in some instances one BMP cell, in lieu of two or three in series, as required by the percent removal method, is sufficient to provide desired water quality from an effluent concentration perspective; in other cases WQABI may eliminate certain types of practices from consideration altogether. Decreasing the need for BMPs in series would substantially lower design, construction, and maintenance costs. Although acceptable water qualities are attainable using the WQABI metric of BMP assessment, it is very important to also consider hydrologic factors. The WQABI metric must be used in combination with a volumetric runoff reduction component.

Conclusions

1. Overall, stormwater BMPs produce a variety of effluent nutrient concentrations. Four BMPs tended to perform better than the others examined, including bioretention, permeable pavement, wetlands, and wet ponds. The overall performance of permeable pavement in this examination should be viewed with caution as only 2 permeable pavement studies were acquired and used. According to WQABI standards

for TN and TP, these 4 BMPs all produce “G” water quality. Two BMPs, vegetated swales and green roofs, tended to perform worse than the other BMPs examined. Caution should be used in considering the performance of green roofs, as only one study was available for this examination. Vegetated swales were unable to produce TP concentrations better than a Fair (F) standard and were also unable to produce TN concentrations above a Poor (P) standard.

2. Some water quality targets may simply not be possible based on current BMP designs. For instance, not one BMP in this study met the “E” WQABI standard for TN. However, “G” WQABI standards for both TN and TP can be met by 4 different BMPs, and further design experimentation and innovation may result in higher water quality ratings as well.

3. Designers selecting BMPs for watersheds where nutrients are of primary concern should consider the use of bioretention, permeable pavement, wetlands, or wetponds. In areas where space or budget restraints apply, dry detention or level spreaders may be used in lieu. Obviously dry detention and LS-VFSs produce “dirtier” effluent than other BMPs examined (Table 3.3); however, even these BMPs may improve water quality to Good-Fair (GF) in urban areas or areas such as greyfields and brownfields that typically have F or P water qualities.

4. The WQABI metric may achieve the same, if not better, water quality as does the percent removal metric, and is more economically feasible. A case study for a

Piedmont NC parking lot showed that using the percent removal metric to meet state specified water quality guidelines, three bioretention cells in series are required; however, using the WQABI metric, only one cell is needed to achieve a similar water quality rating, when based solely on concentration effluent.

5. Hydrologic considerations must be used in combination with the findings of this study. The WQABI method does not account for volumetric reductions achievable by BMPs. Combining the concepts of effluent concentrations and runoff volume reduction may be a goal of future research.

6. The WQABI metric is restrictive because it only accounts for local conditions. For example, a development with BMPs in the NC Piedmont eco-region may satisfy local requirements, but still impact downstream waterbodies in the Coastal eco-region, which have more stringent standards. Thus, whole watersheds should be considered before establishing post-development regulatory standards.

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CHAPTER 4: AN EVALUATION OF THE TOXICITY OF ACCUMULATED SEDIMENTS IN FOREBAYS OF STORMWATER WETLANDS AND WETPONDS

Introduction

When properly designed, constructed and maintained, stormwater best management practices (BMPs) efficiently capture sediment from runoff and decrease peak and total flow (Hsieh and Davis, 2005; Hunt et al., 2006; Li and Davis, 2008; OMOE, 2003; Urbonas et al., 1995). Nutrients and metals in runoff sorb to soils, and accumulate in BMPs such as stormwater wetponds and wetlands, instead of flowing directly into waterbodies (Burton, 1992; Councell et al., 2004; Crowe et al., 2007; Hathaway and Hunt, 2010; Lau and Stenstrom, 2005; Lee et al., 2002; Obarska-Pempkowiak and Klimkowska, 1999). The efficiency of sediment removal is related to the available storage volume of a BMP, particularly in constructed wetponds and stormwater wetlands (Graham and Lei, 2000; Obarska-Pempkowiak and Klimkowska, 1999). While increased sediment accumulation implies that a BMP is performing as intended, as accumulation occurs, the available storage volume decreases, decreasing sediment removal and increasing the risk of flooding/pollutant transfer downstream (Heal et al., 2006; Marsalek and Marsalek, 1997; OMOE, 2003). This relationship necessitates the removal of accumulated sediments on a regular basis, dependent on individual BMP parameters affecting sediment accumulation rate, such as: landuse (including construction activity), watershed area, and the presence/absence of upstream BMPs (Liebens, 2001). The forebay (Figure 4.1), in most cases, is the intended portion of a wetland or wetpond to be excavated for captured sediment and solids removal (Johnson, 2007; Scarborough and Mensinger, 2005; State of Minnesota Stormwater Advisory, 1997). It is important to differentiate the toxicity of forebay sediments, where sediment and gross solid excavation and disposal is anticipated, from the remainder of the pond/wetland sediments, which are likely to remain "untouched". No peer-reviewed and published studies are available with regard to *forebay* sediment metal concentrations specifically; rather, entire wetponds or wetlands have been analyzed.

Figure 4.1. A wetpond with forebay circled.

This study's primary objective is to quantify metal concentrations in accumulated sediments found in stormwater wetland and wetpond forebays in North Carolina. Based on existing toxicity standards for aquatic health and for the land application of biosolids, excavation needs and disposal recommendations will be made.

AQUATIC HEALTH SEDIMENT STANDARDS. Of several methods used to establish sediment guidelines for aquatic health, the most commonly accepted is the effects range approach. This approach uses field data to determine pollutant concentrations in which adverse effects are observed or not observed in aquatic biota. The Australian and New Zealand Environmental and Conservation Council (ANZECC, 2000) and the U.S. National Oceanic and Atmospheric Administration (NOAA, 1999) have both used the effects range approach to establish sediment quality guidelines. Utilizing this approach, the Florida Department of Environmental Protection (FDEP) conducted a comprehensive study, compiling data from across the United States to develop a range of effects including the Threshold Effects Level (TEL) and the Probable Effects Limit (PEL). The TEL is the highest concentration in a range of pollutant concentrations in which no adverse effects are observed, whereas the PEL is the lowest concentration in a range of pollutant concentrations in which adverse effects usually, or do, occur. ANZECC asserts that the FDEP effects range "is one of the most comprehensive documentations of sediment quality assessment" (ANZECC, 2000); therefore, the FDEP guidelines are used to assess biotic health in this study. If sediment toxicity exceeds aquatic health standards, even more reason exists to excavate forebay sediments.

SEDIMENT LAND-APPLICATION STANDARDS. The most desirable and economical disposal measure for forebay sediments would be to: (1) spread sediment on adjacent land upon excavation and (2) seed with a cost-effective grass or ground cover to stabilize the applied soil. If sediments contain pollutant levels capable of causing human or ecological harm, land application is not acceptable; however, excavated spoils may still be used in roadway construction or as a landfill cover (termed "beneficial use"). If sediment toxicity is determined to be too high for both land-application and for beneficial use, sediments may require landfilling which would be the most costly disposal option.

Currently, U.S. hazardous waste disposal legislation (40 CFR 239 to 40 CFR 259) does not include explicit regulations for BMP accumulated sediments. However, several states within the U.S., including Georgia and South Dakota (Polta, 2004; Sonon and Gaskin, 2009), are beginning to adopt Title 40, Part 503, Section 13 of the Federal Code of Regulations (40 CFR 503) to regulate stormwater BMP sediment disposal. 40 CFR 503 is intended for the land application of biosolids and sewage sludge; however, it is serving as initial guidance for disposal of accumulated forebay sediment.

Site Selection and Sampling Protocols

Sediment from 30 across North Carolina was collected for examination. Many property owners requested that specific site locations remain anonymous. Twenty-one sites had a commercial landuse and 9 had a residential landuse. All institutional sites, such as those located on the NC State University campus, and industrial sites, such as gas stations, were included in the commercial landuse category. Percent imperviousness of the watershed, approximate drainage area, BMP age, and forebay surface area were recorded for each forebay. Drainage areas ranged from 0.17 hectares to 43.78 hectares, while surface areas ranged from 27.87 m2 to 1335.48 m2. System age, or years since last forebay maintenance period, ranged from 1 to 10 years and the percent imperviousness of watersheds ranged from 20 percent to 90 percent (Table 4.1).

Table 4.1. Characteristics of each site sampled, including: location, landuse, age, drainage area (DA), and surface area of the forebay (SA).

Site	Municipality	Landuse	Age	DA	SA	% Imp
			yrs.	hectares	m^2	
A	APEX	COM	4	0.38	193.52	73
B	APEX	COM	4	1.14	192.96	73
C*	DURHAM	COM	3	0.17	111.86	65
D*	DURHAM	COM	3	1.15	79.99	90
E	DURHAM	COM	5	4.74	76.18	25
F	DURHAM	COM	7	3.90	297.29	29
G	DURHAM	COM	2	0.67	52.95	67
H	DURHAM	COM	3	2.12	152.18	70
I	HIGH POINT	COM	10	1.33	27.87	40
J	RALEIGH	COM	4	2.78	217.21	52
K	RALEIGH	COM	3	6.93	388.71	50
L	RALEIGH	COM	6	0.95	43.66	48
M	RALEIGH	COM	6	0.47	51.28	86
N	RALEIGH	COM	6	1.61	67.82	58
O	RALEIGH	COM	3	4.30	193.98	52
P	RALEIGH	COM	8	6.93	161.65	70
Q	CHARLOTTE	COM	5	43.78	236.90	35
R *	DURHAM	COM	1	1.00	85.47	73
S *	DURHAM	COM	3	7.40	408.77	69
T *	DURHAM	COM	3	19.70	966.19	85
U*	RALEIGH	COM	4	3.79	40.88	37
V *	DURHAM	RES	5	3.28	404.69	52
W	DURHAM	RES	8	6.02	202.53	64
X	DURHAM	RES	8	2.59	323.30	65
Y	DURHAM	RES	6	13.09	445.94	39
Z	CHARLOTTE	RES	6	5.76	219.25	60
AB*	DURHAM	RES	8	32.47	166.02	20
CD *	DURHAM	RES	5	20.65	1335.48	47
EF *	DURHAM	RES	1	1.69	66.89	27
GH	DURHAM	RES	6	4.74	283.35	32

* Sites that had additional samples taken near outlet structure for spatial variation analysis.

All forebays were sampled for 7 metals commonly found in stormwater: copper (Cu), zinc (Zn), iron (Fe), nickel (Ni), cadmium (Cd), chromium (Cr), and lead (Pb) (Graham, 2000; Liebens, 2002; USEPA, 1983). Total metal concentrations were analyzed by North Carolina State University's Analytical Services Laboratory in the Soil Science Department using EPA

Method 3050B (EPA, 1996). Analytical testing procedures and instruments of analysis are listed in Table 4.2.

Table 4.2. Analytical Testing Procedures and Instruments of Analysis

Metal	Form	Analytical Testing Procedure	Analysis Instrumentation
Cr, Cu, Fe, and Zn	Total	EPA Method 3050B, strong acid digestion	ICP-OES
Pb, Cd, and Ni	Total	EPA Method 3050B, strong acid digestion	ICP-MS

ICP-OES = Inductively Coupled Plasma Optical Emission Spectroscopy
ICP- MS= Inductively Coupled Mass Spectroscopy

SAMPLING. All forebays were divided into a 9-segment grid and a sample was taken within each quadrat (Appendix G). An equal mass of each sample (ranging from 70 to 100 g., by site) was composited. For ten of 30 sites three additional samples were taken near the outlet structure of the pond or wetland. Again, equal masses were composited. Outlet samples were taken to investigate whether sediment toxicity was higher with expected finer sediment located at the BMP outlet. The particle size of each composite sample was analyzed using the Hydrometer Method (Gee and Bauder, 1986, Appendix K).

In most cases, the forebay samples could be taken on foot using a steel auger or large plastic spoon, depending upon underlying substrate; however in some instances, accumulated muck, or considerable forebay water depths prohibited sampling on foot and required a small inflatable raft (Figure 4.2). All outlet samples were collected using the raft. Some forebays contained substantial quantities of rip-rap or dense vegetation, so the exact sampling location varied depending on the specific segment. Regardless of the variations in sampling locations, each quadrat was sampled. Three to four samples were placed at opposite corners in rectangular floating bins during sampling to prevent samples from mixing (Figure 4.2).

a. b. c.

Figure 4.2. Photos of sediment collection procedure (a) steel augur used in sediment sampling (b) floatable bins used in collection (c) inflatable raft used for sampling when forebay was not wadeable.

Samples were then carried to a portable scale, which held a clean plastic container to retain the sediments while being weighed. The scale was initially zeroed (before sediments were added to the clean plastic container) and again between consecutive measurements. Between composite samples, all equipment was rinsed with deionized water. Samples were immediately sealed, and returned to the laboratory for analysis. Each sample bottle was assigned a code based on site location.

Results and Interpretation

For comparison to aquatic health guidelines and 40 CFR 503 limits, all concentrations were reported in milligrams of pollutant per kilogram of soil in dry weight. Mean and median values were calculated overall, as well as mean and median pollutant concentrations for each landuse (Table 4.3).

Table 4.3. Median and Mean Pollutant Concentrations Measured in 30 forebays across North Carolina

		Pollutant Concentration (mg/kg, dry weight)						
Landuse		* Cd	Cr	Cu	Fe	Ni	Pb	Zn
Commercial	Mean	ND	20.9	28.0	24700	14.4	13.7	140.0
(n=21)	Median	ND	20.0	23.0	22100	13.0	13.0	75.0
Residential	Mean	ND	23.8	18.6	22400	16.0	12.3	58.4
(n=9)	Median	ND	28.3	15.1	23300	15.9	11.4	44.0

* Cd was not detected in 28 of 30 cases. In 2 of 30 cases, Cd was reported as < limit of quantitation (LOQ), or [Cd] < 5.0 mg/kg dry weight.

The median values sampled follow some trends exhibited by the extensive National Urban Runoff Program (NURP) study (USEPA, 1983). For example, commercial landuse areas, as compared to areas of residential landuse, exhibited higher pollutant concentrations of Zn. Unlike the NURP study, commercial areas exhibited higher median concentrations of Pb and Cu when compared to residential areas. This difference may be due to increased vehicular use since 1983 (the year the NURP study was conducted), particularly in commercial areas. Vehicular use, particularly brake and tire wear are associated with Pb, Cu, Cd, and Zn in stormwater runoff (Davis et al., 2001). In this study, median Fe, Cr and Ni concentrations were higher, sometimes slightly, in residential landuses, when compared to commercial landuses. Median Cd concentrations were below detection limits in nearly all cases.

Sample pollutant concentrations were also compared to other published studies to determine the relative toxicity of forebays to overall pond/wetland concentrations. Median metal concentrations measured in North Carolina forebays appear similar, but somewhat lower, to respective pollutant medians measured in overall stormwater wetponds and wetlands and roadside swales. The fact that the NC forebay concentrations are somewhat lower than those associated with entire ponds is not unexpected, when considering probable differences in particle size distribution (discussed in the next section). In all cases, median forebay sediment concentrations were higher than mean soil metal concentrations in native North Carolina soils (Table 4.4). Additional graphs are included in Appendix H.

Table 4.4. Levels of Metal in sediments from various sources (mg/kg dry weight, unless otherwise noted)

Reference - Location	Cd	Cr	Cu	Fe	Ni	Pb	Zn
Sediment type (Landuse)	All pollutants reported in mg/kg, dry weight.						
Heal et al., 2006 - Scotland							
Halbeath Pond (mixed)	0.21	70.7	18.8	4.41 (%)	63.3	26.3	78.4
Linburn Pond (mixed)	0.22	78.2	20.9	4.74 (%)	68.4	25.4	110
Pond 7 (mixed)	0.32	118	16.3	3.87 (%)	83.9	18.2	77
Wetland (mixed)	0.39	76.7	17.4	7.16 (%)	63.6	22.6	93.1
Liebens, 2001 - FL, United States							
Pond (Commercial), n=8	1.15	22.48	27.41	-	13.42	142.31	253.85
Pond (Residential), n=16	0.38	14.27	11.04	-	7.87	15.68	39.73
Roadside Swale (Commercial)	1.69	24.53	27.1	-	12.46	121.1	268.39
Roadside Swale (Agricultural)	0.64	12.08	9.21	-	10.59	48.46	38.29
VanLoon et al., 2006- Ottawa, Canada							
Wetpond, WP1	0.46	41.5	27.78	27500	24.9	19.76	127.1
Wetpond, WP2	0.53	31.3	22.03	18300	15.3	18.87	94.8
Hardy et al., 2008 - NC, United States							
Mean North Carolina Soils	0.1	0.2	9.2	-	0.8	4.2	27.2
Forebay Study, 2009 - NC, United States							
Median (Commercial)	ND	20	23	22100	13	13	75
Median (Residential)	ND	28.3	15.1	23300	15.85	11.4	44

ANALYSIS OF SPATIAL VARIABILITY. Metal concentrations, by soil type, are shown to be highest among clayey soils (Liebens, 2001). Thus, forebays, closer to the inlet than the remainder of the wetland or wetpond, should theoretically contain proportionally more coarse sand particles than finer silt and clay particles. Therefore, they are expected to exhibit lower metal concentrations than the rest of the wetpond or wetland.

As expected, of 10 sites analyzed for particle size distribution, nearly all forebay sediment samples analyzed, in both commercial and residential landuses, exhibited a higher fraction of sand when compared to samples of sediment that had accumulated near the outlet structure (Figure 4.3, Appendix I). All data were normal (Cramer von-Mises, $p>0.05$); thus, a t-test was used for statistical analysis. The fraction of sand in forebay sediment samples was significantly higher than the fraction of sand found in accumulated sediment samples taken near the outlet structures ($p=0.04$). Age appeared to have little impact on particle size fractionation (Figure 4.3, Appendix J).

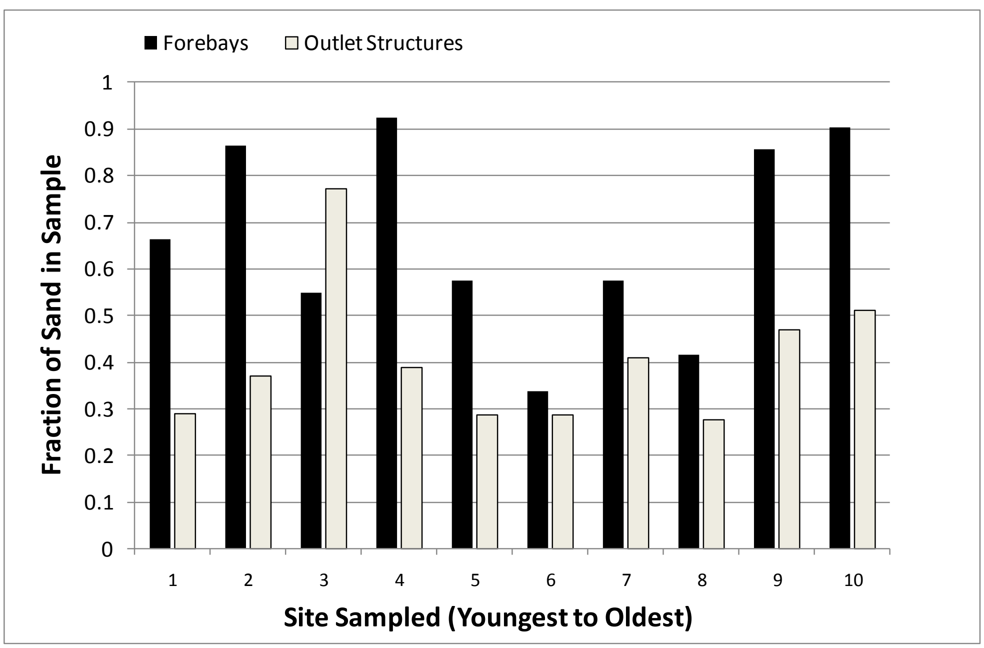

Figure 4.3. Fraction of Sand in forebay sediments (solid-markers) and sediments near the outlet structure (un-filled markers) at 10 ponds/ wetlands.

Furthermore, raw data show that metal concentrations measured at the outlet structure in wetponds and wetlands are somewhat greater than metal concentrations found in the forebay, or inlet of the pond/wetland (Table 4.5). This hypothesis was statistically analyzed using t-tests on normalized data and non-parametric statistics for non-normal data. Data for Cu, Zn, and Pb were normal ($p < 0.05$) in raw form, and Fe data was normal upon log transformation, using Cramer-von Mises test for normality. Cr and Ni were non-normal in raw and log-transformed states ($p > 0.05$), thus these metals were analyzed using non-parametric statistics. Contrasting initial supposition, t-tests for Cu, Fe, Pb, and Zn showed no significant differences ($p > 0.05$) between forebay metal concentrations and outlet metal concentrations. Also, the tau-associations for Cr and Ni, 0.33 and 0.13, respectively, were relatively low, indicating there was minimal significant difference between forebay metal concentrations and outlet metal concentrations. Perhaps the lack of statistical evidence is partially a result of a limited number of samples (10) tested. A more detailed description of the statistical analysis used is provided in Appendix I.

Table 4.5. Comparison of metal concentrations at the inlet/forebay and outlet structures of ten stormwater wetlands/wetponds in North Carolina, where IN = sediments in the forebay and OUT = sediments near the outlet structure.

	Metal Concentrations, mg/L						
	Conc. Cr	Conc. Cu	Conc. Fe	Conc. Zn	Conc. Cd	Conc. Ni	Conc. Pb
Average IN	21.4	18.1	19462	93.9	ND	16.9	11.7
Average OUT	27.4	25.2	24950	151.0	ND	18.5	16.9
Difference (OUT-IN)	**6**	**7.1**	**5488**	**57.1**	**ND**	**1.6**	**5.2**

STATISTICAL ANALYSIS ON LANDUSE. Raw data suggested that residential landuses would have (1) higher Ni and Cr concentrations, (2) slightly higher Fe concentrations, and (3) lower Cu, Pb and Zn concentrations than commercial landuses. To determine the statistical significance of raw data observations, the effect of landuse on pollutant concentrations was tested using SAS statistical software package (2004). Histograms were first constructed to visually inspect the distribution of data about respective means. Data exhibited normal distributions in the majority of the pollutants assessed; however one non-normal dataset (Zn) required the use of non-parametric methods to analyze data for landuse effects. The Kendall tau test of association for non-parametric statistics showed that Zn concentrations were not significantly different between residential and commercial landuses ($\tau = -0.06$). Likewise, all other pollutants displayed highly insignificant differences in concentrations between residential and commercial landuses.

ASSESSING TOXICITY. The toxicity of all forebay samples was assessed using aquatic health and land-application benchmarks. The standard for pollutant concentrations varies in stringency depending on the desired use of sediments. Sediments may violate all aquatic health standards and still be fit for land application.

Nineteen of the 30 forebays sampled exceeded one or more aquatic limit standards considered for comparison. Fifteen forebays exceeded the FDEP TEL threshold for Cu, indicating that 15 of 30 sites are above the upper, or more conservative, limit in which no adverse effects are observed. Additionally, 12 forebays exceeded the FDEP TEL for Ni, and 9 for Zn. One forebay exceeded the PEL threshold for Cu, indicating that the Cu concentration at this particular site is above the lower, or most lenient, limit of the range of values associated with adverse effects. Also, 4 forebays exceeded the PEL for Zn (Table 4.6). Exceedance of the PEL standards will almost certainly result in adverse effects on aquatic health (FDEP, 1994), providing a reason for forebay sediment removal.

Table 4.6. Sites sampled that exceeded the limits of any of the metrics considered for comparison.

Toxicity Assesment Metric	Number of Violations per Pollutant Type					
Reference, metric	Cd	Cr	Cu	Ni	Pb	Zn
Aquatic Health						
FDEP, 1994						
TEL	2*	0	15	12	0	9
PEL	0	0	1	0	0	4
Land Application						
40 CFR503	0	0	0	0	0	0

In 2 of 30 cases, Cd was reported as < limit of quantitation (LOQ), or [Cd] < 5.0 mg/kg dry weight. These two sites may or may not exceed the TEL.

When comparing to the land application benchmark, there were clearly no observed concentrations of Cu, Zn, Ni, Cd, Cr, or Pb that exceeded the 40 CFR 503 limits. <u>All pollutants sampled met the 40 CFR 503 limits with factors of safety ranging from 5 to 13</u>, suggesting that the land application of accumulated forebay sediment will not pose an environmental threat, provided the spoils are quickly stabilized. A representative graph of Zn is provided (Figure 4.4). Graphs for the remaining metals are provided in Appendix H.

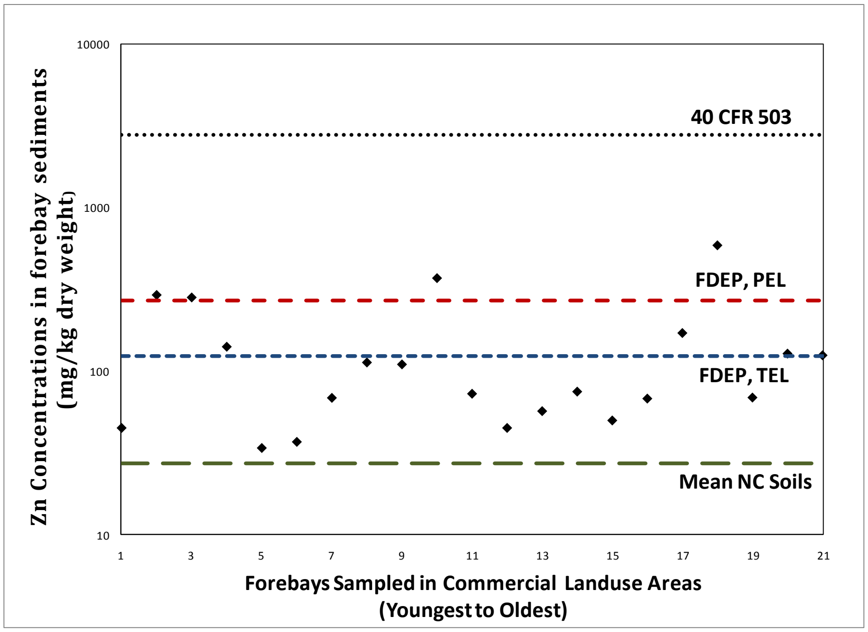

Figure 4.4. Zn concentrations for all sites in commercial landuse (youngest to oldest) relative to background concentrations, aquatic health metrics, and land application limits.

If forebay sediments are held to 40 CFR 503 standards, Arsenic (As), mercury (Hg), molybdenum (Mo), and selenium (Se) concentrations must also meet standards. Furthermore, other factors, such as PAHs, pathogens, and PCBs may also impact aquatic health. None of these pollutants were included in this study due to budgetary constraints.

Conclusions

1. Using 40 CFR503 land application standards as a benchmark, sampled forebay sediments are suitable for land application with respect to soil concentrations of Cu, Zn, Fe, Ni, Cd, Cr, and Pb. Other factors may hinder land application of sediments such as community resistance to land spreading sediment, resulting from temporary

foul odors. Also, pathogens, PCBs, and PAHs were *not* assessed in this study and could possibly limit land application.

2. Using the FDEP Effects Range Approach limits as benchmarks, concentrations of Cu and Zn are *likely* to pose a threat to aquatic health, which further emphasizes the necessity for forebay sediments to be routinely excavated. Concentrations of Cd, Cr, and Pb did not violate aquatic standards, and thus such constituents are not a rationale for sediment excavation.

3. There was no significant difference in metal concentrations between residential and commercial landuses; however, all violations of PEL standards occurred in commercial landuses and the maximum concentration exhibited by each pollutant was found in the commercial landuse category, indicating that commercial landuses may need to undergo more thorough testing than residential sites, before land application of sediments.

4. The fraction of sand was significantly higher in forebay sediments when compared to the fraction of sand in sediments accumulated near outlet structures; however, there was not a statistical difference in metal concentrations between the two. Even so, raw data showed that metal concentrations in outlet sediments were higher for all metals detected. When compared to metal concentrations associated with other studies where entire wet ponds were examined, the metal concentrations of the NC forebays tended

to be slightly lower. This possibly indicates the impact of coarser sediments at the inlet as well.

5. Authors recommend that forebay sediments be tested, near time of excavation and prior to land application, for bioavailable nitrogen, phosphorus, calcium, and potassium, to ensure adequate plant growth, and thus prevent erosion of sediments and associated contaminants. A simple example of nutrient calculations for potential plant growth is provided in Appendix L. Perhaps a subsequent study could establish that bioavailabilty of nutrients is not detrimental for plant growth. This would alleviate the need for case-by-case determination.

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APPENDICES

Appendix A: Methods of Data Collection for Chapter 1.

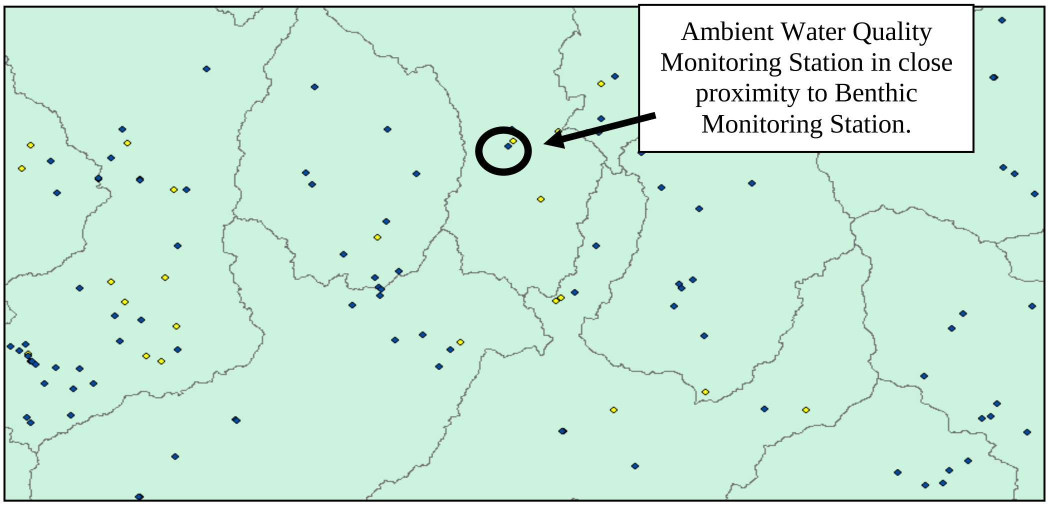
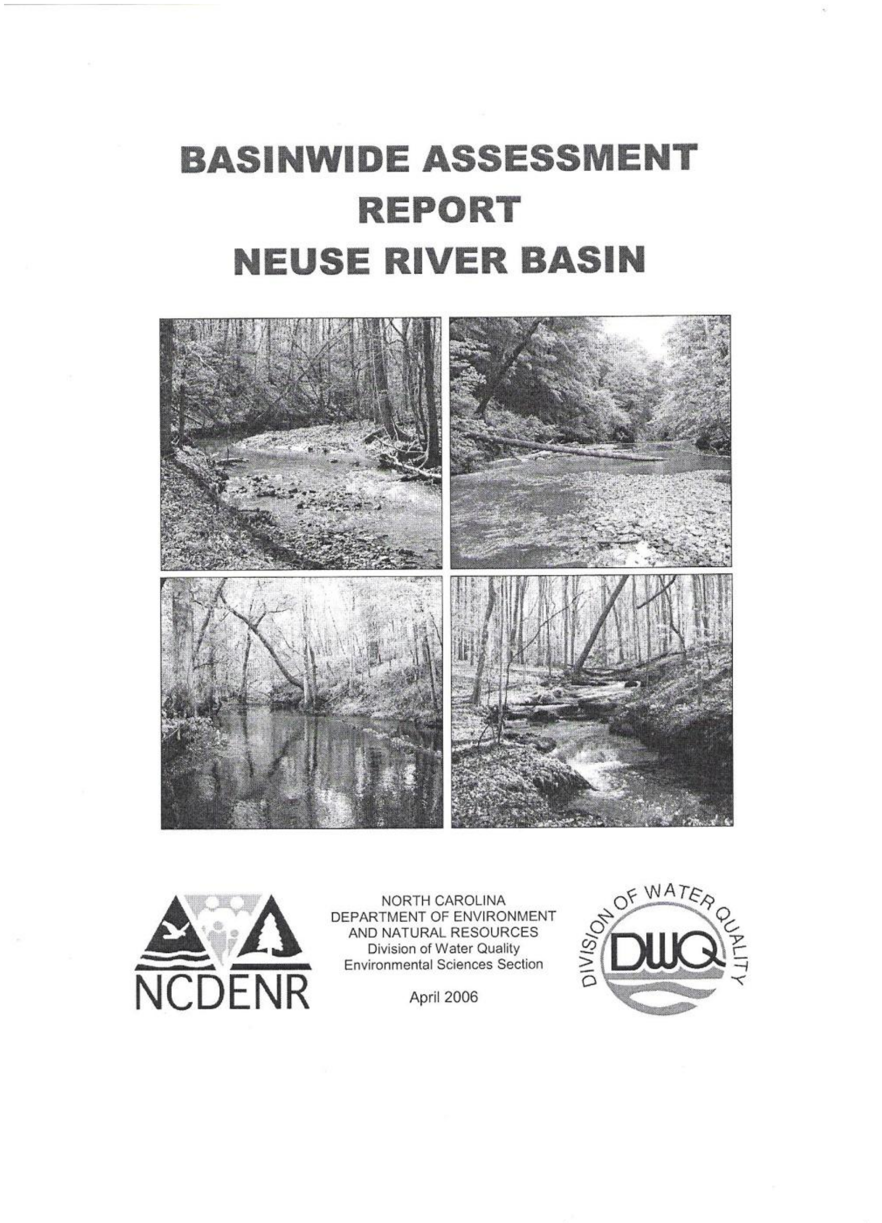


Figure A.1. GIS layer used to pair ambient water quality monitoring stations with benthic rating stations. (Only Sites within ¼ mile range of eachother were paired).



BASINWIDE ASSESSMENT REPORT
NEUSE RIVER BASIN

NCDENR

NORTH CAROLINA
DEPARTMENT OF ENVIRONMENT
AND NATURAL RESOURCES
Division of Water Quality
Environmental Sciences Section

April 2006

DIVISION OF WATER QUALITY
DWQ

Figure A.2. Cover sheet for a Basin Wide Assessment Report containing benthic ratings and ambient water quality data for the Neuse River Basin.

Ambient Monitoring System Station Summaries
NCDENR, Division of Water Quality
Basinwide Assessment Report

Location: ENO RIV AT US 501 NR DURHAM
Station #: J0770000 **Subbasin:** NEU01
Latitude: 36.07197 **Longitude:** -78.90864 **Stream class:** WS-IV NSW
Agency: NCAMBNT **NC stream index:** 27-2-(19)

Time period: 09/14/2000 to 08/23/2005

	#	#		Results not meeting EL				Percentiles					
	result	ND	EL	#	%	95%	Min	10th	25th	50th	75th	90th	Max
Field													
D.O. (mg/L)	65	0	<4	0	0		5.6	6.5	7.5	9.7	11.7	12.8	18.1
	65	0	<5	0	0		5.6	6.5	7.5	9.7	11.7	12.8	18.1
pH (SU)	65	0	<6	3	4.6		5.7	6.4	6.6	6.9	7.1	7.4	7.7
	65	0	>9	0	0		5.7	6.4	6.6	6.9	7.1	7.4	7.7
Salinity (ppt)	33	0	N/A				0	0	0	0.1	0.1	0.1	0.1
Spec. conductance (umhos/cm at 25°C)	65	0	N/A				30	80	98	112	138	183	293
Water Temperature (°C)	65	0	>32	0	0		3	7.5	10	17.2	24.8	26.9	29.3
Other													
Chloride (mg/L)	3	0	>250	0	0		6	6	6	14	14	14	14
Hardness (mg/L as CaCO3)	3	0	>100	0	0		10	10	10	18	21	21	21
TSS (mg/L)	30	4	N/A				0	2	4	5	8	28	41
Turbidity (NTU)	58	0	>50	2	3.4		2	4	4	6	15	34	110
Nutrients (mg/L)													
NH3 as N	22	9	N/A				0.01	0.01	0.02	0.02	0.08	0.27	0.5
NO2 + NO3 as N	22	0	>10	0	0		0.02	0.04	0.11	0.25	0.47	0.55	0.69
TKN as N	20	1	N/A				0.2	0.25	0.3	0.36	0.54	1	1.4
Total Phosphorus	22	2	N/A				0.01	0.02	0.03	0.04	0.07	0.14	0.5
Metals (ug/L)													
Aluminum, total (Al)	19	0	N/A				75	79	120	170	460	1300	2300
Arsenic, total (As)	19	19	>10	0	0		5	5	10	10	10	10	10
Cadmium, total (Cd)	19	19	>2	0	0		2	2	2	2	2	2	2
Chromium, total (Cr)	19	19	>50	0	0		25	25	25	25	25	25	25
Copper, total (Cu)	19	11	>7	0	0		2	2	2	2	3	3	5
Iron, total (Fe)	19	0	>1000	2	10.5	No	220	340	550	690	920	1800	2000
Lead, total (Pb)	19	19	>25	0	0		10	10	10	10	10	10	10
Manganese, total (Mn)	19	1	>200	0	0		10	37	45	53	110	160	180
Mercury, total (Hg)	19	19	>0.012	0	0		0.2	0.2	0.2	0.2	0.2	0.2	0.2
Nickel, total (Ni)	19	19	>25	0	0		10	10	10	10	10	10	10
Zinc, total (Zn)	19	14	>50	0	0		10	10	10	10	12	15	20

Fecal coliform (#/100mL)

# results:	Geomean	# > 400:	% > 400:	95%:
55	75	10	18	

Key:
result: number of observations
ND: number of observations reported to be below detection level (non-detect)
EL: Evaluation Level; applicable numeric or narrative water quality standard or action level
Results not meeting EL: number and percentages of observations not meeting evaluation level
95% : States whether there is 95% statistical confidence that the actual percentage of exceedances is at least 10% (20% for Fecal Coliform)
Stations with less than 10 results for a given parameter were not evaluated for statistical confidence

NCDENR, Division of Water Quality
Ambient Monitoring System Report
Neuse River Basin – March 2006
AMS-85

Figure A.3. Portion of the Neuse Basin Wide Assessment Report containing ambient water quality data from one station location used in Chapter 1.

There are eight ambient monitoring sites located in this subbasin. Four stations (Eno River at US 501, Eno River at SR 1004, Little River at SR 1461, and Little River at SR 1628) have had stable water chemistry since 2000. The remaining four stations have had parameters that have exceeded water quality standards or action level standards greater than 10% of the time at a 95% statistical confidence level (see Ambient Monitoring Report).

Table 2. Waterbodies monitored in Subbasin 01 in the Neuse River basin for basinwide assessment, 2000 and 2005.

Map #[1]	Waterbody	County	Location	2000	2005
B-1	Sevenmile Cr	Orange	SR 1120	Good-Fair	Good-Fair
B-2	Eno R	Orange	SR 1336	Good	Good-Fair
B-3	Eno R	Orange	SR 1569	Excellent	Good-Fair
B-4	Eno R	Durham	US 15/501	Excellent	Good-Fair
B-5	Eno R	Durham	SR 1004	Good	Good-Fair
B-6	Little R	Durham	SR 1461	Excellent	Good
B-7	S Fk Little R	Orange	SR 1538	Good	Good
B-8	N Fk Little R	Orange	SR 1538	Good-Fair	Good
B-9	Flat R	Durham	SR 1614	Good	Good
B-10	Deep Cr	Person	SR 1715	Good	Good
B-11	Smith Cr	Granville	SR 1710	Good	Good-Fair
B-12	New Light Cr	Wake	SR 1912	Good	Good-Fair
B-13	Upper Barton Cr	Wake	NC 50	Good-Fair	Fair
F-1	Eno R	Orange	SR 1336	Excellent	Excellent
F-2	S Fk Little R	Durham	SR 1461	Excellent	Excellent
F-3	N Fk Little R	Durham	SR 1461	Good	Good
F-4	N Flat R	Person	SR 1715	Excellent	Good
F-5	S Flat R	Person	NC 157	Good	Good
F-6	Deep Cr	Person	SR 1734	Excellent	Excellent
F-7	Knap of Reeds Cr	Granville	off SR 1117	--	Good-Fair (2004)
F-8	Ellerbe Cr	Durham	SR 1709	--	Poor
F-9	Smith Cr	Granville	SR 1710	Good-Fair	Good-Fair
F-10	Newlight Cr	Wake	SR 1911	--	Good
F-11	Upper Barton Cr	Wake	NC 50	Good	Good
F-12	Lower Barton Cr	Wake	SR 1844	--	Good (2004)
F-13	Horse Cr	Wake	SR 1923	--	Good (2004)

[1]B = benthic macroinvertebrate monitoring sites; F = fish community monitoring sites.

River and Stream Assessment

A benthic community sample was not collected in 2005 from North Fork Little River at SR 1519 (Orange County) due to low flow conditions. A collection at this site in 2000 indicated an increase in the number of EPT taxa collected (11 in 1995 to 17 in 2000).

Three of the fish community sites in subbasin 01 were sampled for the first time in 2004 as part of an urban index study conducted by North Carolina State University (NCSU), and are being supplementally included in this monitoring cycle. There are five NPDES facilities located above the fish community sites in this subbasin. The Wildwood Green WWTP (NC0063614, 0.1MGD) located about 2.5 miles above the Lower Barton Creek fish community site had an effluent toxicity limit violation on 8/26/03.

Figure A.4. Portion of the Neuse Basin Wide Assessment Report containing benthic macro-invertebrate ratings from multiple station locations used in Chapter 1.

Time period when AWQ data was collected.

Eco-region (P=Piedmont, C=Coastal, M=Mountain)

Location: ENO RIV AT US 501 NR DURHAM **P** Station #: J0770000

Time period: 09/14/2000 to 08/23/2005

	N	mg/L
DO	65	9.7
TSS	30	5
NH3	22	0.02
NO2+NO3	22	0.25
TKN	20	0.36
TP	22	0.04
Fecal (#/100 mL)	55	75

Date	B-rating
6/28/2005	GF
8/8/2000	E

Benthic Rating corresponding to AWQ data.

Date	F-rating
10/1/2003	E

Figure A.5. Pairing of ambient water quality data and benthic ratings at one station location in Neuse River Basin.

Appendix B: Statistical Analysis for Chapter 1, by Dr. Jason Osborne

Analysis of Story and Hunt's BMR data

Jason A. Osborne, December, 2008

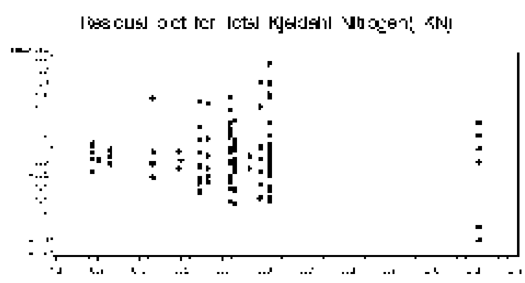
To investigate the potential association between the chemical composition of watersheds and benthic macroinvertebrate rating (BMR), seperate statistical models of the observed concentration of each of seven compounds were considered. The "design" of the survey was complete and crossed with respect to BMR and ecoregion, in the sense that there were stations in all $5 \times 3 = 15$ combinations of these two explanatory variables, as shown in the table of station counts for each combination given below:

	Benthic macroinvertebrate rating (BMR)					
Region	Excellent	Good	Good-Fair	Fair	Poor	Total
coastal	10	14	16	3	2	45
mountain	9	19	10	4	1	43
piedmont	8	20	31	39	8	106
Total	27	53	57	46	11	194
hline						

An average of approximately 50 measurements of concentration taken over time was used as the response variable in a model in which BMR and ecoregion were treated as factorial effects. The completeness of the design allows estimation and testing of possible ecoregion by BMR interactions and in their absence, main effects of each factor. The model may be stated as

$$\text{concentration} = \mu + \alpha_i + \beta_j + (\alpha\beta)_{ij} + \text{error}_{ijk}.$$

In this model, i, j and k are indices for ecoregion, rating and replicate station, respectively. The errors are assumed normally distributed with variances that are homogeneous across combinations of BMR and ecoregion. These models were fit using the GLM procedure of the SAS statistical software package (2004). For all concentrations except DO, diagnostic residual plots like the one for TSS below, revealed variability among concentration measurements that increased with the mean, so that it was necessary to consider the log-transformation of the concentration measurements to stabilize this variance. After this transformation, the assumptions of normality and homogeneity of variance seemed plausible.



1.2

F-ratios for tests of interactions and main effects of ecoregion and BMR are presented, along with corresponding p-values in parentheses, in the table below.

1

compound	interaction	ecoregion main effect	rating main effect
DO	0.47 (0.8728)	45.04 ($<$ .0001)	1.03 (0.3935)
Fecal	1.57 (0.1371)	33.76 ($<$ .0001)	0.48 (0.7517)
NH3	1.31 (0.2434)	4.50 (0.0126)	11.90 ($<$ .0001)
NO2-3	0.56 (0.8129)	7.45 (0.0008)	3.27 (0.0131)
TKN	2.57 (0.0115)	19.85 ($<$ .0001)	4.83 (0.0011)
TN	0.84 (0.5702)	10.76 ($<$ .0001)	4.12 (0.0033)
TP	0.70 (0.6896)	7.66 (0.0007)	5.52 (0.0003)
TSS	0.99 (0.4495)	5.29 (0.0059)	1.19 (0.3173)

F-ratios (p-values) for factorial effects

For all compounds except TKN, there was no evidence of interaction, so that effects of BMR and ecoregion were plausibly additive. Among these, NH3,NO2,TN and TP exhibited significant BMR effects. The mean concentration of each, is then tabulated against BMR below. Upon inspection, the concentration of each of these compounds appears to increase monotonically with BMR, so that multiple pairwise comparisons may not be needed.

	Compound			
BMR	NH3	NO2	TN	TP
E	-3.8243	-1.3524	-0.4599	-3.2556
G	-3.3021	-1.3545	-0.3673	-2.7143
GF	-3.1751	-0.9316	-0.1083	-2.4484
F	-2.6400	-0.5378	0.1805	-1.9827
P	-2.1521	0.2064	0.6537	-1.6182

For TKN, where there was significant evidence of ecoregion-BMR interaction, seperate tests for BMR effects on concentration were conducted for each ecoregion (using the `SLICE` option in an `LSMEANS` statement) of the `GLM` procedure. In the table below, F-ratios and p-values for tests of equality across BMR are given in the last two rows. The analysis indicates that the same monotonic association between the other chemical concentrations BMR is observed for TKN in the mountain and piedmont regions, but apparently not in the coastal region.

	ecoregion		
BMR	coastal	mountain	piedmont
E	-0.8329	-1.5958	-1.2312
G	-0.6166	-1.7340	-0.8928
GF	-0.6899	-1.4865	-0.7533
F	-0.5772	-1.1473	-0.5724
P	-0.9426	-0.9163	0.04318
F-ratio	0.67	2.51	12.40
(p-value)	(.6161)	(.0437)	$<$ .0001)

Reference:

- SAS Institute Inc., SAS 9.1.3 Help and Documentation, Cary, NC: SAS Institute Inc., 2000-2004.

2

Appendix C: Statistical Code and Output Examples for Chapter 2

All statistical analysis was completed with SAS statistical software package (2004). First, the distribution of data was checked for normality using Cramer-von Mises. Cramer-von Mises tests the following hypotheses:

H_o: The data are normally distributed.

H_a: The data are not normally distributed.

If $p < 0.05$, the null is rejected, and it is assumed that data are not normally distributed. If $p = 0.05$, the null is not rejected, and it is assumed that data are normally distributed.

The following is an example (using TN and TP data from site CP of Chapter 2) of the SAS code used to test for normality. The code below tests the distribution of both raw and log-transformed data. Raw data were defined as “TNnormal (TNin-TNout)” and “TPnormal (TPin-TPout)”, while log-transformed data were defined as “lTNnormal ($\log_{10}$TNin-$\log_{10}$TNout)” and “lTPnormal ($\log_{10}$TPin-$\log_{10}$TPout). The same basic code was modified for each data set analyzed, to test for individual data distributions.

```
      data CP;
input TNin TNout TPin TPout;
lTNin=log10(TNin);
lTNout=log10(TNout);
lTPin=log10(TPin);
lTPout=log10(TPout);
cards;
3      2.8    0.3    0.5
1.7    2.3    0.2    0.4
0.1    0.9    0.05   0.4
1.4    1.2    0.05   0.3
1      1.7    0.05   0.2
0.1    1.3    0.05   0.2
1.3    1.6    0.3    0.1
4.9    3.2    0.7    0.3
```

```
1.7   3.1   0.4   0.4
0.9   2.7   0.1   0.4
1.7   7.2   0.05  0.05
4.8   5.7   0.1   0.5

;
options formdlim = "_";

data c; set CP;
ods select BasicMeasures TestsForLocation GoodnessOfFit;
TNnormal=TNin-TNout;
lTNnormal= lTNin -lTNout;
TPnormal=TPin-TPout;
lTPnormal= lTNPn -lTPout;
proc univariate data =c;
var TNnormal lTNnormal TPnormal lTPnormal;
histogram/normal;
proc univariate plot normal;
histogram TNnormal/normal;
histogram lTNnormal/normal;
histogram TPnormal/normal;
histogram lTPnormal/normal;
run;
```

In this particular example, TN data at site CP were not normally distributed in raw form ($p<0.05$); however, upon log-transformation** the data reached a normal distribution ($p = 0.05$), as shown in the following SAS output. Also shown below are outputs of basic statistics for influent and effluent TN and TP data.

```
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                            The UNIVARIATE Procedure
                              Variable:  TNnormal

                           Basic Statistical Measures

            Location                      Variability

    Mean      -0.92500     Std Deviation             1.70514
    Median    -0.75000     Variance                  2.90750
    Mode        .          Range                     7.20000
                           Interquartile Range       1.25000
```

Tests for Location: Mu0=0

Test	-Statistic-		-----p Value------	
Student's t	t	-1.8792	Pr > |t|	0.0870
Sign	M	-3	Pr >= |M|	0.1460
Signed Rank	S	-26	Pr >= |S|	0.0425

Goodness-of-Fit Tests for Normal Distribution

Test	---Statistic----		-----p Value-----	
Kolmogorov-Smirnov	D	0.22362090	Pr > D	0.095
Cramer-von Mises	W-Sq	0.14267536	Pr > W-Sq	**0.025**
Anderson-Darling	A-Sq	0.85468804	Pr > A-Sq	0.021

** P< 0.05, thus CP data is NOT normal, try log-transformation:

The UNIVARIATE Procedure
Variable: **lTNnormal**

Basic Statistical Measures

Location		Variability	
Mean	1.514008	Std Deviation	1.42767
Median	1.152259	Variance	2.03824
Mode	.	Range	4.40879
		Interquartile Range	1.32646

Tests for Location: Mu0=0

Test	-Statistic-		-----p Value------	
Student's t	t	3.673591	Pr > |t|	0.0037
Sign	M	5	Pr >= |M|	0.0063
Signed Rank	S	38	Pr >= |S|	0.0010

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The UNIVARIATE Procedure
Fitted Distribution for **lTNnormal2**

Goodness-of-Fit Tests for Normal Distribution

Test	---Statistic----		-----p Value-----	
Kolmogorov-Smirnov	D	0.21105627	Pr > D	0.143
Cramer-von Mises	W-Sq	0.08595809	Pr > W-Sq	0.159
Anderson-Darling	A-Sq	0.50693915	Pr > A-Sq	0.166

The following are basic statistics for influent and effluent TN and TP data:

The UNIVARIATE Procedure
Variable: TNin

Moments

N	12	Sum Weights	12
Mean	1.88333333	Sum Observations	22.6
Std Deviation	1.58506285	Variance	2.51242424
Skewness	1.10516432	Kurtosis	0.4272222
Uncorrected SS	70.2	Corrected SS	27.6366667
Coeff Variation	84.1626293	Std Error Mean	0.45756823

Basic Statistical Measures

Location		Variability	
Mean	1.883333	Std Deviation	1.58506
Median	1.550000	Variance	2.51242
Mode	1.700000	Range	4.80000
		Interquartile Range	1.40000

Tests for Location: Mu0=0

Test	-Statistic-		-----p Value------	
Student's t	t	4.115962	Pr > |t|	0.0017
Sign	M	6	Pr >= |M|	0.0005
Signed Rank	S	39	Pr >= |S|	0.0005

Tests for Normality

Test	--Statistic---		-----p Value------	
Shapiro-Wilk	W	0.847842	Pr < W	0.0345

```
Kolmogorov-Smirnov    D      0.29604    Pr > D     <0.0100
Cramer-von Mises      W-Sq  0.136134    Pr > W-Sq   0.0320
Anderson-Darling      A-Sq  0.786533    Pr > A-Sq   0.0302


                 Quantiles (Definition 5)

                  Quantile      Estimate

                  100% Max          4.90
                  99%               4.90
                  95%               4.90
                  90%               4.80
                  75% Q3            2.35
                  50% Median        1.55
```

```
The UNIVARIATE Procedure
                     Variable:  TNin

                 Quantiles (Definition 5)

                  Quantile      Estimate

                  25% Q1            0.95
                  10%               0.10
                  5%                0.10
                  1%                0.10
                  0% Min            0.10


                   Extreme Observations

              ----Lowest----        ----Highest---

              Value      Obs        Value      Obs

                0.1        6          1.7        9
                0.1        3          1.7       11
                0.9       10          3.0        1
                1.0        5          4.8       12
                1.3        7          4.9        8


         Stem Leaf                     #             Boxplot
            4 89                       2                0
            3 0                        1                |
            2                                        +-----+
            1 034777                   6             *--+--*
            0 119                      3                |
              ----+----+----+----+
```

```
                 Normal Probability Plot
4.5+                                  *  ++*+++
   |                               *+++++
2.5+                        +++++++
   |                  *+*+*+*   *
0.5+         *    *++*++
    +----+----+----+----+----+----+----+----+----+----+
        -2        -1         0        +1        +2
```

```
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                         The UNIVARIATE Procedure
                             Variable:  TNout

                                 Moments

        N                          12    Sum Weights                 12
        Mean               2.80833333    Sum Observations          33.7
        Std Deviation      1.89134792    Variance            3.57719697
        Skewness           1.46995376    Kurtosis             1.7637234
        Uncorrected SS         133.99    Corrected SS        39.3491667
        Coeff Variation    67.3477006    Std Error Mean      0.54598512


                        Basic Statistical Measures

             Location                    Variability

             Mean     2.808333     Std Deviation            1.89135
             Median   2.500000     Variance                 3.57720
             Mode      .           Range                    6.30000
                                   Interquartile Range      1.70000


                        Tests for Location: Mu0=0

              Test           -Statistic-    -----p Value------

              Student's t    t  5.143608    Pr > |t|    0.0003
              Sign           M         6    Pr >= |M|   0.0005
              Signed Rank    S        39    Pr >= |S|   0.0005


                           Tests for Normality

           Test                  --Statistic---    -----p Value------
```

Shapiro-Wilk	W	0.838233	Pr < W	0.0264
Kolmogorov-Smirnov	D	0.251306	Pr > D	0.0355
Cramer-von Mises	W-Sq	0.130309	Pr > W-Sq	0.0389
Anderson-Darling	A-Sq	0.779862	Pr > A-Sq	0.0317

Quantiles (Definition 5)

Quantile	Estimate
100% Max	7.20
99%	7.20
95%	7.20
90%	5.70
75% Q3	3.15
50% Median	2.50

The UNIVARIATE Procedure
Variable: TNout

Quantiles (Definition 5)

Quantile	Estimate
25% Q1	1.45
10%	1.20
5%	0.90
1%	0.90
0% Min	0.90

Extreme Observations

----Lowest----		----Highest---	
Value	Obs	Value	Obs
0.9	3	2.8	1
1.2	4	3.1	9
1.3	6	3.2	8
1.6	7	5.7	12
1.7	5	7.2	11

```
Stem Leaf                     #            Boxplot
   7 2                        1               0
```

```
    6
    5 7                          1                |
    4                                             |
    3 12                         2             +-----+
    2 378                        3             *--+--*
    1 2367                       4             +-----+
    0 9                          1                |
      ----+----+----+----+
```

```
                   Normal Probability Plot
7.5+                                         *      +++
   |                                          ++++++
   |                                    *+++++
   |                                +++++
   |                           ++++*+ *
   |                     ++++*+* *
   |               * +*++* *
0.5+          * +++++
    +----+----+----+----+----+----+----+----+----+----+
        -2        -1         0        +1        +2
```

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The UNIVARIATE Procedure
Variable: TPin

Moments

N	12	Sum Weights	12
Mean	0.19583333	Sum Observations	2.35
Std Deviation	0.20052016	Variance	0.04020833
Skewness	1.6460487	Kurtosis	2.69312709
Uncorrected SS	0.9025	Corrected SS	0.44229167
Coeff Variation	102.393272	Std Error Mean	0.05788518

Basic Statistical Measures

Location		Variability	
Mean	0.195833	Std Deviation	0.20052

Median	0.100000	Variance	0.04021
Mode	0.050000	Range	0.65000
		Interquartile Range	0.25000

Tests for Location: Mu0=0

Test	-Statistic-		-----p Value------	
Student's t	t	3.383134	Pr > |t|	0.0061
Sign	M	6	Pr >= |M|	0.0005
Signed Rank	S	39	Pr >= |S|	0.0005

Tests for Normality

Test	--Statistic---		-----p Value------	
Shapiro-Wilk	W	0.77229	Pr < W	0.0046
Kolmogorov-Smirnov	D	0.266981	Pr > D	0.0189
Cramer-von Mises	W-Sq	0.175063	Pr > W-Sq	0.0091
Anderson-Darling	A-Sq	1.050762	Pr > A-Sq	0.0061

Quantiles (Definition 5)

Quantile	Estimate
100% Max	0.70
99%	0.70
95%	0.70
90%	0.40
75% Q3	0.30
50% Median	0.10

The UNIVARIATE Procedure
Variable: TPin

Quantiles (Definition 5)

Quantile	Estimate
25% Q1	0.05
10%	0.05
5%	0.05
1%	0.05
0% Min	0.05

```
                    Extreme Observations

               ----Lowest----        ----Highest---

               Value      Obs        Value      Obs

                0.05       11          0.2        2
                0.05        6          0.3        1
                0.05        5          0.3        7
                0.05        4          0.4        9
                0.05        3          0.7        8


         Stem Leaf                     #             Boxplot
            7 0                        1                0
            6
            5
            4 0                        1                |
            3 00                       2             +-----+
            2 0                        1             |  +  |
            1 00                       2             *-----*
            0 55555                    5             +-----+
              ----+----+----+----+
          Multiply Stem.Leaf by 10**-1


                          Normal Probability Plot
       0.75+                                         *
           |                                              +++++
           |                                         +++++
           |                                    *++++
           |                              *++*++
           |                          ++*++
           |                     +++*+*
       0.05+         *    * +*++* *
            +----+----+----+----+----+----+----+----+----+----+
                -2        -1         0        +1        +2
```

```
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                              The UNIVARIATE Procedure
                                  Variable:  TPout

                                      Moments

        N                           12    Sum Weights                 12
```

Mean	0.3125	Sum Observations	3.75
Std Deviation	0.14790199	Variance	0.021875
Skewness	-0.4947116	Kurtosis	-0.8015832
Uncorrected SS	1.4125	Corrected SS	0.240625
Coeff Variation	47.3286383	Std Error Mean	0.04269563

Basic Statistical Measures

Location		Variability	
Mean	0.312500	Std Deviation	0.14790
Median	0.350000	Variance	0.02188
Mode	0.400000	Range	0.45000
		Interquartile Range	0.20000

Tests for Location: Mu0=0

Test	-Statistic-		-----p Value------	
Student's t	t	7.319251	Pr > |t|	<.0001
Sign	M	6	Pr >= |M|	0.0005
Signed Rank	S	39	Pr >= |S|	0.0005

Tests for Normality

Test	--Statistic---		-----p Value------	
Shapiro-Wilk	W	0.919332	Pr < W	0.2804
Kolmogorov-Smirnov	D	0.222943	Pr > D	0.0962
Cramer-von Mises	W-Sq	0.0743	Pr > W-Sq	0.2300
Anderson-Darling	A-Sq	0.432976	Pr > A-Sq	>0.2500

Quantiles (Definition 5)

Quantile	Estimate
100% Max	0.50
99%	0.50
95%	0.50
90%	0.50
75% Q3	0.40
50% Median	0.35

```
                 The UNIVARIATE Procedure
                     Variable:  TPout

                 Quantiles (Definition 5)

                  Quantile      Estimate

                  25% Q1            0.20
                  10%               0.10
                  5%                0.05
                  1%                0.05
                  0% Min            0.05


                   Extreme Observations

           ----Lowest----        ----Highest---

           Value      Obs        Value      Obs

            0.05       11          0.4        3
            0.10        7          0.4        9
            0.20        6          0.4       10
            0.20        5          0.5        1
            0.30        8          0.5       12


     Stem Leaf                     #             Boxplot
        5 00                       2                |
        4 0000                     4             +-----+
        3 00                       2             *--+--*
        2 00                       2             +-----+
        1 0                        1                |
        0 5                        1                |
          ----+----+----+----+
      Multiply Stem.Leaf by 10**-1


                        Normal Probability Plot
   0.55+                                   * +++*+++
       |                         * * *+++*++++
       |                     * *++++++
       |                *++*++++
       |          +++*+++
   0.05+    ++++++*+
        +----+----+----+----+----+----+----+----+----+----+
            -2        -1         0        +1        +2
```

If data were normally distributed, as in this example, a t-test was used to test for significant differences between two parameters of interest; for instance, influent versus effluent nutrient concentrations in bioretention cells (Chapter 2), metal concentrations in residential versus commercial landuses (Chapter 4), and metal concentrations in forebays versus outlet structures (Chapter 4). The following hypotheses are tested when using a t-test:

Ho: There is no significant difference between two different parameters (e.g. there is not a significant difference between influent and effluent TN concentrations.)

Ha: There is a significant difference between the two parameters (e.g. there is a significant difference between influent and effluent TN concentrations.

Still using CP as an example, the following code was used for the completion of a t-test:

```
data CP_Ttest;
input Type$ TN TP;
lTN=log10(TN);
lTP=log10(TP);
datalines;
IN     3      0.3
IN     1.7    0.2
IN     0.1    0.05
IN     1.4    0.05
IN     1      0.05
IN     0.1    0.05
IN     1.3    0.3
IN     4.9    0.7
IN     1.7    0.4
IN     0.9    0.1
IN     1.7    0.05
IN     4.8    0.1
OUT    2.8    0.5
OUT    2.3    0.4
OUT    0.9    0.4
OUT    1.2    0.3
OUT    1.7    0.2
OUT    1.3    0.2
OUT    1.6    0.1
OUT    3.2    0.3
OUT    3.1    0.4
OUT    2.7    0.4
OUT    7.2    0.05
OUT    5.7    0.5
;
run;
```

```
proc ttest data=CP_Ttest;
class Type;
var lTN;
run;
proc ttest data=CP_Ttest;
class Type;
var TP;
run;
```

If $p < 0.05$ for any given t-test, the null hypothesis is rejected and the alternate hypothesis (there is a significant difference between the two variables in question) is assumed to be correct. . In the CP example, SAS output shows $p > 0.05$ for log-transformed TN data and raw TP data, thus there is no significant difference between influent and effluent TN and TP concentrations.

```
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                               The TTEST Procedure

                                   Statistics

                              Lower CL            Upper CL   Lower CL             Upper CL
Variable  Type             N      Mean      Mean      Mean    Std Dev   Std Dev    Std Dev   Std Err

lTN       IN              12    -0.286    0.0629    0.4117      0.389    0.5491     0.9323    0.1585
lTN       OUT             12    0.1981    0.3693    0.5406     0.1909    0.2695     0.4576    0.0778
lTN       Diff (1-2)            -0.673    -0.306    0.0597     0.3345    0.4325     0.6122    0.1766


                                     T-Tests

             Variable     Method           Variances       DF     t Value     Pr > |t|

             lTN          Pooled           Equal           22       -1.74       0.0966
             lTN          Satterthwaite    Unequal         16       -1.74       0.1018


                              Equality of Variances

                 Variable     Method       Num DF    Den DF    F Value    Pr > F

                 lTN          Folded F         11        11       4.15    0.0263
```

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The TTEST Procedure

Statistics

Variable	Type	N	Lower CL Mean	Mean	Upper CL Mean	Lower CL Std Dev	Std Dev	Upper CL Std Dev	Std Err
TP	IN	12	0.0684	0.1958	0.3232	0.142	0.2005	0.3405	0.0579
TP	OUT	12	0.2185	0.3125	0.4065	0.1048	0.1479	0.2511	0.0427
TP	Diff (1-2)		-0.266	-0.117	0.0325	0.1363	0.1762	0.2494	0.0719

T-Tests

Variable	Method	Variances	DF	t Value	Pr > |t|
TP	Pooled	Equal	22	-1.62	0.1190
TP	**Satterthwaite**	**Unequal**	**20.2**	**-1.62**	**0.1203**

Equality of Variances

Variable	Method	Num DF	Den DF	F Value	Pr > F
TP	**Folded F**	**11**	**11**	**1.84**	**0.3273**

Another method of analysis used in Chapter 2 involved the creation of general linear models (GLM)s. For each normalized data set, a model was created using influent concentrations as predictors for effluent concentrations. Carrying on the CP example, the following code was employed:

```
proc glm data=CP;
model lTNout=lTNin;
run;
proc glm data=CP;
model TPout=TPin;
run;
```

The GLM for TN was classified as “fair” whereas the GLM for TP was “very poor,” based on R^2 values. SAS output is provided for site CP below.

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The GLM Procedure

Number of Observations Read	12
Number of Observations Used	12

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The GLM Procedure

Dependent Variable: lTNout

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.39148770	0.39148770	9.60	0.0113
Error	10	0.40766622	0.04076662		
Corrected Total	11	0.79915391			

R-Square	Coeff Var	Root MSE	lTNout Mean
0.489878	54.66926	0.201907	0.369325

Source	DF	Type I SS	Mean Square	F Value	Pr > F
lTNin	1	0.39148770	0.39148770	9.60	0.0113

Source	DF	Type III SS	Mean Square	F Value	Pr > F
lTNin	1	0.39148770	0.39148770	9.60	0.0113

Parameter	Estimate	Standard Error	t Value	Pr > |t|
Intercept	0.3477311155	0.05870074	5.92	0.0001
lTNin	0.3435755408	0.11087038	3.10	0.0113

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The GLM Procedure

Number of Observations Read	12
Number of Observations Used	12

The GLM Procedure

Dependent Variable: TPout

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.00420484	0.00420484	0.18	0.6821
Error	10	0.23642016	0.02364202		
Corrected Total	11	0.24062500			

R-Square	Coeff Var	Root MSE	TPout Mean
0.017475	49.20307	0.153760	0.312500

Source	DF	Type I SS	Mean Square	F Value	Pr > F
TPin	1	0.00420484	0.00420484	0.18	0.6821

Source	DF	Type III SS	Mean Square	F Value	Pr > F
TPin	1	0.00420484	0.00420484	0.18	0.6821

Parameter	Estimate	Standard Error	t Value	Pr > |t|
Intercept	0.2934055582	0.06340461	4.63	0.0009
TPin	0.0975035327	0.23120002	0.42	0.6821

If data were not normally distributed, t-tests and GLMs should not be used and non-parametric tests would be more appropriate. The Kendall tau test was used to test for an association between two variables of interest. High tau values indicate a reasonable association whereas low tau values indicate little to no association. Due to the normal distributions of CP data, site L1 is now used as an example. Normality tests for site L1 showed that data could not be normalized with basic transformations, thus, the following code was used in lieu of a t-test (after testing for normality):

```
data L1;
input TNin TNout TPin TPout;
lTNin=log10(TNin);
lTNout=log10(TNout);
lTPin=log10(TPin);
lTPout=log10(TPout);
cards;
1.01  1.04  0.07  0.12
2.11  1.55  0.33  0.18
2.11  1.55  0.33  0.18
4.01  1.55  0.15  0.18
2.11  1.55  0.33  0.18
2.11  1.55  0.33  0.18
1.74  1.52  0.21  0.18
2.11  1.55  0.33  0.18
2.11  1.55  0.33  0.18
2.11  1.55  0.33  0.18
1.44  2.51  0.29  0.33
2.11  1.55  0.33  0.18
2.11  1.55  0.33  0.18
2.11  1.55  0.33  0.18
2.11  1.55  0.33  0.18
1.87  1.48  0.24  0.19
2.11  1.51  0.33  0.19
2.11  1.55  0.33  0.18
1.35  1.16  0.23  0.18
3.34  1.50  0.92  0.13
2.11  1.55  0.33  0.18
2.11  1.55  0.33  0.18
2.07  2.22  0.37  0.17
2.13  0.85  0.53  0.16
1.65  1.33  0.5   0.8
1.17  0.57  0.28  0.27
0.31  0.37  0.14  0.27
1.66  0.74  0.34  0.38
1.66  0.74  0.34  0.38
1.66  0.74  0.34  0.38
;
proc corr pearson spearman kendall data=L1;
```

```
var TNin TNout;
run;
proc corr pearson spearman kendall data=L1;
var TPin TPout;
   run;
```

Again, high tau values indicate a reasonable association whereas low tau values indicate little to no association. TN data showed more association between influent and effluent than respective TP data, however even the association between TNin and TNout ($\tau = 0.4$) was not very strong. Output for the non-parametric associations between influent and effluent TN and TP data for site L1 are provided.

```
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                                 The CORR Procedure

                          2  Variables:    TNin     TNout


                                 Simple Statistics

   Variable           N          Mean       Std Dev        Median       Minimum       Maximum

   TNin              30       1.95733       0.63787       2.11000       0.31000       4.01000
   TNout             30       1.38433       0.44832       1.55000       0.37000       2.51000


                        Pearson Correlation Coefficients, N = 30
                               Prob > |r| under H0: Rho=0

                                         TNin          TNout

                          TNin        1.00000        0.43662
                                                      0.0159


                          TNout       0.43662        1.00000
                                       0.0159


                        Spearman Correlation Coefficients, N = 30
```

Prob > |r| under H0: Rho=0

	TNin	TNout
TNin	1.00000	0.49377 0.0056
TNout	0.49377 0.0056	1.00000

Kendall Tau b Correlation Coefficients, N = 30
Prob > |r| under H0: Rho=0

	TNin	TNout
TNin	1.00000	**0.40061** 0.0072
TNout	0.40061 0.0072	1.00000

The CORR Procedure

2 Variables: TPin TPout

Simple Statistics

Variable	N	Mean	Std Dev	Median	Minimum	Maximum
TPin	30	0.33000	0.14283	0.33000	0.07000	0.92000
TPout	30	0.22767	0.12889	0.18000	0.12000	0.80000

Pearson Correlation Coefficients, N = 30
Prob > |r| under H0: Rho=0

	TPin	TPout
TPin	1.00000	0.12101 0.5241
TPout	0.12101 0.5241	1.00000

Spearman Correlation Coefficients, N = 30

```
           Prob > |r| under H0: Rho=0

                      TPin         TPout

    TPin           1.00000      -0.01053
                                  0.9560

    TPout         -0.01053       1.00000
                    0.9560


     Kendall Tau b Correlation Coefficients, N = 30
          Prob > |r| under H0: Rho=0

                      TPin         TPout

    TPin           1.00000      -0.01935
                                  0.8987

    TPout         -0.01935       1.00000
                        0.8987
```

UNIQUE ANALYSIS FOR CHAPTER 2

The last analysis conducted for Chapter 2 compared "cleaner" influent and "dirtier" influent. As described in Chapter 2, composited data were ranked in order of increasing magnitude and separated into 2 groups using average parking lot runoff concentrations of TN and TP (defined by Passeport and Hunt, (2009)) as points of separation. TNin and TNout data pairs were separated into those with TNin concentrations greater than or equal to 1.57 mg/L (median TN parking lot concentration found by Passeport and Hunt) and those with TNin concentrations less than 1.57 mg/L. Similarly, TP data were segregated using Passeport and Hunt's (2009) median concentration of 0.19 mg/L. Concentrations from each group (above median and below median) were then averaged (mean). Concentrations lower than "average" are labeled with LO", and those higher than average are labeled with "HI." Data distributions were checked for normality as previously described. Data sets were not normally distributed, therefore non-parametric analysis were used. Visual interpretations were also created in the form of scatterplots. All code is shown below.

```
data DirtyVsClean_Ttest;
input TNin_LO  TNout_LO  TPin_LO  TPout_LO       TNin_HI    TNout_HI
TPin_HI TPout_HI;
lTNin_LO=log10(TNin_LO);
lTNout_LO=log10(TNout_LO);
lTPin_LO=log10(TPin_LO);
lTPout_LO=log10(TPout_LO);
lTNin_HI=log10(TNin_HI);
lTNout_HI=log10(TNout_HI);
lTPin_HI =log10(TPin_HI);
lTPout_HI =log10(TPout_HI);
datalines;
0.1   0.77  0.005 0.35  1.59  1.09  0.2   0.046
0.1   0.9   0.01  0.46  1.6   0.7   0.2   .
0.1   1.3   0.03  1.3   1.63  .     0.2   0.4
0.1   0.4   0.04  0.10  1.65  1.33  0.2   0.1
0.1   0.8   0.04  0.45  1.66  0.74  0.2   0.1
0.2   0.6   0.039 0.111 1.66  0.74  0.20  0.08
0.31  0.37  0.039 0.044 1.66  0.74  0.21  0.18
0.31  0.91  0.04  0.11  1.66  1.05  0.23  0.18
0.381 0.33171     0.04  0.09  1.67  .     0.24  0.19
0.381 0.989 0.0429      0.094 1.7   2.3   0.242 .
0.40  0.37  0.05  0.09  1.7   3.1   0.242 .
0.57  0.20  0.046 0.039 1.7   7.2   0.25  0.36
0.57  0.36  0.046 0.045 1.74  1.52  0.25  0.07
0.581 1.016 0.05  0.08  1.78  1.06  0.25  0.45
0.60  0.60  0.05  0.4   1.84  1.09  0.25  0.21
0.63  0.65  0.05  0.3   1.87  1.48  0.25  0.24
0.70  0.65  0.05  0.2   1.91  0.96  0.258 0.035
0.71  0.758 0.05  0.2   1.91  0.59  0.258 .
0.71  0.785 0.05  0.05  1.96  0.6   0.26  0.18
0.75  0.18  0.05  0.05  1.96  0.47  0.27  0.19
0.75  0.20  0.05  0.1   1.97  1.09  0.28  0.27
0.75  0.71  0.05  0.05  2.039 1.05  0.29  0.33
0.762 0.47605     0.05  0.05  2.039 .     0.3   0.5
0.83  1.34  0.05  0.05  2.07  2.22  0.3   0.1
0.857 0.627 0.05  0.05  2.11  1.55  0.32  0.1
0.857 0.825 0.05  0.05  2.11  1.55  0.33  0.18
0.87  0.22  0.06  0.08  2.11  1.55  0.33  0.18
0.87  2.31  0.059 0.049 2.11  1.55  0.33  0.18
0.878 0.845 0.059 0.056 2.11  1.55  0.33  0.18
0.878 0.991 0.06  0.06  2.11  1.55  0.33  0.18
0.879 0.609 0.06  0.08  2.11  1.55  0.33  0.18
0.89  0.79  0.0636      0.0469      2.11  1.55  0.33  0.18
0.89  0.12  0.066 0.051 2.11  1.55  0.33  0.18
0.9   2.7   0.066 0.041 2.11  1.55  0.33  0.18
0.9   1     0.07  0.12  2.11  1.55  0.33  0.18
```

```
0.94  0.68  0.07  0.05  2.11  1.51  0.33  0.18
0.94  0.91  0.07  0.07  2.11  1.55  0.33  0.19
0.954 0.984 0.07  0.12  2.11  1.55  0.33  0.18
0.99  1.2   0.07  0.09  2.11  1.55  0.33  0.18
1     1.7   0.0707      0.0395      2.13  0.85  0.33  0.18
1     1.1   0.0707      0.055 2.25  0.69  0.333 0.046
1.01  0.43  0.073 0.05  2.25  0.70  0.333 0.1
1.01  0.44  0.073 0.047 2.26  1.39  0.34  0.38
1.01  1.04  0.0777      0.058 2.28  1.54  0.34  0.38
1.02  0.27  0.078 0.03  2.281 0.81  0.34  0.38
1.02  0.66  0.078 .     2.281 .     0.37  0.17
1.06  0.59  0.08  0.11  2.32  1.17  0.37  0.08
1.068 0.05  0.08  0.13  2.34  0.45  0.399 0.046
1.068 0.77  0.08  0.09  2.38  1.32  0.399 .
1.08  0.82  0.0819      0.061 2.45  0.57  0.4   0.4
1.09  0.09  0.08  0.06  2.45  0.64  0.40  0.09
1.11  0.49  0.0961      0.077 2.608 1.28  0.40  0.25
1.14  0.45  0.1   0.19  2.608 .     0.40  .
1.14  0.69  0.1   0.4   2.80  1.55  0.5   0.8
1.17  0.57  0.1   0.5   2.84  1.27  0.53  0.16
1.195 0.827 0.10  0.07  2.92  1.04  0.60  .
1.195 1.091 0.10  0.05  2.93  0.94  0.7   0.3
1.2   0.3   0.10  0.09  3     2.8   0.74  0.17
1.2   0.6   0.10  .     3.2   1.92  0.92  0.13
1.2   0.4   0.11  0.05  3.22  0.68  .     .
1.20  0.43  0.11  0.15  3.25  1.15  .     .
1.24  1.09  0.12  0.05  3.34  1.50  .     .
1.24  1.51  0.13  .     3.40  1.06  .     .
1.25  1.00  0.13  .     3.442 .     .     .
1.3   1.6   0.135 0.055 3.442 .     .     .
1.31  1.74  0.135 .     3.63  1.43  .     .
1.314 0.70  0.14  0.27  4.01  1.55  .     .
1.314 1.243 0.14  0.06  4.082 1.48  .     .
1.332 1.24  0.15  0.18  4.082 .     .     .
1.332 .     0.15  0.11  4.30  2.82  .     .
1.35  1.16  0.15  0.12  4.30  3.19  .     .
1.37  .     0.16  0.14  4.8   5.7   .     .
1.37  0.91  0.16  0.08  4.9   3.2   .     .
1.4   1.2   0.16  0.11  7.00  .     .     .
1.43  1.91  0.17  0.27  7.3   0.93  .     .
1.44  2.51  .     .     .     .     .     .
1.46  0.74  .     .     .     .     .     .
1.47  0.89  .     .     .     .     .     .
1.5   1.04  .     .     .     .     .     .
;
options formdlim = "_";
data c; set DirtyVsClean_Ttest;
ods select BasicMeasures TestsForLocation GoodnessOfFit;
TNin_LOvHI = TNin_HI - TNin_LO;
TNout_LOvHI= TNout_HI - TNout_LO;
TNlo_INvOUT = TNin_LO - TNout_LO;
TNhi_INvOUT= TNin_HI - TNout_HI;
```

```
LOG_TNin_LOvHI = lTNin_LO - lTNin_HI;
LOG_TNout_LOvHI= lTNout_LO - lTNout_HI;
LOG_TNlo_INvOUT = lTNin_LO - lTNout_LO;
LOG_TNhi_INvOUT= lTNin_HI - lTNout_HI;

TPin_LOvHI = TPin_LO - TPin_HI;
TPout_LOvHI= TPout_LO - TPout_HI;
TPlo_INvOUT = TPin_LO - TPout_LO;
TPhi_INvOUT=TPin_HI - TPout_HI;
LOG_TPin_LOvHI = lTPin_LO - lTPin_HI;
LOG_TPout_LOvHI= lTPout_LO - lTPout_HI;
LOG_TPlo_INvOUT = lTPin_LO - lTPout_LO;
LOG_TPhi_INvOUT= lTPin_HI - lTPout_HI;
proc univariate data =c;
var TNin_LOvHI TNout_LOvHI  TNlo_INvOUT  TNhi_INvOUT  LOG_TNin_LOvHI
LOG_TNout_LOvHI
LOG_TNlo_INvOUT  LOG_TNhi_INvOUT  TPin_LOvHI  TPout_LOvHI
TPlo_INvOUT  TPhi_INvOUT
LOG_TPin_LOvHI   LOG_TPout_LOvHI  LOG_TPlo_INvOUT  LOG_TPhi_INvOUT;
histogram/normal;
proc univariate plot normal;
histogram TNin_LOvHI/normal;
histogram TNout_LOvHI/normal;
histogram TNlo_INvOUT/normal;
histogram TNhi_INvOUT/normal;
histogram LOG_TNin_LOvHI/normal;
histogram LOG_TNout_LOvHI/normal;
histogram LOG_TNhi_INvOUT/normal;
histogram LOG_TNlo_INvOUT /normal;
histogram TPin_LOvHI/normal;
histogram TPout_LOvHI/normal;
histogram TPlo_INvOUT/normal;
histogram TPhi_INvOUT/normal;
histogram LOG_TPin_LOvHI/normal;
histogram LOG_TPout_LOvHI/normal;
histogram LOG_TPlo_INvOUT/normal;
histogram LOG_TPhi_INvOUT/normal;
run;
proc corr  kendall data=DirtyVsClean_Ttest;
var TNin_LO  TNin_HI;
run;
proc corr  kendall data=DirtyVsClean_Ttest;
var TNout_LO  TNout_HI;
run;
proc corr  kendall data=DirtyVsClean_Ttest;
var TNin_LO  TNout_LO;
run;
proc corr  kendall data=DirtyVsClean_Ttest;
var TNin_HI  TNout_HI;
run;
proc corr  kendall data=DirtyVsClean_Ttest;
var TPin_LO  TPin_HI;
```

```
run;
proc corr  kendall data=DirtyVsClean_Ttest;
var TPout_LO  TPout_HI;
run;
proc corr kendall data=DirtyVsClean_Ttest;
var TPin_LO  TPout_LO;
run;
proc corr  kendall data=DirtyVsClean_Ttest;
var TPin_HI  TPout_HI;
run;
proc gplot data=DirtyVsClean_Ttest;
plot TNout_LO * TNin_LO TNout_HI * TNin_HI;
symbol1 v=diamond i=r;
symbol2 v=circle i=r;
run;
proc gplot data=DirtyVsClean_Ttest;
plot TPout_LO *TPin_LO TPout_HI* TPin_HI;
symbol v=diamond i=r;
   run;
```

Statistical output comparing all variables is provided below, as well as scatterplots for visual interpretation. Tau-values of association are bolded.

```
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                                       The CORR Procedure

                                  2  Variables:    TNin_LO  TNin_HI
```

Simple Statistics

Variable	N	Mean	Std Dev	Median	Minimum	Maximum
TNin_LO	79	0.93033	0.36994	1.00000	0.10000	1.50000
TNin_HI	75	2.58259	1.10308	2.11000	1.59000	7.30000

Kendall Tau b Correlation Coefficients
Prob > |r| under H0: Rho=0
Number of Observations

	TNin_LO	TNin_HI
TNin_LO	1.00000	**0.97245** <.0001

	79	75
TNin_HI	0.97245	1.00000
	<.0001	
	75	75

The CORR Procedure

2 Variables: TNout_LO TNout_HI

Simple Statistics

Variable	N	Mean	Std Dev	Median	Minimum	Maximum
TNout_LO	77	0.84114	0.51974	0.77000	0.05000	2.70000
TNout_HI	66	1.51439	1.08647	1.41000	0.45000	7.20000

Kendall Tau b Correlation Coefficients
Prob > |r| under H0: Rho=0
Number of Observations

	TNout_LO	TNout_HI
TNout_LO	1.00000	**0.10288**
		0.2424
	77	64
TNout_HI	0.10288	1.00000
	0.2424	
	64	66

The CORR Procedure

2 Variables: TNin_LO TNout_LO

Simple Statistics

Variable	N	Mean	Std Dev	Median	Minimum	Maximum

TNin_LO	79	0.93033	0.36994	1.00000	0.10000	1.50000
TNout_LO	77	0.84114	0.51974	0.77000	0.05000	2.70000

Kendall Tau b Correlation Coefficients
Prob > |r| under H0: Rho=0
Number of Observations

	TNin_LO	TNout_LO
TNin_LO	1.00000 79	**0.20076** 0.0104 77
TNout_LO	0.20076 0.0104 77	1.00000 77

The CORR Procedure

2 Variables: TNin_HI TNout_HI

Simple Statistics

Variable	N	Mean	Std Dev	Median	Minimum	Maximum
TNin_HI	75	2.58259	1.10308	2.11000	1.59000	7.30000
TNout_HI	66	1.51439	1.08647	1.41000	0.45000	7.20000

Kendall Tau b Correlation Coefficients
Prob > |r| under H0: Rho=0
Number of Observations

	TNin_HI	TNout_HI
TNin_HI	1.00000 75	**0.10196** 0.2475 66
TNout_HI	0.10196 0.2475 66	1.00000 66

The CORR Procedure

2 Variables: TPin_LO TPin_HI

Simple Statistics

Variable	N	Mean	Std Dev	Median	Minimum	Maximum
TPin_LO	75	0.07907	0.03842	0.07000	0.00500	0.17000
TPin_HI	59	0.33753	0.13329	0.33000	0.20000	0.92000

Kendall Tau b Correlation Coefficients
Prob > |r| under H0: Rho=0
Number of Observations

	TPin_LO	TPin_HI
TPin_LO	1.00000	**0.91733**
		<.0001
	75	59
TPin_HI	0.91733	1.00000
	<.0001	
	59	59

The CORR Procedure

2 Variables: TPout_LO TPout_HI

Simple Statistics

Variable	N	Mean	Std Dev	Median	Minimum	Maximum
TPout_LO	70	0.13685	0.17936	0.08000	0.03000	1.30000
TPout_HI	52	0.20967	0.13777	0.18000	0.03500	0.80000

Kendall Tau b Correlation Coefficients
Prob > |r| under H0: Rho=0
Number of Observations

	TPout_LO	TPout_HI
TPout_LO	1.00000	**-0.11993**
		0.2496
	70	50
TPout_HI	-0.11993	1.00000
	0.2496	
	50	52

The CORR Procedure

2 Variables: TPin_LO TPout_LO

Simple Statistics

Variable	N	Mean	Std Dev	Median	Minimum	Maximum
TPin_LO	75	0.07907	0.03842	0.07000	0.00500	0.17000
TPout_LO	70	0.13685	0.17936	0.08000	0.03000	1.30000

Kendall Tau b Correlation Coefficients
Prob > |r| under H0: Rho=0
Number of Observations

	TPin_LO	TPout_LO
TPin_LO	1.00000	**0.04886**
		0.5668
	75	70
TPout_LO	0.04886	1.00000
	0.5668	
	70	70

The CORR Procedure

2 Variables: TPin_HI TPout_HI

Simple Statistics

Variable	N	Mean	Std Dev	Median	Minimum	Maximum
TPin_HI	59	0.33753	0.13329	0.33000	0.20000	0.92000
TPout_HI	52	0.20967	0.13777	0.18000	0.03500	0.80000

Kendall Tau b Correlation Coefficients
Prob > |r| under H0: Rho=0
Number of Observations

	TPin_HI	TPout_HI
TPin_HI	1.00000	**0.00763**
		0.9414
	59	52
TPout_HI	0.00763	1.00000
	0.9414	
	52	52

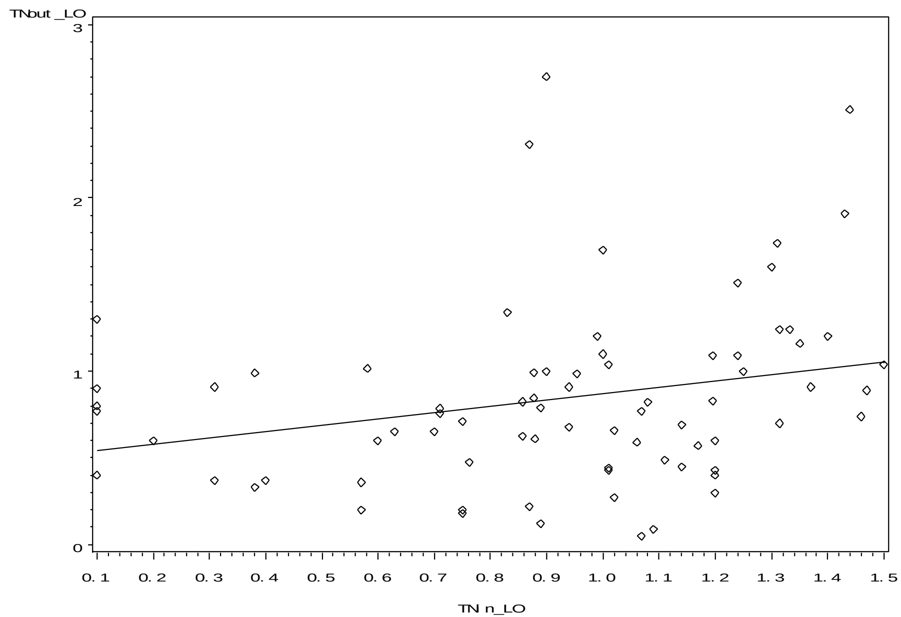


Figure C.1. Effluent TN concentrations versus respective "cleaner than average" influent TN concentrations.

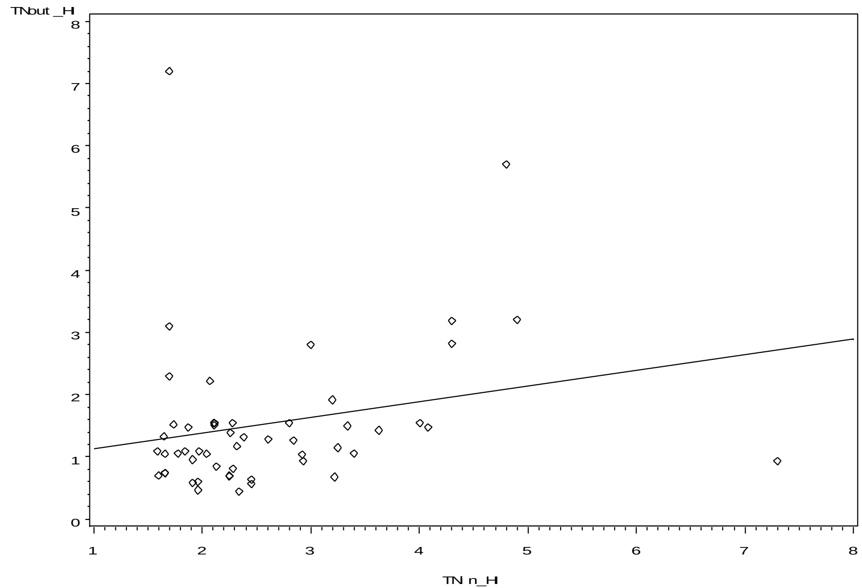


Figure C.2. Effluent TN concentrations versus respective "dirtier than average" influent TN concentrations.

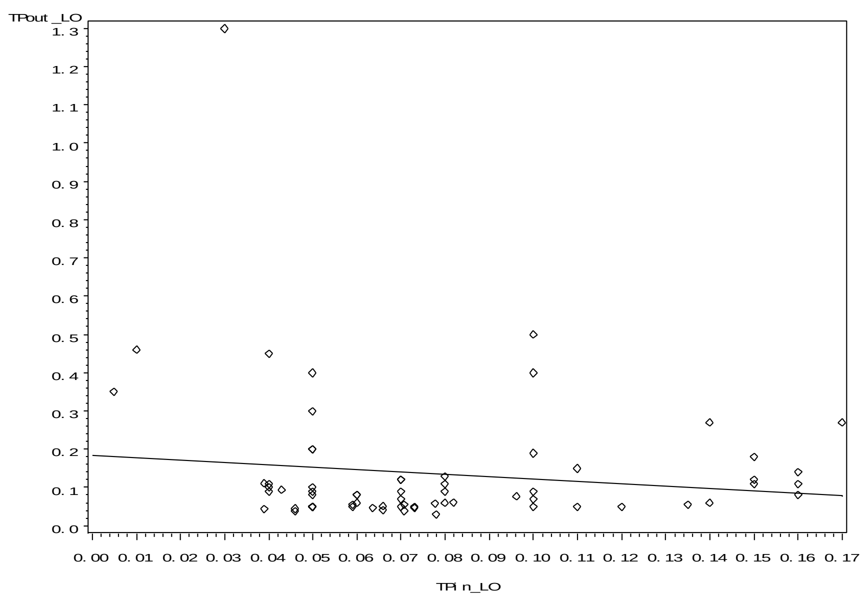


Figure C.3. Effluent TP concentrations versus respective "cleaner than average" influent TP concentrations.

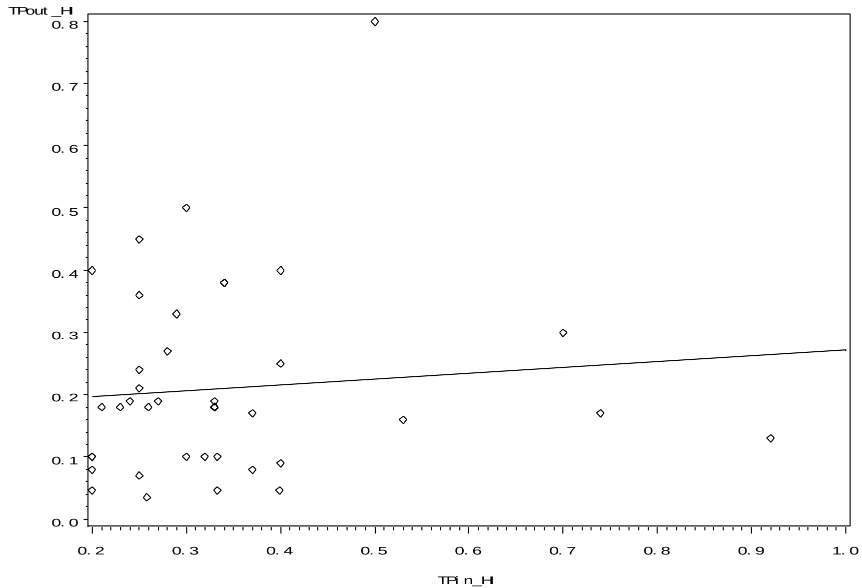


Figure C.4. Effluent TP concentrations versus respective "dirtier than average" influent TP concentrations.

Appendix D: An example of each model strength exhibited in Chapter 2 (influent concentrations used as predictors of effluent concentrations).

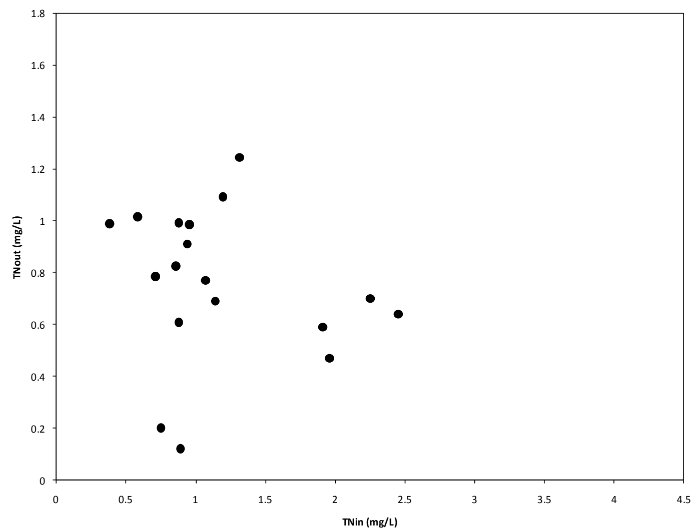


Figure D.1. An example of a "very poor" model using TNin as predictor for TNout (Graham S).

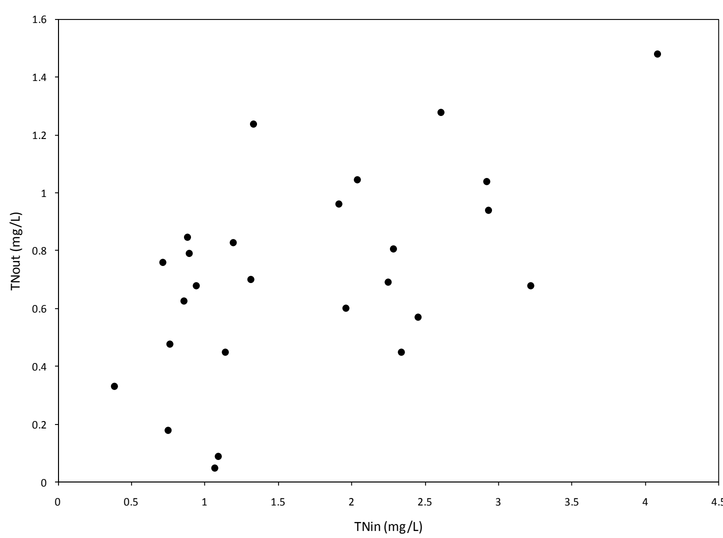


Figure D.2. An example of a "poor" model using TNin as predictor of TNout (Graham N).

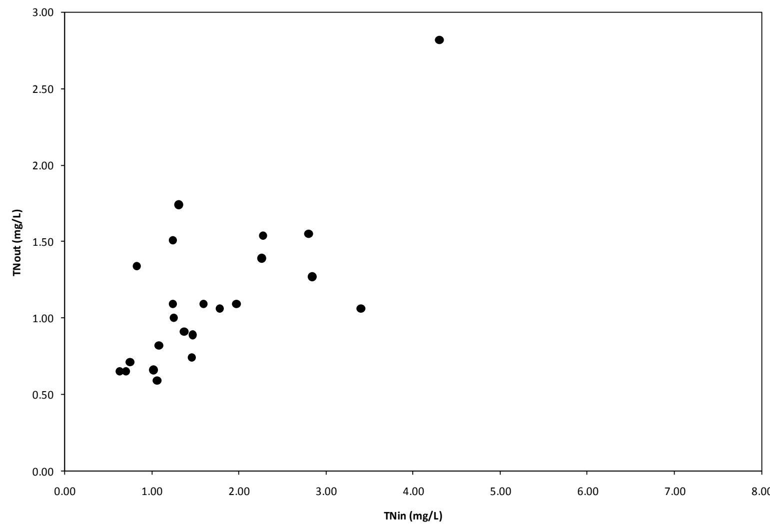


Figure D.3. An example of a "fair" model using TNin as predictor for TNout (HMBC site).

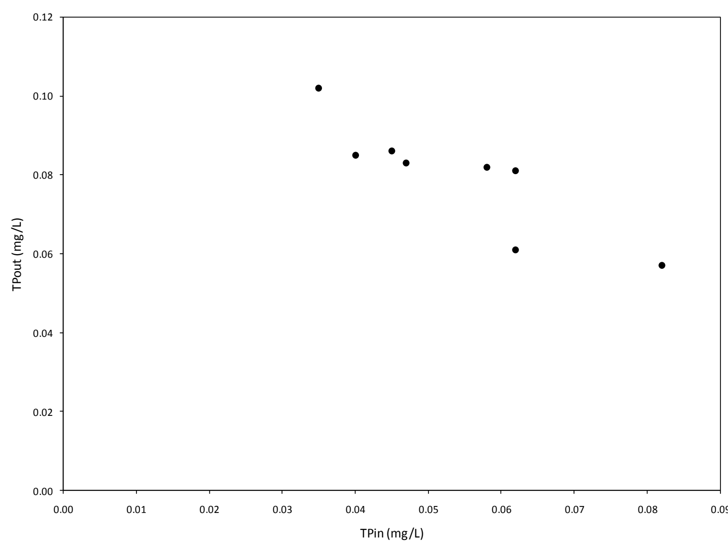


Figure D.4. An example of a "good" model using TPin as predictor for TPout (RMgrass).

APPENDIX E: Exceedance Probability Plots for Chapter 2.

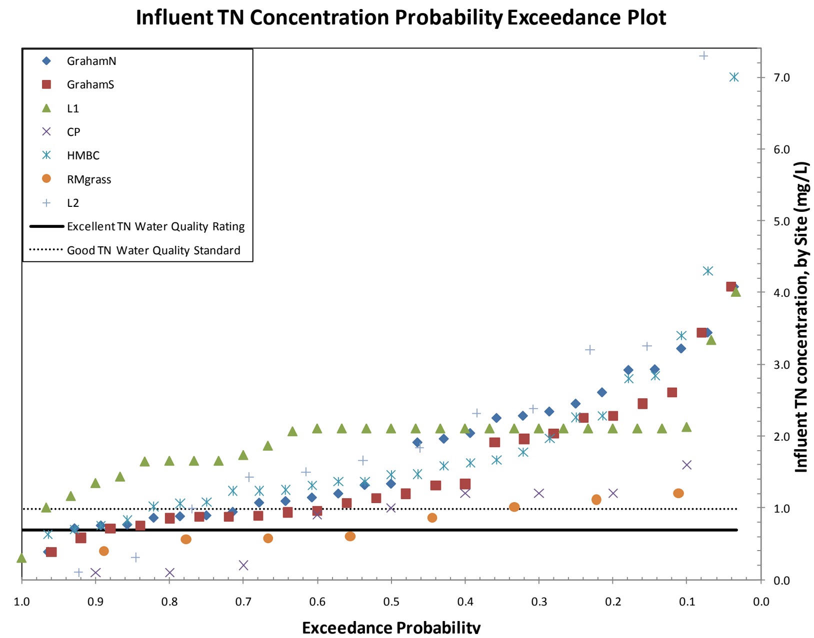


Figure E.1. Influent TN concentration Exceedance Probability Plot for all sites, as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

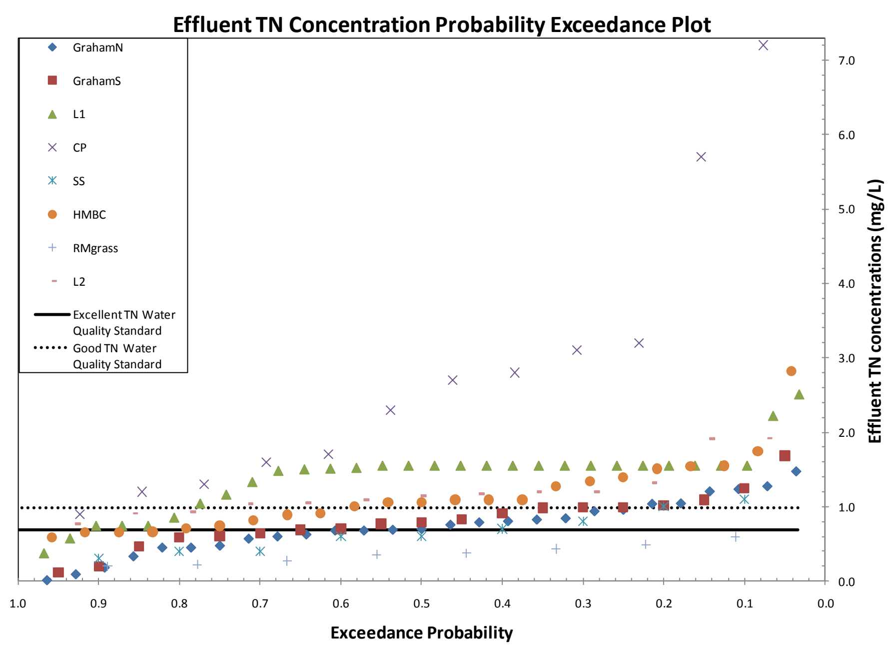


Figure E.2. Effluent TN concentration Exceedance Probability Plot for all sites, as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

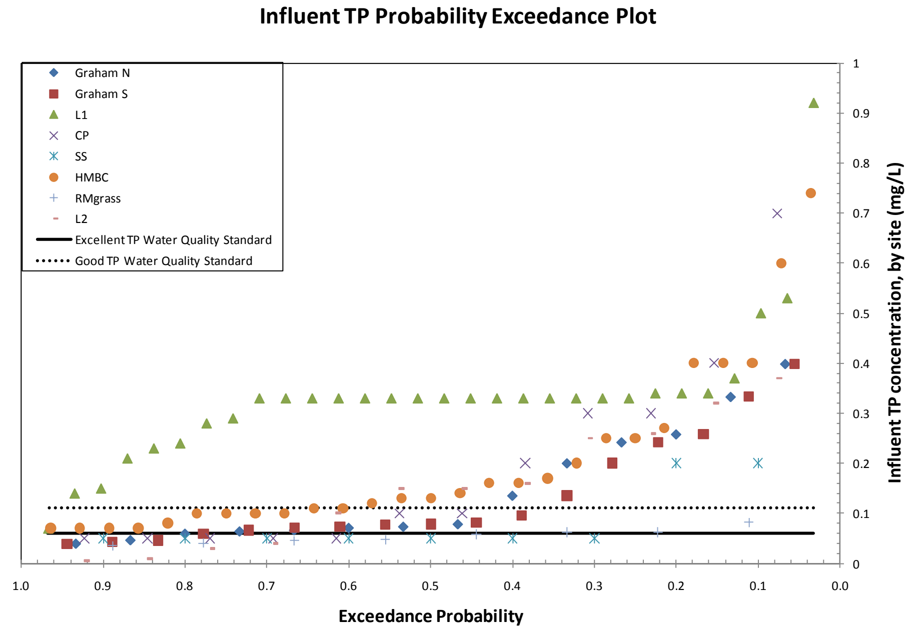


Figure E.3. Influent TP concentration Exceedance Probability Plot, for all sites as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

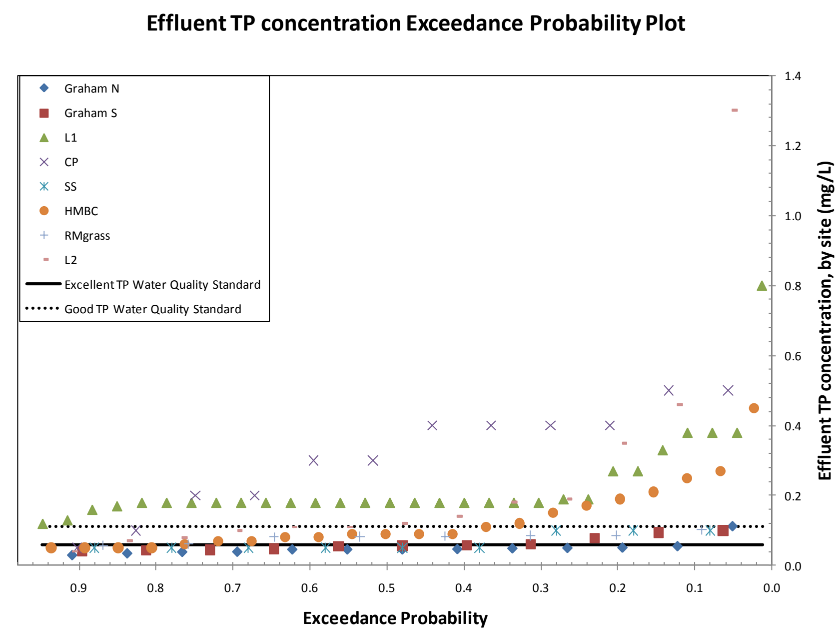


Figure E.4. Effluent TP concentration Exceedance Probability Plot for all sites,as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

TN Exceedance Probability Plot for Graham N/Graham S

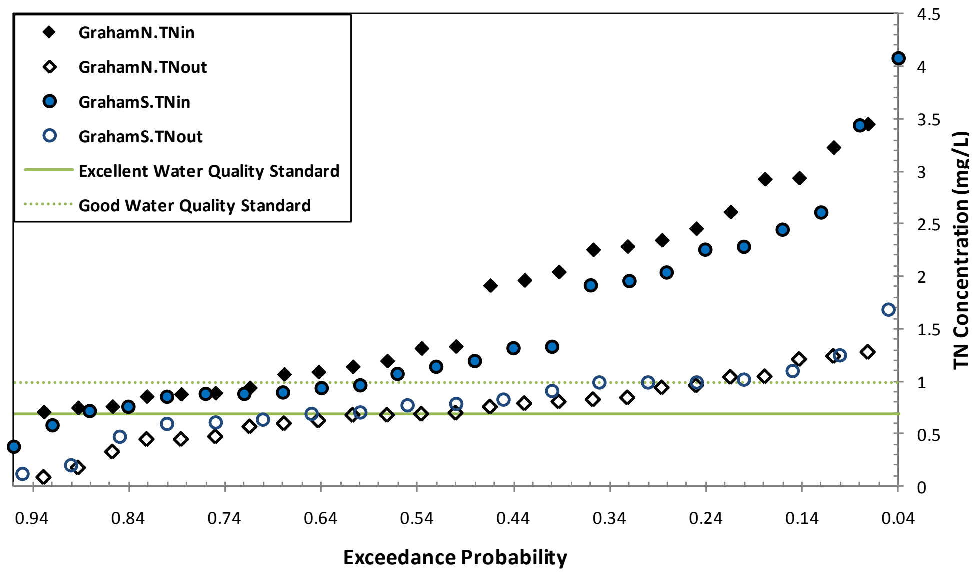


Figure E.5. Influent and Effluent TN concentration Exceedance Probabilities for Graham Sites, as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

TP Exceedance Probability Plot for Graham N/Graham S

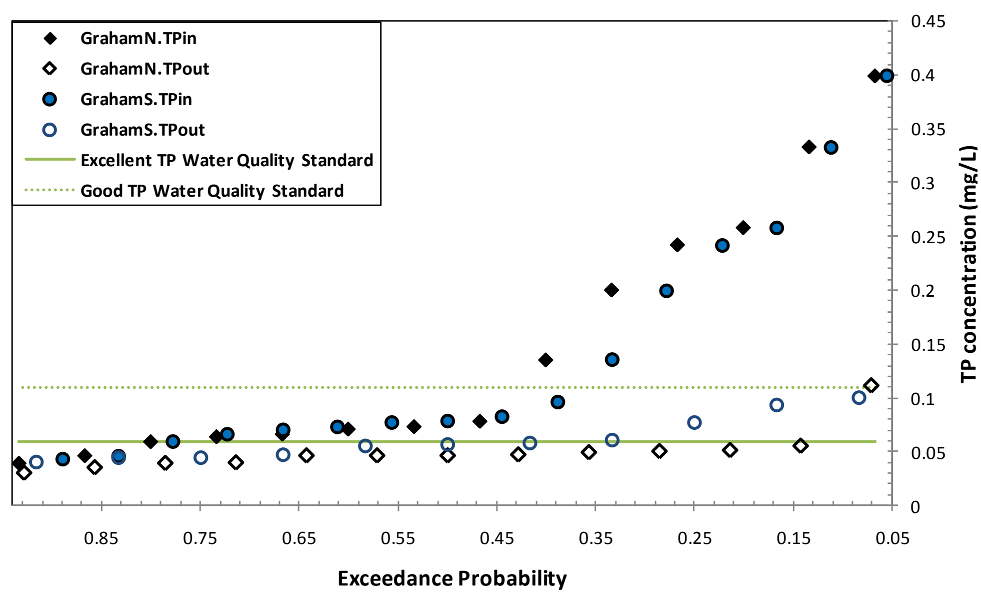


Figure E.6. Influent and Effluent TP concentration Exceedance Probabilities for Graham Sites, as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

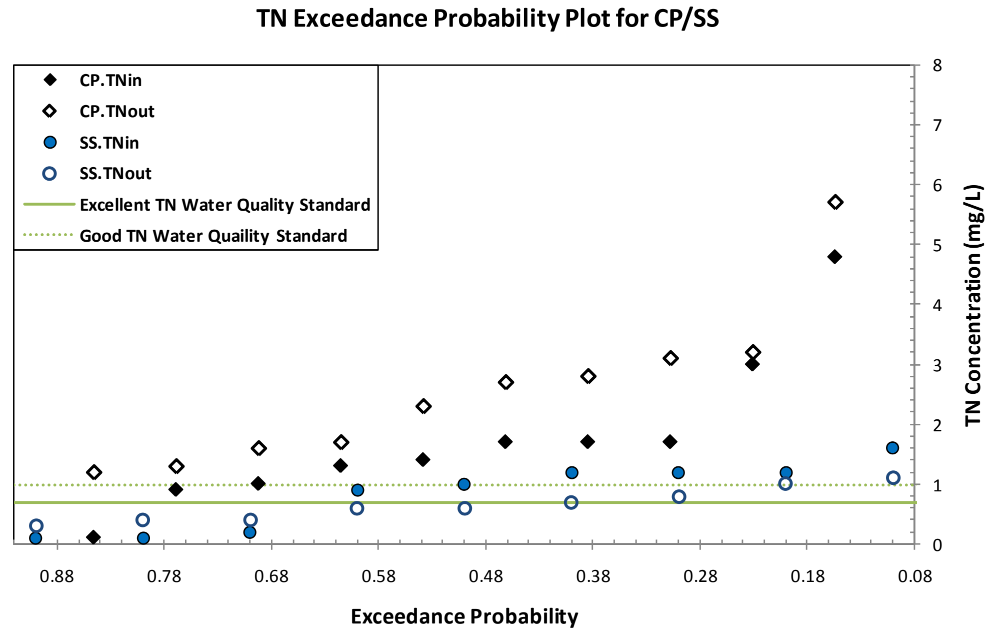


Figure E.7. Influent and Effluent TN concentration Exceedance Probabilities for CP and SS, as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

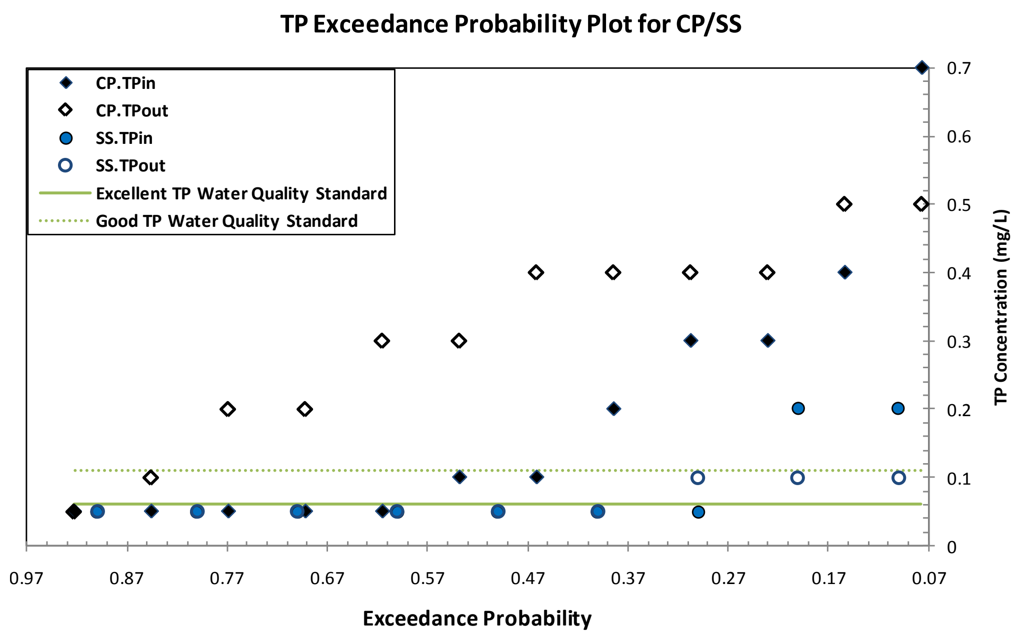


Figure E.8. Influent and Effluent TP concentration Exceedance Probabilities for CP and SS, as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

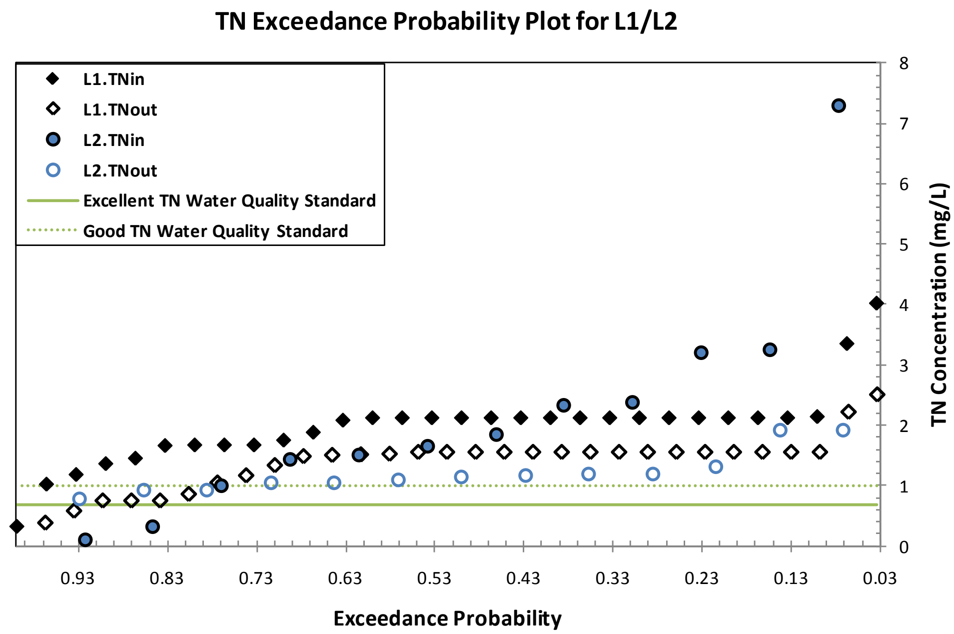


Figure E.9. Influent and Effluent TN concentration Exceedance Probabilities for L1 and L2, as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

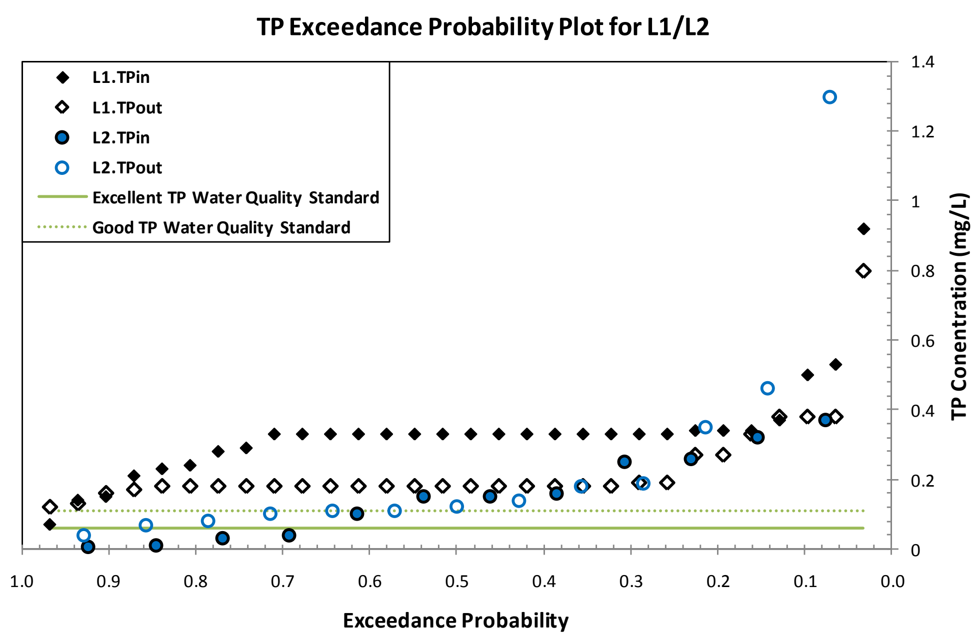


Figure E. 10. Influent and Effluent TP concentration Exceedance Probabilities for L1 and L2, as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

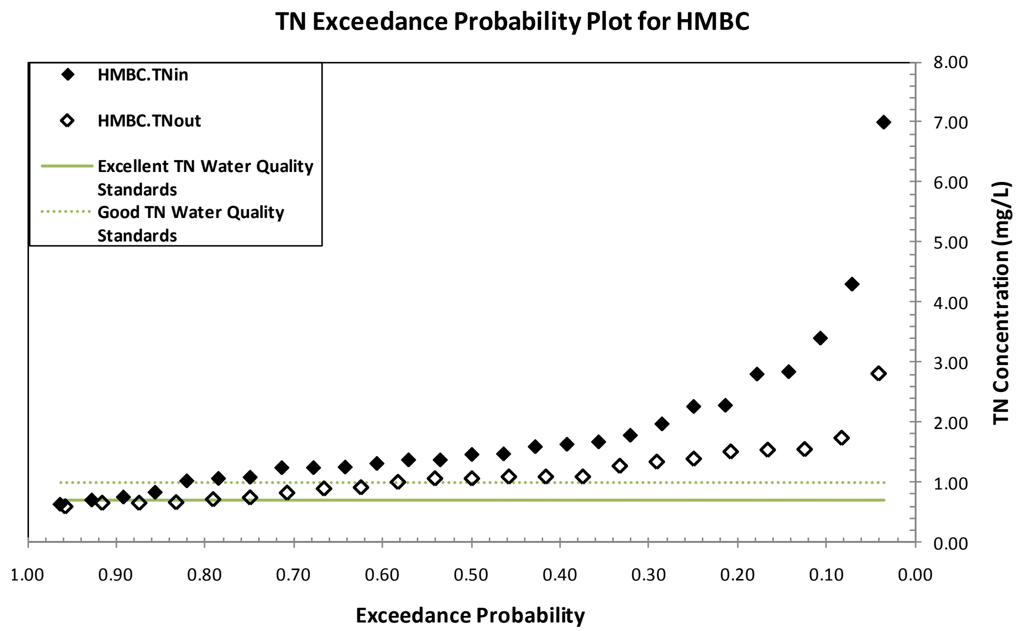


Figure E.11. Influent and Effluent TN concentration Exceedance Probabilities for HMBC, as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

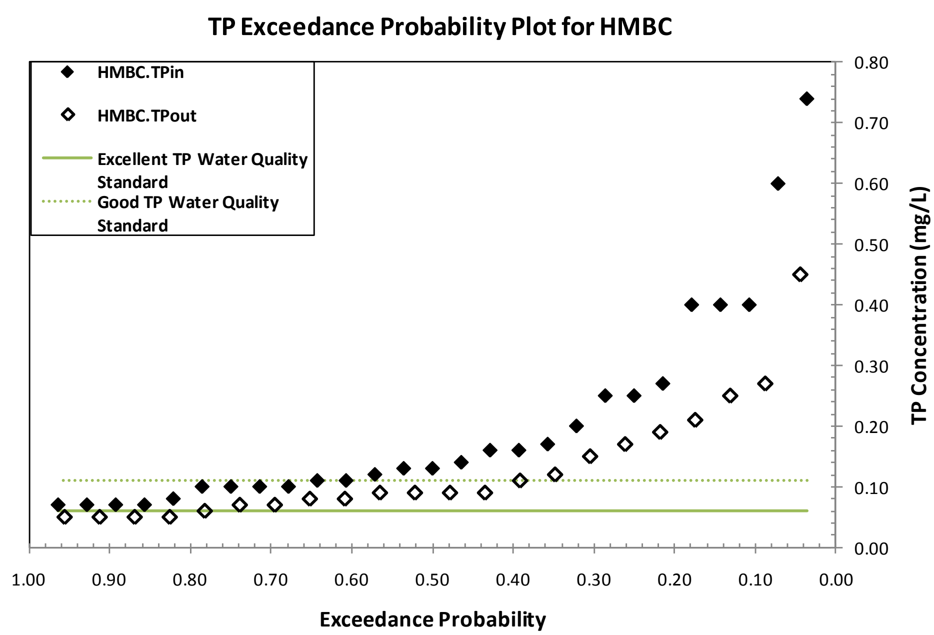


Figure E.12. Influent and Effluent TP concentration Exceedance Probabilities for HMBC, as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

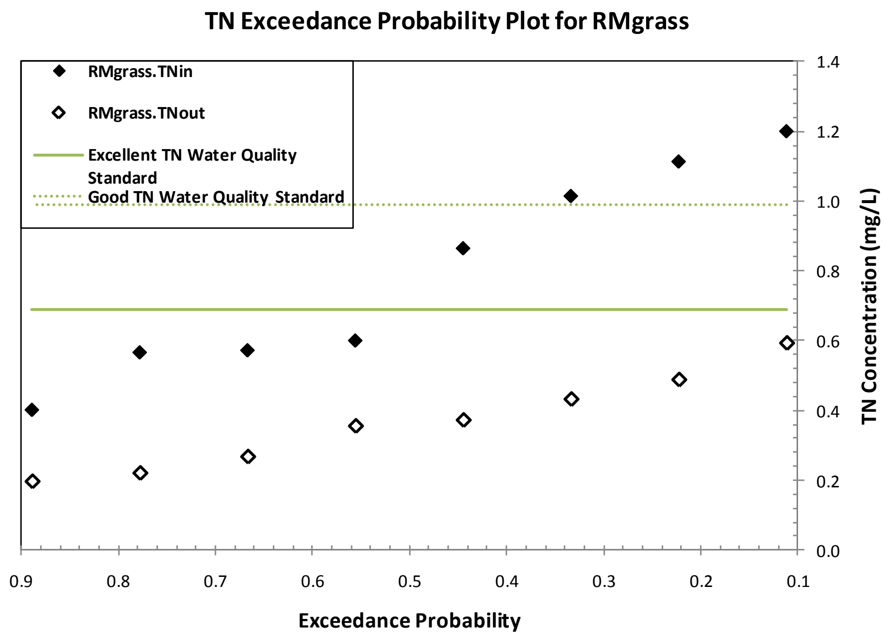


Figure E.13. Influent and Effluent TN concentration Exceedance Probabilities for RMgrass, as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

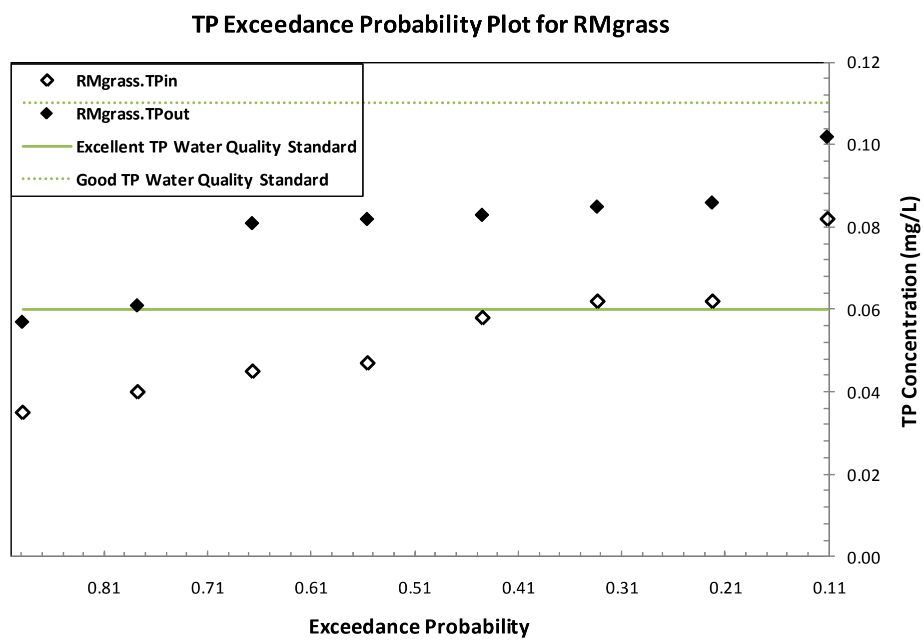


Figure E.14. Influent and Effluent TP concentration Exceedance Probabilities for RMgrass, as compared to "Excellent" and "Good" Water quality Standards (McNett et al., 2010)

Appendix F: Methods of Curve Fitting for Chapter 2 TN and TP Concentrations

Graphs were constructed showing the relationship between influent and effluent nutrient concentrations. Each graph was curve-fitted using an exponential function, a logarithmic function, a polynomial function with a maximum order of 2, and a power function in order to achieve the best fit.

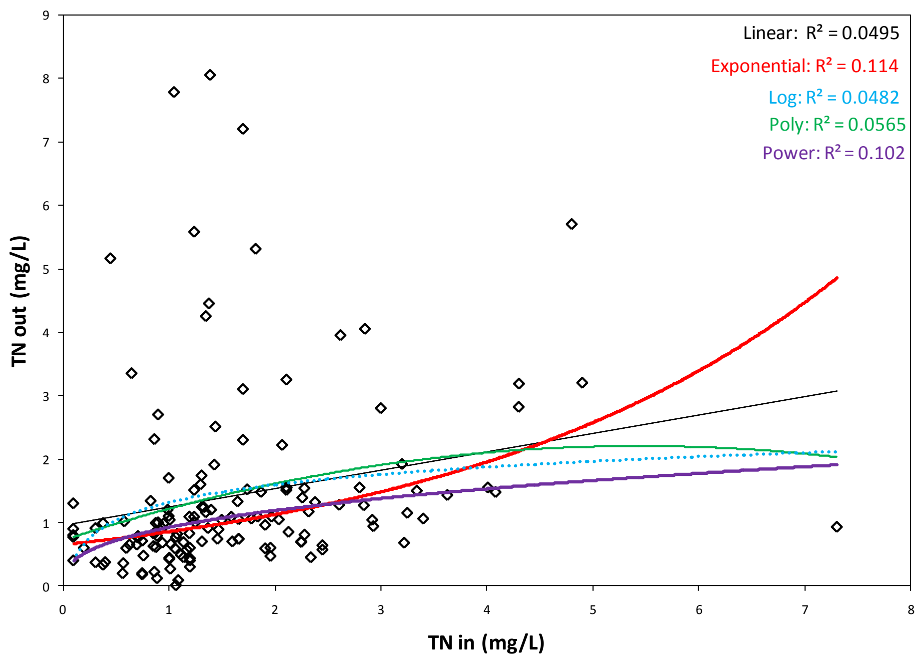


Figure F.1. Five functions used to determine the best-fit trend-line for TN (Best-fit = Power (purple).

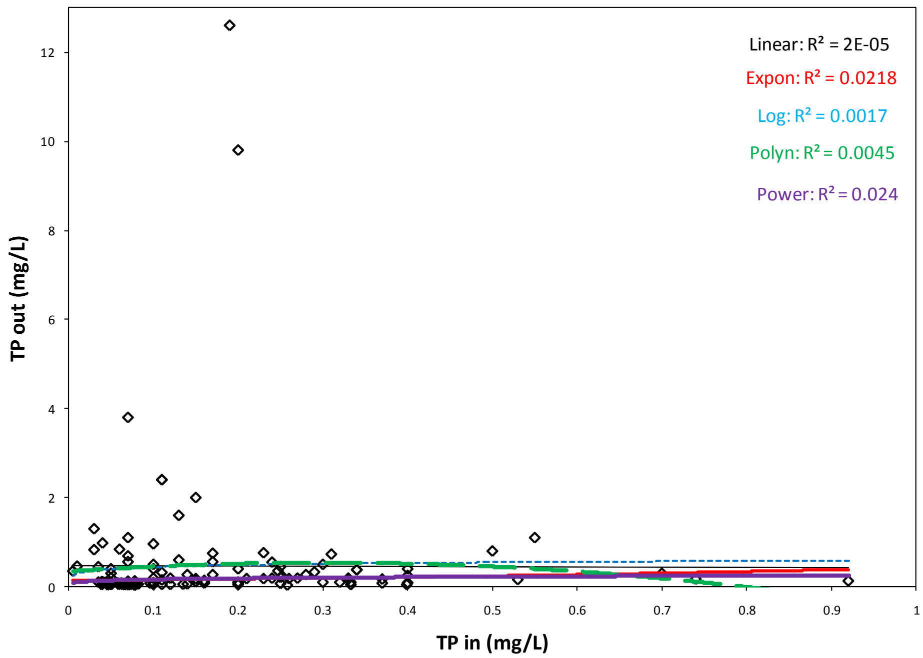


Figure F.2. Five functions used to determine the best-fit trend-line for TP.

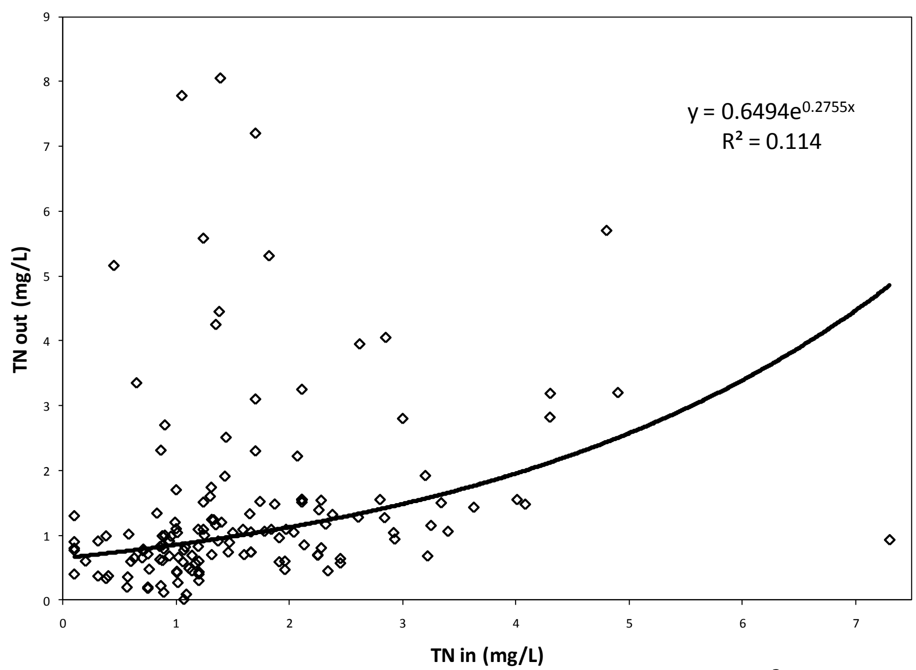


Figure F.3. Best curve-fit for TN: Exponential Function (R^2 = 0.11)

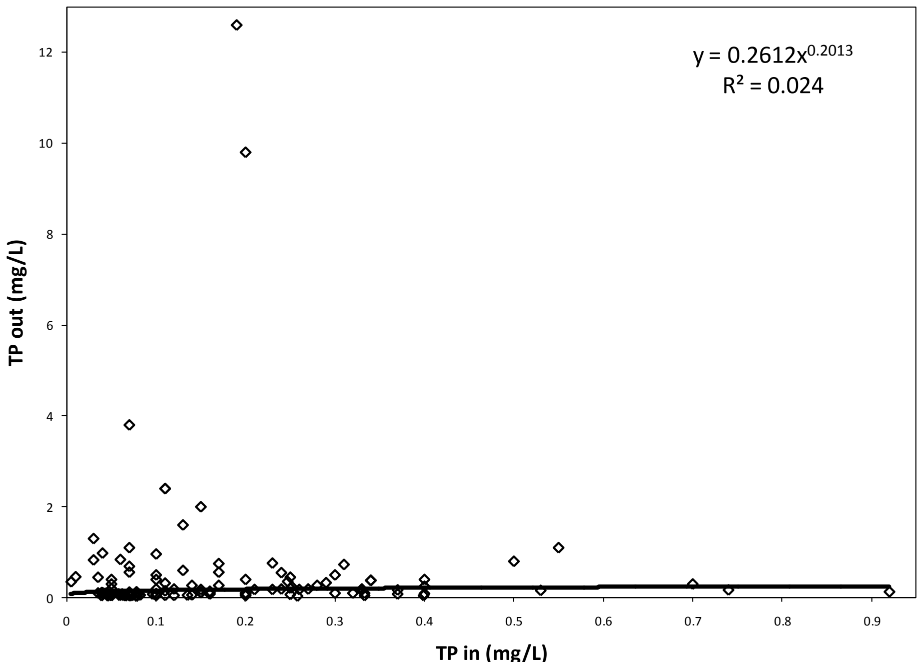


Figure F.4. Best curve-fit for TP: Power Function (R^2 = 0.02)

APPENDIX G: Forebay sampling Protocol and methodology

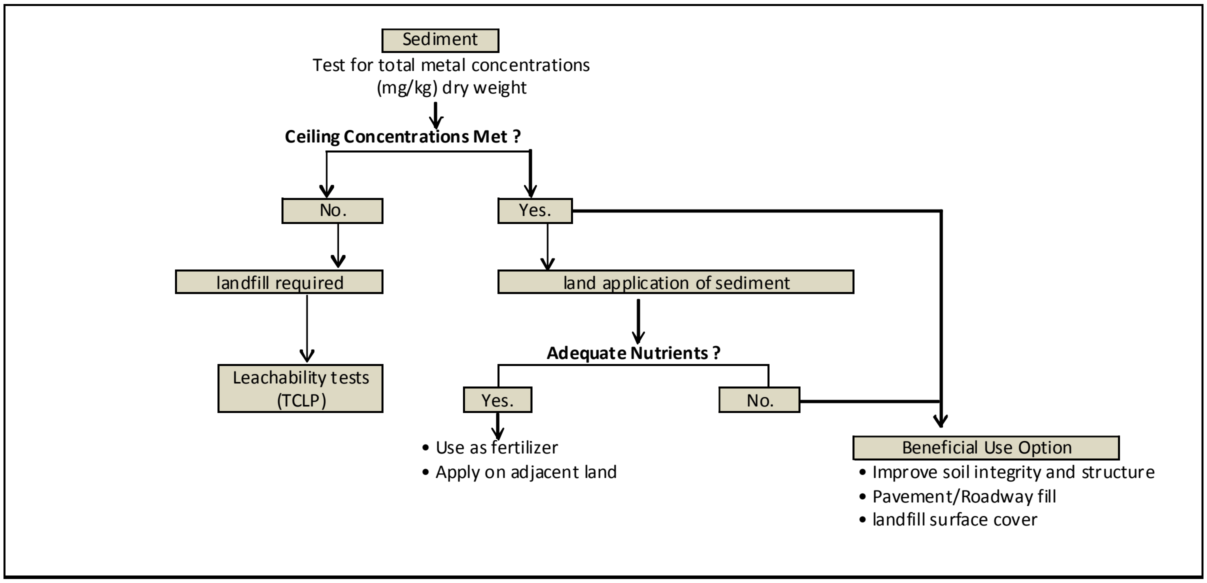


Figure G.1. The method of evaluating forebay sediment toxicity and establishing appropriate disposal practices.

Table G.1. The breakdown of forebays sampled by landuse and location

	Landuse		
Location	**Commercial**	**Residential**	**Totals**
Durham	9 (5)	8 (4)	17 (9)
Raleigh	8 (1)	0 (0)	8 (1)
Apex	2 (0)	0 (0)	2 (0)
Charlotte	1 (0)	1 (0)	2 (0)
High Point	1 (0)	0 (0)	1 (0)
	21 (6)	**9 (4)**	**30 (10)**

() values in parenthesis represent sites that had additional samples taken at the outlet structure.

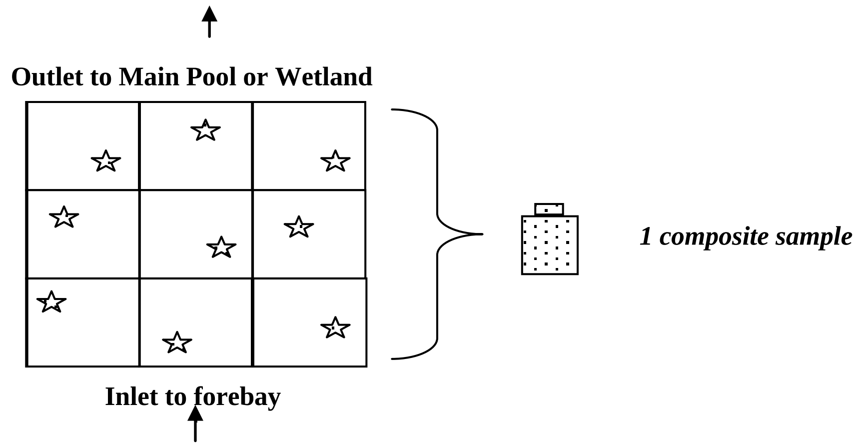
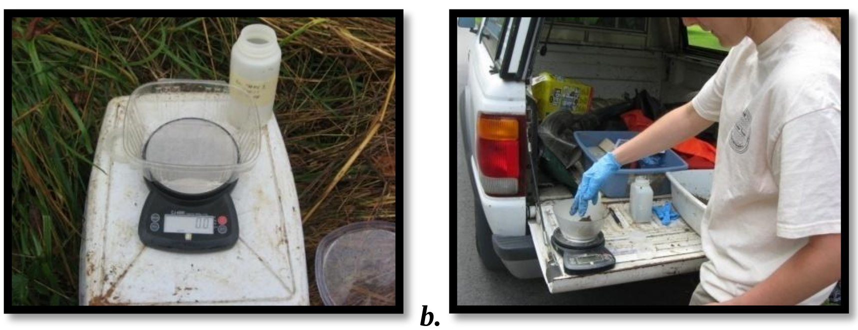


Figure G.2. A theoretical example of the grid-like pattern used in sampling of accumulated sediment in forebays, where a star indicates sampling location.



b.

Figure G.3. (a) Photo displaying field scale zeroed between samples (b) photo of sediments being weighed

Appendix H: Tables and figures related to the determination of forebay sediment toxicity

Table H.1. Standards set for the land application of biosolids by 40 CFR 503.

	*Ceiling Concentration Limits (CCL)	**Pollutant Concentration Limits (PCL)	***Cumulative Pollutant Loading Rate Limits for Biosolids (CPLR)	****Annual Pollutant Loading Rate Limits for Biosolids (APLR)
Pollutant	**mg/kg (dry wt.)**	**mg/kg (dry wt.)**	**kg/ha**	**kg/ha/year**
Cd	85	39	39	2
Cr	-	-	-	-
Cu	4300	1500	1500	75
Fe	-	-	-	-
Ni	420	420	420	21
Pb	840	300	300	15
Zn	7500	2800	2800	140

* Biosolids that are land applied cannot exceed these concentrations.

** Biosolids that are land applied do not need a permit if pollutants exist at or below the listed concentrations.

*** Total cumulative loading of pollutants cannot exceed listed values.

**** Annual cumulative loading of pollutants cannot exceed listed values.

Table H.2. Pollutant limits by standard compared to maximum observed concentration with factors of safety.

	Aquatic Health, FDEP		40 CFR 503	Forebay Study, 2009		Safety Factor		
Pollutant	TEL (mg/kg) dw	PEL (mg/kg) dw	Monthly Average Ceiling Concentrations (mg/kg) dw	(Max value reported)[1] (mg/kg) dw	Landuse of Max Value Reported	TEL	PEL	40 CFR 503
Cd	0.7	4.2	39	< 5.0	COM/RES	Exceeded by factor of 7	Exceeded by factor of 1.2	7.8
Cr	52.3	160	-	39	COM	1.3	4.1	-
Cu	18.7	108	1500	116	COM	Exceeded by factor of 6	Exceeded by factor of 1.1	12.9
Ni	15.9	42.8	420	34	COM	Exceedance by factor of 2	1.3	12.4
Pb	30.2	112	300	29	COM	1.04	4	10.3
Zn	124	271	2800	587	COM	Exceeded by factor of 4.7	Exceeded by factor of 2.2	4.8

dw = dry weight basis

[1] Of all forebays this is the "worst case" scenario

Table H.3. Study Data Compared to Existing Standards and Average Soils in North Carolina

Regulation (Source)		Cd	Cr	Cu	Ni	Pb	Zn
		Pollutant Concentrations (mg/kg, dry weight basis)					
Aquatic Health, FDEP Effects Range Approach Standards (FDEP, 1994)	***TEL***	0.7	52.3	18.7	15.9	30.2	124
	PEL	4.2	160.0	108	42.8	112	271
Freshwater Sediment Quality Guidelines, Consensus Based (MacDonald et al., 2000)	***TEC***	0.99	43.4	31.6	22.7	35.8	121
	PEC	4.98	111	149	48.6	128	459
40 CFR 503 Monthly Average Ceiling Concentrations		39.0	-	1500	420.0	300	2800
Mean N.C. soils		0.1	0.2	9.2	0.8	4.2	27.2
Site	**Landuse**	**Cd**	**Cr**	**Cu**	**Ni**	**Pb**	**Zn**
A	COM	ND	17.8	**29.0**	12.9	15.8	**370.0**
B	COM	ND	24.0	12.5	9.4	14.9	72.8
C*	COM	ND	22.7	**33.0**	13.0	12.7	**282.0**
D*	COM	ND	32.0	**31.8**	**22.0**	18.1	**141.0**
E	COM	ND	22.7	**23.0**	**31.0**	10.3	75.0
F	COM	ND	19.4	14.8	**16.9**	12.0	69.0
G	COM	ND	35.0	**116.0**	**34.0**	18.0	**292.0**
H	COM	ND	20.0	10.3	14.0	17.8	34.0
I	COM	ND	33.8	**53.0**	**18.4**	9.4	**125.0**
J	COM	ND	30.0	**36.6**	9.4	15.6	45.0
K	COM	ND	4.8	6.8	3.0	3.3	37.0
L	COM	ND	13.2	17.0	6.6	29.0	68.0
M	COM	ND	10.2	15.6	5.9	10.0	**171.0**
N	COM	ND	10.0	**28.5**	6.8	13.0	**587.0**
O	COM	ND	22.0	**39.0**	14.2	7.9	68.7
P	COM	ND	30.0	**28.7**	13.4	19.6	**128.0**
Q	COM	ND	14.0	15.4	5.9	19.0	50.0
R	COM	ND	11.5	15.0	8.8	6.3	45.0
S*	COM	ND	15.7	**18.9**	**16.7**	10.0	113.0
T*	COM	**2.5**	39.0	**27.0**	**29.3**	13.5	110.0
U*	COM	ND	11.0	16.0	10.0	11.0	57.0
V*	RES	ND	28.3	16.2	**15.9**	10.3	42.5
W	RES	ND	30.6	**34.9**	**18.3**	12.1	44.0
X	RES	ND	36.0	**19.0**	**18.4**	11.7	46.9
Y	RES	ND	16.1	13.1	10.6	8.5	26.4
Z	RES	**2.5**	35.0	**43.0**	**17.2**	25.4	**166.0**
AB*	RES	ND	14.8	5.1	14.0	22.3	40.5
CD*	RES	ND	33.0	15.1	**22.4**	11.4	63.0
EF*	RES	ND	11.0	10.6	14.8	3.6	53.0
GH	RES	ND	9.8	10.6	12.7	5.5	43.0

** Bolded concentrations indicate sediments violate one or more regulatory limit. Shaded concentrations are those that exceed a PEL or PEC limit. Fe is not accounted for in the regulations listed and were hence excluded from the table.

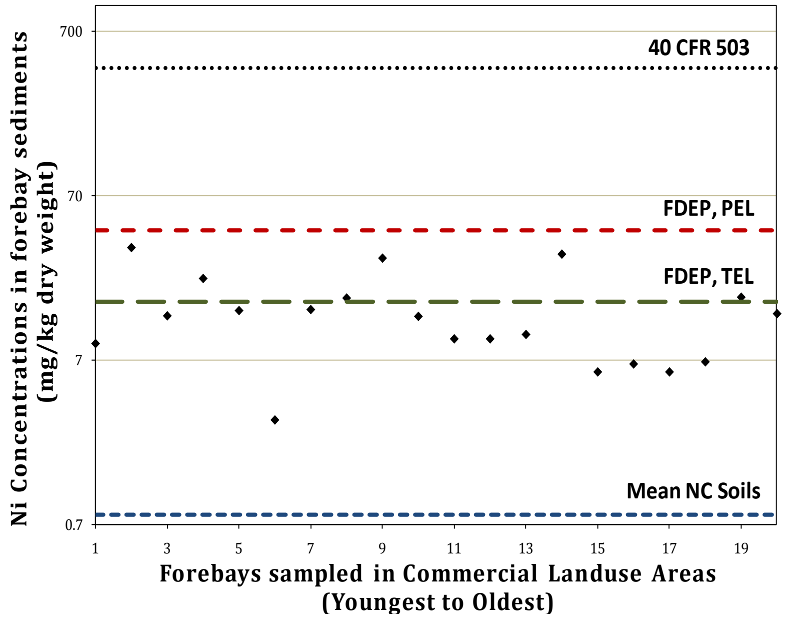


Figure H.1. Ni Concentrations in forebay sediments by Site in Commercial landuses, Youngest to Oldest, as compared to regulatory standards and guidelines.

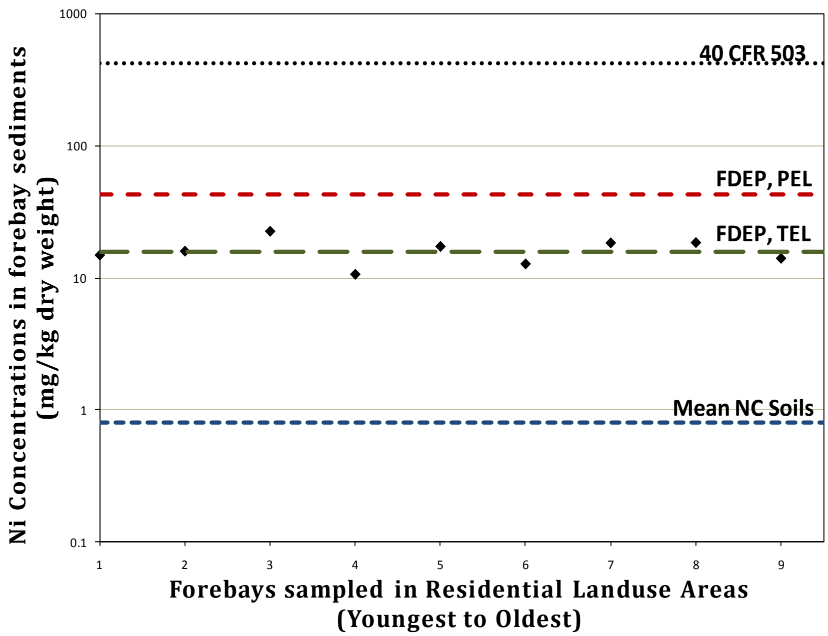


Figure H.2. Ni Concentrations in forebay sediments by Site in Residential landuses, Youngest to Oldest, as compared to regulatory standards and guidelines.

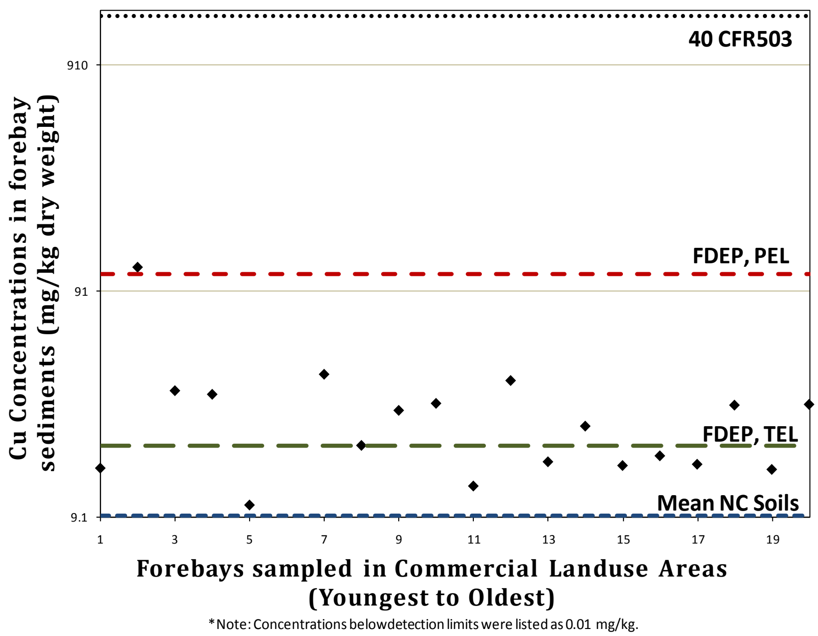


Figure H.3. Cu Concentrations in forebay sediments by Site in Commercial landuses, Youngest to Oldest, as compared to regulatory standards and guidelines.

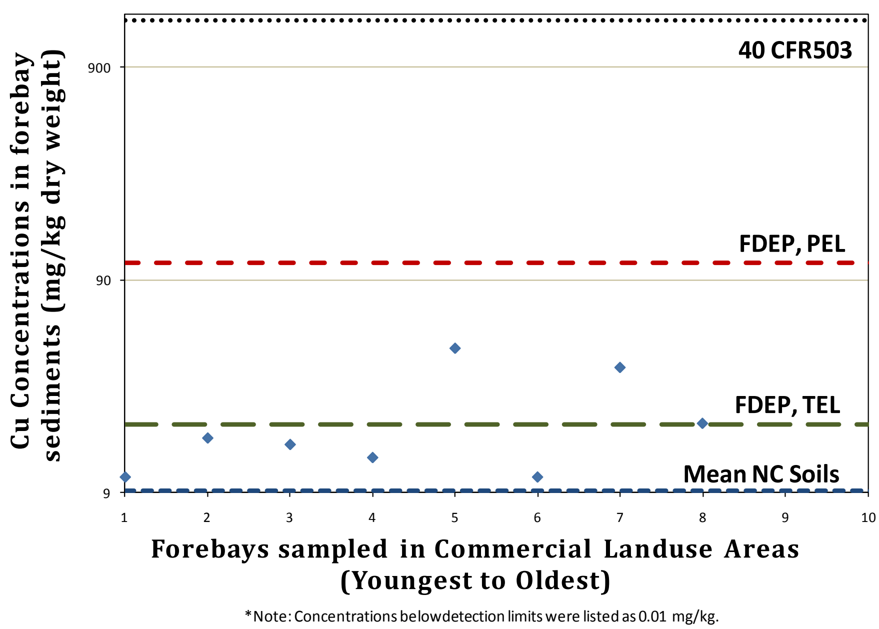


Figure H.4. Cu Concentrations in forebay sediments by Site in Commercial landuses, Youngest to Oldest, as compared to regulatory standards and guidelines.

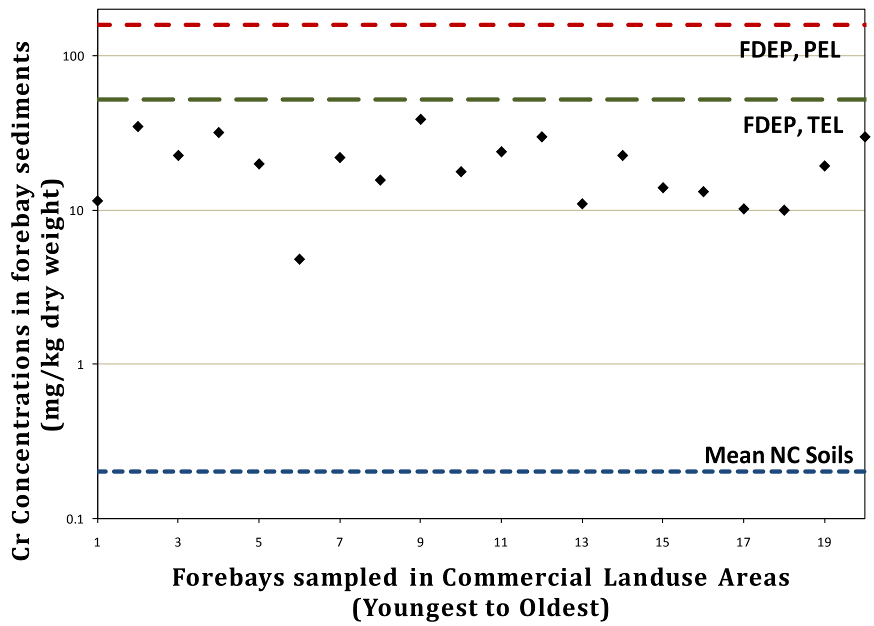


Figure H.5. Cr Concentrations in forebay sediments by Site in Commercial landuses, Youngest to Oldest, as compared to regulatory standards and guidelines.

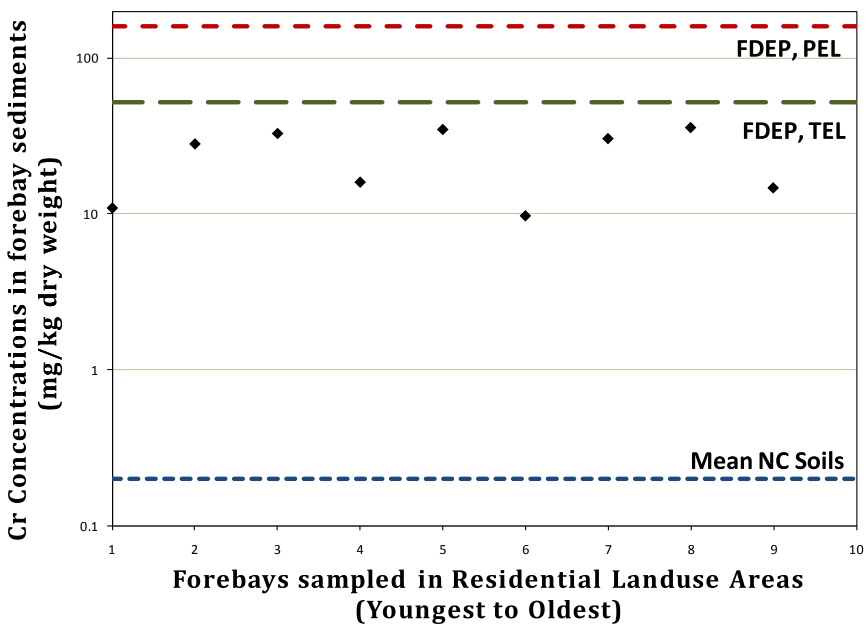


Figure H.6. Cr Concentrations in forebay sediments by Site in Residential landuses, Youngest to Oldest, as compared to regulatory standards and guidelines.

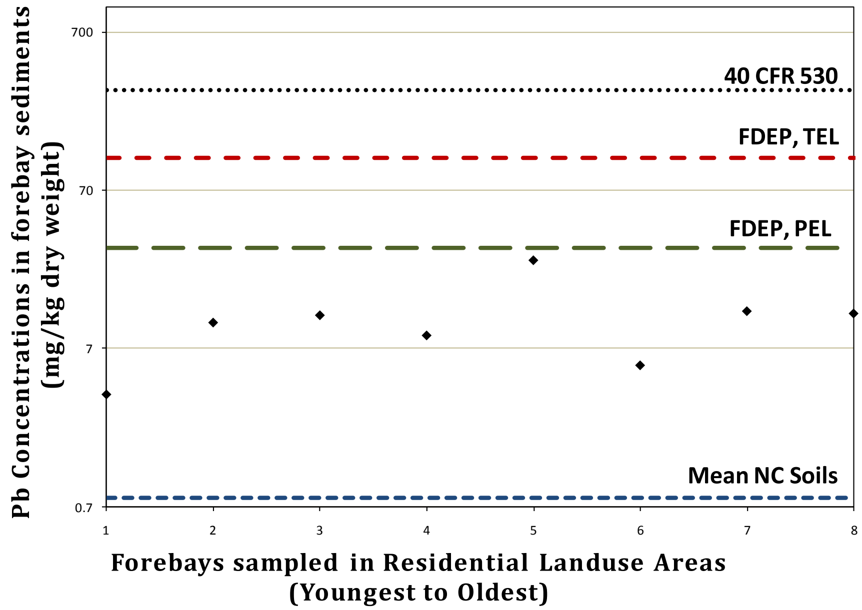


Figure H.7. Pb Concentrations in forebay sediments by Site in Residential landuses, Youngest to Oldest, as compared to regulatory standards and guidelines.

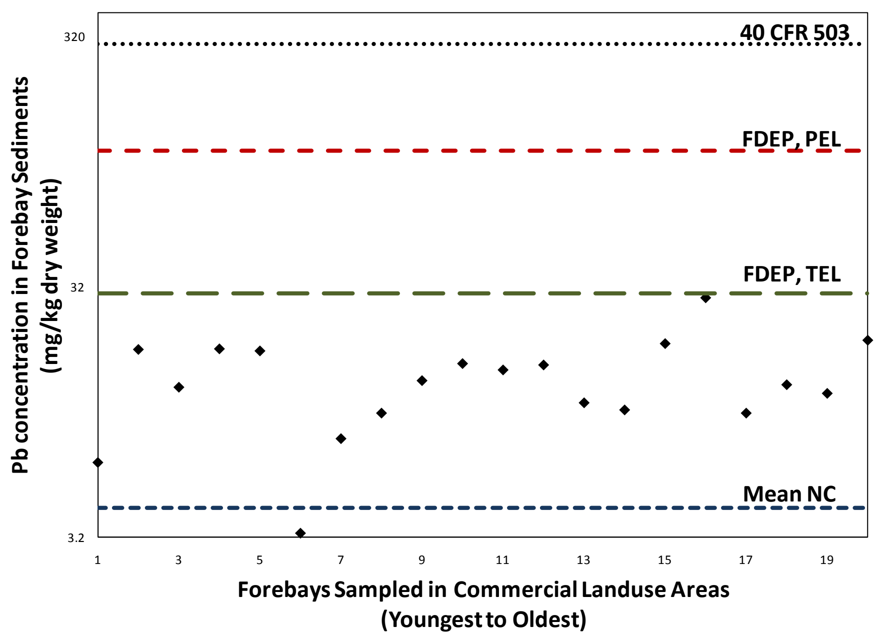


Figure H.8. Pb Concentrations in forebay sediments by Site in Commercial landuses, Youngest to Oldest, as compared to regulatory standards and guidelines.

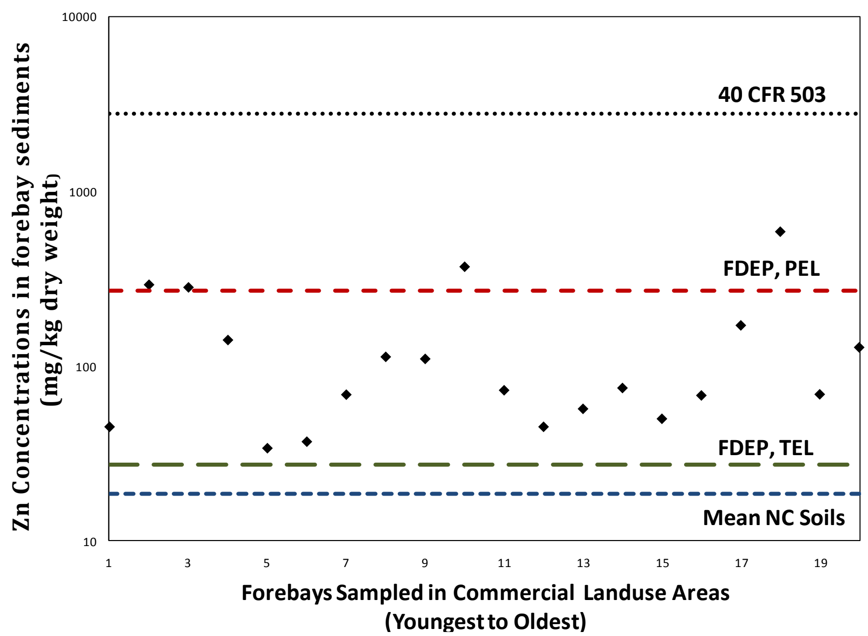


Figure H.9. Zn Concentrations in forebay sediments by Site in Commercial landuses, Youngest to Oldest, as compared to regulatory standards and guidelines.

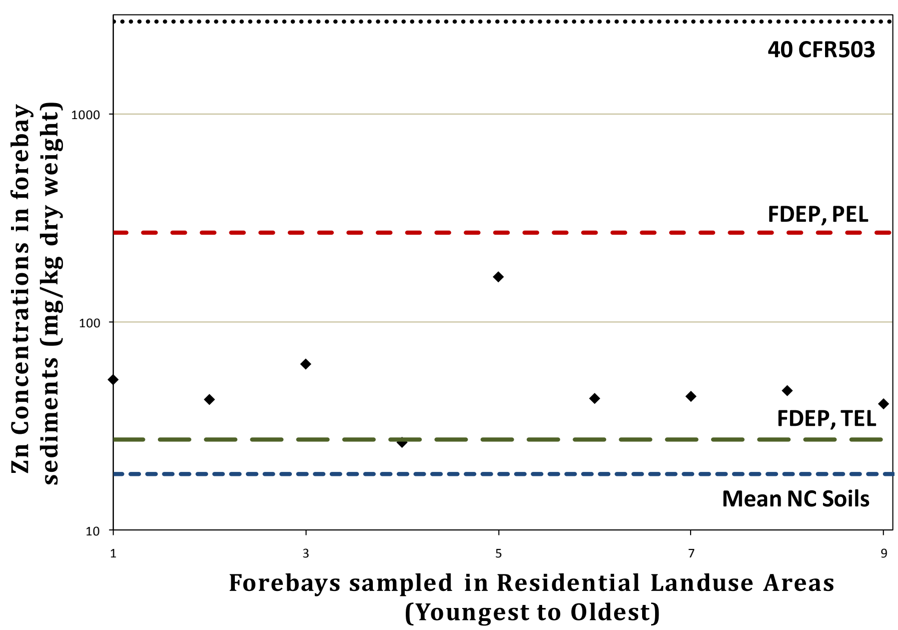


Figure H.10. Zn Concentrations in forebay sediments by Site in Residential landuses, Youngest to Oldest, as compared to regulatory standards and guidelines.

Appendix I: Statistical code and output used in Chapter 4.

The following data pairs were analyzed for significant differences:

1. Metal concentrations in forebay sediments from commercial landuses (COM) versus forebay sediments from residential landuses (RES).

2. Metal concentrations in forebay sediments (IN) versus metal concentrations in sediments taken near outlet structures (OUT).

3. Fraction of sand in forebay sediments (IN) versus fraction of sand in sediments taken near outlet structures (OUT).

All data distributions were checked for normality as described in Appendix C. Again, t-tests were used for normally distributed data and the Kendall tau test of association was used for non-normal data sets. An example code and output of each data pair analysis are provided below. Normalized copper (Cu) data is used as a representative example for a t-test between metal concentrations in forebay sediments and metals concentrations in sediments taken near the outlet structure, and chromium (Cr) data is used as a representative example for anon-parametric analysis of association between metal concentrations in forebay sediments and metal concentrations of sediments taken near outlet structures.

LANDUSE COMPARISON-CODE

```
data LandUse;
input Site$ Landuse$ Cr Cu    Fe    Zn    Ni    Pb;
logCr=log10(Cr);
logCu=log10(Cu);
```

```
logFe=log10(Fe);
logZn=log10(Zn);
logNi=log10(Ni);
logPb=log10(Pb);
cards;
A      COM   17.8  29    13300.00     370   12.9  15.8
B      COM   24    12.5  21400.00     72.8  9.4   14.9
C      COM   22.7  33    26200.00     282   13    12.7
D      COM   32    31.8  32900.00     141   22    18.1
E      COM   22.7  23    19269.00     75    31    10.3
F      COM   19.4  14.8  22127.00     69    16.9  12
G      COM   35    116   29216.00     292   34    18
H      COM   20    10.3  21185.00     33.95 14    17.76
I      COM   33.8  53    26900.00     125   18.4  9.4
J      COM   30    36.59 57800.00     45    9.4   15.6
K      COM   4.8   6.8   14000.00     37    3     3.3
L      COM   13.2  17    26000.00     68    6.6   29
M      COM   10.2  15.6  21600.00     171   5.9   10
N      COM   10    28.48 21900.00     587   6.8   13
O      COM   22    39    37300.00     68.7  14.2  7.9
P      COM   30    28.7  42300.00     128   13.4  19.6
Q      COM   14    15.4  11100.00     50    5.9   19
R      COM   11.5  15    9700.00      45    8.8   6.34
S      COM   15.7  18.9  27800.00     113   16.7  10
T      COM   39    27    24600.00     110   29.3  13.5
U      COM   11    16    12000.00     57    10    11
V      RES   28.3  16.2  25500.00     42.5  15.85 10.25
W      RES   30.6  34.9  36100.00     44    18.3  12.1
X      RES   36    19    36100.00     46.9  18.4  11.7
Y      RES   16.1  13.1  22500.00     26.4  10.6  8.5
Z      RES   35    43    27900.00     166   17.2  25.4
AB     RES   14.8  5.1   9400.00      40.5  14    22.3
CD     RES   33    15.1  23300.00     63    22.4  11.4
EF     RES   11    10.6  10400.00     53    14.8  3.6
GH     RES   9.8   10.6  10220.00     43    12.7  5.5
;

options formdlim = "_";
proc print data= LandUse;
run;

data c; set LandUse;
ods select BasicMeasures TestsForLocation GoodnessOfFit;

proc univariate data =c;
var Cr Cu    Fe    Zn    Ni    Pb logCr LogCu logFe logZn logNi logPb;
histogram/normal;
proc univariate plot normal;
histogram Cr/normal;
histogram Cu/normal;
histogram Fe/normal;
histogram Zn/normal;
```

```
histogram Ni/normal;
histogram Pb/normal;
proc univariate plot normal;
histogram logCr/normal;
histogram logCu/normal;
histogram logFe/normal;
histogram logZn/normal;
histogram logNi/normal;
histogram logPb/normal;
run;
proc ttest data=LandUse;
class Landuse;
var Cr;
run;
proc ttest data=LandUse;
class Landuse;
var Fe;
run;
proc ttest data=LandUse;
class Landuse;
var Ni;
run;
proc ttest data=LandUse;
class Landuse;
var Pb;
run;
proc ttest data=LandUse;
class Landuse;
var logCu;
run;

data NonParLand_ZN;
input COM RES;
datalines;
370   42.5
72.8  44
282   46.9
141   26.4
75    166
69    40.5
292   63
33.95 53
125   43
45    .
37    .
68    .
171   .
587   .
68.7  .
128   .
50    .
45    .
```

```
113   .
110   .
57    .
;
run;
```

LANDUSE COMPARISON-OUTPUT

Output for a landuse t-test, using Cr for an example is provided after Table I.1.

Table I.1. P-values testing for a significant pollutant differences between residential and commercial landuses

Pollutant	** P-Value
*Cd	-
Cr	0.49
log (Cu)	0.15
Fe	0.60
Ni	0.58
Pb	0.62
** Zn	-0.06

* Cd was not detected in almost all cases, thus a landuse comparison was not possible. ** Indicates significant difference between commercial and residential land uses.
** indicates tau value of association. Do not mistake for p-value.

```
                                 The SAS System         23:02 Tuesday, March 9, 2010
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                               The TTEST Procedure

                                   Statistics

                            Lower CL            Upper CL  Lower CL            Upper CL
Variable  Landuse         N     Mean     Mean      Mean   Std Dev   Std Dev   Std Dev   Std Err

Cr        COM            21   16.548   20.895    25.243    7.3071     9.551    13.792    2.0842
Cr        RES             9   15.577   23.844    32.112    7.2652    10.756    20.606    3.5853
Cr        Diff (1-2)          -11.04   -2.949    5.1386    7.8646    9.9103    13.403    3.9483


                                    T-Tests

             Variable    Method           Variances      DF    t Value    Pr > |t|

             Cr          Pooled           Equal          28      -0.75      0.4613
             Cr          Satterthwaite    Unequal      13.7      -0.71      0.4889
```

```
                    Equality of Variances

Variable    Method      Num DF    Den DF    F Value    Pr > F

Cr          Folded F         8        20       1.27    0.6272
```

SPATIAL ANALYSIS (IN VS. OUT) - CODE

```
data MetalsTtest;
input Type$ CR    CU    FE    ZN    CD    NI    PB;
lCU=log10(CU);
datalines;
IN     25    12.8  17800 36    .     13.7  8.9
IN     33    15.1  23300 63    .     22.4  11.4
IN     11    10.6  10400 53    .     14.8  3.6
IN     9.8   10.6  10220 43    .     12.7  5.5
IN     22.7  33    26200 282   .     13    12.7
IN     32    31.8  32900 141   .     22    18.1
IN     15.7  18.9  27800 113   .     16.7  10
IN     39    27    24600 110   .     29.3  13.5
IN     14.8  5.1   9400  40.5  .     14    22.3
IN     11    16    12000 57    .     10    11
OUT    30    18.3  21400 42    .     16.4  10.6
OUT    38    15    28300 53    .     22.1  15
OUT    35.2  20.1  26600 56    .     28.2  10.4
OUT    30    14.4  24500 39.4  .     19.8  13.7
OUT    28.5  48.7  31800 430   .     19.93 19.9
OUT    28.5  48.7  31800 430   .     19.93 19.9
OUT    21    14    23400 67.4  .     14.4  11.3
OUT    15.1  9.63  19900 32    .     16.1  7.3
OUT    19.3  11.1  15400 82    .     12.8  21.78
OUT    28    52    26400 278   .     15.2  39

;

proc ttest data=MetalsTtest;
class Type;
var CU;

data NonParCR;
input Type$ IN OUT;
datalines;
CR     8.9   11.4
CR     3.6   5.5
CR     12.7  18.1
CR     10    30
CR     22.3  11
```

```
CR     10.6  15
CR     10.4  13.7
CR     19.9  19.9
CR     11.3  7.3
CR     21.78 39

;
run;
proc corr pearson spearman kendall data=NonParCR;
var IN OUT;
run;
```

SPATIAL ANALYSIS (IN VS. OUT) - OUTPUT

Sediment samples taken near the outlet structures exhibited higher concentrations for every metal sampled, when compared to forebay sediment samples; however, none were significantly higher statistically (Table I.2). An example of SAS output for normalized copper data (t-test) and non-normalized (non-parametric) chromium data is shown following Table I.2.

Table I.2. P-values and tau-associations for differences in forebay metal concentrations and outlet metal concentrations.

Parameter	Distribution	Test	P-value or Tau (τ)
Cu	normal	Satterthwaite T-test	0.27
Zn	normal	Pooled T-test	0.33
Pb	normal	Satterthwaite T-test	0.14
Fe	log-transformed normal	Pooled T-test	0.07
Cr	non-normal	Kendall Tau	*0.33
Ni	non-normal	Kendall Tau	*0.13

* indicates tau value for association. Do not mistake for p-value.

The TTEST Procedure

Statistics

Variable	Type	N	Lower CL Mean	Mean	Upper CL Mean	Lower CL Std Dev	Std Dev	Upper CL Std Dev	Std Err
CU	IN	10	11.293	18.09	24.887	6.5356	9.5017	17.346	3.0047
CU	OUT	10	12.84	25.193	37.546	11.878	17.268	31.525	5.4607
CU	Diff (1-2)		-20.2	-7.103	5.9916	10.531	13.937	20.61	6.2328

T-Tests

Variable	Method	Variances	DF	t Value	Pr > |t|
CU	Pooled	Equal	18	-1.14	0.2694
CU	Satterthwaite	Unequal	14	-1.14	**0.2736**

Equality of Variances

Variable	Method	Num DF	Den DF	F Value	Pr > F
CU	Folded F	9	9	3.30	0.0898

The UNIVARIATE Procedure
Variable: Cr

Moments

N	30	Sum Weights	30
Mean	21.78	Sum Observations	653.4
Std Deviation	9.83443633	Variance	96.7161379
Skewness	0.13900191	Kurtosis	-1.3063495
Uncorrected SS	17035.82	Corrected SS	2804.768
Coeff Variation	45.1535185	Std Error Mean	1.79551421

Basic Statistical Measures

Location		Variability	
Mean	21.78000	Std Deviation	9.83444
Median	21.00000	Variance	96.71614
Mode	11.00000	Range	34.20000
		Interquartile Range	17.40000

NOTE: The mode displayed is the smallest of 4 modes with a count of 2.

Tests for Location: Mu0=0

Test	-Statistic-		-----p Value------	
Student's t	t	12.13023	Pr > |t|	<.0001
Sign	M	15	Pr >= |M|	<.0001
Signed Rank	S	232.5	Pr >= |S|	<.0001

Tests for Normality

Test	--Statistic---		-----p Value------	
Shapiro-Wilk	W	0.939167	Pr < W	0.0864
Kolmogorov-Smirnov	D	0.13171	Pr > D	>0.1500
Cramer-von Mises	W-Sq	0.104815	Pr > W-Sq	**0.0942**
Anderson-Darling	A-Sq	0.669179	Pr > A-Sq	0.0766

Quantiles (Definition 5)

Quantile	Estimate
100% Max	39.0
99%	39.0
95%	36.0
90%	35.0

The UNIVARIATE Procedure
Variable: Cr

Quantiles (Definition 5)

Quantile	Estimate
75% Q3	30.6
50% Median	21.0
25% Q1	13.2
10%	10.1
5%	9.8
1%	4.8
0% Min	4.8

Extreme Observations

```
        ----Lowest----        ----Highest---

        Value      Obs        Value      Obs

          4.8       11         33.8        9
          9.8       30         35.0        7
         10.0       14         35.0       26
         10.2       13         36.0       24
         11.0       29         39.0       20


  Stem Leaf                     #             Boxplot
     3 5569                     4                |
     3 001234                   6             +-----+
     2 8                        1             |     |
     2 02334                    5             *--+--*
     1 56689                    5             |     |
     1 00011234                 8             +-----+
     0 5                        1                |
     0
       ----+----+----+----+
   Multiply Stem.Leaf by 10**+1
```

```
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                                   The UNIVARIATE Procedure
                                        Variable:  Cr

                                        Normal Probability Plot
                        37.5+                                       * *+*++ *
                            |                                ********+++
                            |                               *++++
                            |                          ++**+*
                            |                     ++****
                            |             * *+*+****
                            |          +*++++
                         2.5+     ++*++
                             +----+----+----+----+----+----+----+----+----+----+
                                 -2        -1         0        +1        +2
```

```
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```

The UNIVARIATE Procedure
Fitted Distribution for Cr

Parameters for Normal Distribution

Parameter	Symbol	Estimate
Mean	Mu	21.78
Std Dev	Sigma	9.834436

Goodness-of-Fit Tests for Normal Distribution

Test	---Statistic----		-----p Value-----	
Kolmogorov-Smirnov	D	0.13171044	Pr > D	>0.150
Cramer-von Mises	W-Sq	0.10481453	Pr > W-Sq	**0.094**
Anderson-Darling	A-Sq	0.66917904	Pr > A-Sq	0.077

Quantiles for Normal Distribution

	-------Quantile------	
Percent	Observed	Estimated
1.0	4.80000	-1.09832
5.0	9.80000	5.60379
10.0	10.10000	9.17666
25.0	13.20000	15.14677
50.0	21.00000	21.78000
75.0	30.60000	28.41323
90.0	35.00000	34.38334
95.0	36.00000	37.95621
99.0	39.00000	44.65832

The CORR Procedure

2 Variables: IN OUT

Simple Statistics

Variable	N	Mean	Std Dev	Median	Minimum	Maximum
IN	10	13.14800	6.15214	10.95000	3.60000	22.30000
OUT	10	17.09000	10.38765	14.35000	5.50000	39.00000

Pearson Correlation Coefficients, N = 10

```
                    Prob > |r| under H0: Rho=0

                              IN            OUT

                 IN        1.00000        0.48480
                                           0.1556

                 OUT       0.48480        1.00000
                            0.1556


             Spearman Correlation Coefficients, N = 10
                     Prob > |r| under H0: Rho=0

                              IN            OUT

                 IN        1.00000        0.34545
                                           0.3282

                 OUT       0.34545        1.00000
                            0.3282


          Kendall Tau b Correlation Coefficients, N = 10
                    Prob > |r| under H0: Rho=0

                              IN            OUT

                 IN        1.00000        0.33333
                                           0.1797

                 OUT       0.33333        1.00000
                                0.1797
```

SPATIAL VARIATION IN PARTICLE SIZE- CODE

```
data SpatialVar;
input Site$ Type$ Sand Silt Clay;
logSand=log10(Sand);
logSilt=log10(Silt);
logClay=log10(Clay);
cards;
GH     IN     86     10     5
T      IN     55     27     18
U      IN     92     3      4
V      IN     57     28     15
AB     IN     90     6      4
CD     IN     42     37     21
EF     IN     87     10     3
A      IN     57     22     21
B      IN     34     36     30
```

```
R      IN    66    24    10
GH     OUT   46.88951011 33.01185759 20.05197673
T      OUT   77.04613625 13.66173501 9.292128745
U      OUT   38.9180189  48.29476698 12.78721411
V      OUT   41.03642814 38.59377694 20.3225099
AB     OUT   51.03466873 32.81768662 16.11016063
CD     OUT   27.67625927 41.35773142 30.89412696
EF     OUT   37.0838466  40.54658037 22.36957303
A      OUT   28.77789685 41.81953496 29.40256819
B      OUT   28.77789685 41.81953496 29.40256819
R      OUT   29.00765404 45.99681697 24.99552899

run;
options formdlim = "_";
proc print data= SpatialVar;
run;
data c; set SpatialVar;
ods select BasicMeasures TestsForLocation GoodnessOfFit;
proc univariate data =c;
var Sand Silt Clay;
histogram/normal;
proc univariate plot normal;
histogram Sand/normal;
histogram Silt/normal;
histogram Clay/normal;
proc univariate plot normal;
histogram logSand/normal;
histogram logSilt/normal;
histogram logClay/normal;
run;
```

SPATIAL VARIATION IN PARTICLE SIZE- OUTPUT

```
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```

Obs	Site	Type	Sand	Silt	Clay	logSand	logSilt	logClay
1	GH	IN	86.0000	10.0000	5.0000	1.93450	1.00000	0.69897
2	T	IN	55.0000	27.0000	18.0000	1.74036	1.43136	1.25527
3	U	IN	92.0000	3.0000	4.0000	1.96379	0.47712	0.60206
4	V	IN	57.0000	28.0000	15.0000	1.75587	1.44716	1.17609
5	AB	IN	90.0000	6.0000	4.0000	1.95424	0.77815	0.60206
6	CD	IN	42.0000	37.0000	21.0000	1.62325	1.56820	1.32222
7	EF	IN	87.0000	10.0000	3.0000	1.93952	1.00000	0.47712
8	A	IN	57.0000	22.0000	21.0000	1.75587	1.34242	1.32222
9	B	IN	34.0000	36.0000	30.0000	1.53148	1.55630	1.47712
10	R	IN	66.0000	24.0000	10.0000	1.81954	1.38021	1.00000

11	GH	OUT	46.8895	33.0119	20.0520	1.67108	1.51867	1.30216
12	T	OUT	77.0461	13.6617	9.2921	1.88675	1.13551	0.96812
13	U	OUT	38.9180	48.2948	12.7872	1.59015	1.68390	1.10678
14	V	OUT	41.0364	38.5938	20.3225	1.61317	1.58652	1.30798
15	AB	OUT	51.0347	32.8177	16.1102	1.70787	1.51611	1.20710
16	CD	OUT	27.6763	41.3577	30.8941	1.44211	1.61656	1.48988
17	EF	OUT	37.0838	40.5466	22.3696	1.56918	1.60795	1.34966
18	A	OUT	28.7779	41.8195	29.4026	1.45906	1.62138	1.46839
19	B	OUT	28.7779	41.8195	29.4026	1.45906	1.62138	1.46839
20	R	OUT	29.0077	45.9968	24.9955	1.46251	1.66273	1.39786

The UNIVARIATE Procedure
Variable: Sand

Basic Statistical Measures

Location		Variability	
Mean	53.61242	Std Deviation	22.26517
Median	48.96209	Variance	495.73760
Mode	28.77790	Range	64.32374
		Interquartile Range	35.98114

NOTE: The mode displayed is the smallest of 2 modes with a count of 2.

Tests for Location: Mu0=0

Test	-Statistic-		-----p Value------	
Student's t	t	10.76848	Pr > |t|	<.0001
Sign	M	10	Pr >= |M|	<.0001
Signed Rank	S	105	Pr >= |S|	<.0001

The UNIVARIATE Procedure
Fitted Distribution for Sand

Goodness-of-Fit Tests for Normal Distribution

Test	---Statistic----		-----p Value-----	
Kolmogorov-Smirnov	D	0.14900845	Pr > D	>0.150

Cramer-von Mises	W-Sq	0.10488850	Pr > W-Sq	**0.092**
Anderson-Darling	A-Sq	0.72575566	Pr > A-Sq	0.049

The UNIVARIATE Procedure
Variable: Silt

Basic Statistical Measures

Location		Variability	
Mean	29.04600	Std Deviation	14.05408
Median	32.91477	Variance	197.51727
Mode	10.00000	Range	45.29477
		Interquartile Range	23.12129

NOTE: The mode displayed is the smallest of 2 modes with a count of 2.

Tests for Location: Mu0=0

Test	-Statistic-		-----p Value------	
Student's t	t	9.242699	Pr > |t|	<.0001
Sign	M	10	Pr >= |M|	<.0001
Signed Rank	S	105	Pr >= |S|	<.0001

The UNIVARIATE Procedure
Fitted Distribution for Silt

Goodness-of-Fit Tests for Normal Distribution

Test	---Statistic----		-----p Value-----	
Kolmogorov-Smirnov	D	0.15579249	Pr > D	>0.150
Cramer-von Mises	W-Sq	0.09625105	Pr > W-Sq	**0.121**
Anderson-Darling	A-Sq	0.60542701	Pr > A-Sq	0.100

```
                              The UNIVARIATE Procedure
                                 Variable:  Clay

                             Basic Statistical Measures

                   Location                    Variability

               Mean     17.33142     Std Deviation            9.24111
               Median   19.02599     Variance                85.39803
               Mode      4.00000     Range                   27.89413
                                     Interquartile Range     14.03649

       NOTE: The mode displayed is the smallest of 3 modes with a count of 2.


                             Tests for Location: Mu0=0

                  Test           -Statistic-    -----p Value------

                  Student's t    t  8.387358    Pr > |t|    <.0001
                  Sign           M        10    Pr >= |M|   <.0001
                  Signed Rank    S       105    Pr >= |S|   <.0001
```

```
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                              The UNIVARIATE Procedure
                             Fitted Distribution for Clay

                      Goodness-of-Fit Tests for Normal Distribution

               Test                  ---Statistic----   -----p Value-----

               Kolmogorov-Smirnov    D     0.11577294   Pr > D     >0.150
               Cramer-von Mises      W-Sq  0.04644654   Pr > W-Sq  >0.250
               Anderson-Darling      A-Sq  0.38470594   Pr > A-Sq  >0.250
```

```
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                                The TTEST Procedure

                                    Statistics

                             Lower CL          Upper CL  Lower CL           Upper CL
Variable  Type            N      Mean     Mean      Mean   Std Dev  Std Dev   Std Dev  Std Err

Sand      IN             10    51.571     66.6    81.629    14.451   21.009    38.354   6.6436
Sand      OUT            10    29.768   40.625    51.482    10.439   15.177    27.707   4.7994
```

Sand	Diff (1-2)		8.7563	25.975	43.194	13.848	18.326	27.102	8.1958

T-Tests

Variable	Method	Variances	DF	t Value	Pr > |t|
Sand	Pooled	Equal	18	3.17	0.0053
Sand	Satterthwaite	Unequal	16.4	3.17	**0.0058**

Equality of Variances

Variable	Method	Num DF	Den DF	F Value	Pr > F
Sand	Folded F	9	9	1.92	0.3468

The TTEST Procedure

Statistics

Variable	Type	N	Lower CL Mean	Mean	Upper CL Mean	Lower CL Std Dev	Std Dev	Upper CL Std Dev	Std Err
Silt	IN	10	11.5	20.3	29.1	8.4619	12.302	22.459	3.8903
Silt	OUT	10	30.796	37.792	44.788	6.7272	9.7802	17.855	3.0928
Silt	Diff (1-2)		-27.93	-17.49	-7.051	8.3971	11.113	16.434	4.9699

T-Tests

Variable	Method	Variances	DF	t Value	Pr > |t|
Silt	Pooled	Equal	18	-3.52	0.0024
Silt	Satterthwaite	Unequal	17.1	-3.52	**0.0026**

Equality of Variances

Variable	Method	Num DF	Den DF	F Value	Pr > F
Silt	Folded F	9	9	1.58	0.5050

The TTEST Procedure

Statistics

Variable	Type	N	Lower CL Mean	Mean	Upper CL Mean	Lower CL Std Dev	Std Dev	Upper CL Std Dev	Std Err
Clay	IN	10	6.4365	13.1	19.763	6.4071	9.3149	17.005	2.9456
Clay	OUT	10	16.319	21.563	26.806	5.0419	7.33	13.382	2.318
Clay	Diff (1-2)		-16.34	-8.463	-0.588	6.3331	8.3814	12.395	3.7483

T-Tests

Variable	Method	Variances	DF	t Value	Pr > |t|
Clay	Pooled	Equal	18	-2.26	0.0366
Clay	Satterthwaite	Unequal	17.1	-2.26	**0.0374**

Equality of Variances

Variable	Method	Num DF	Den DF	F Value	Pr > F
Clay	Folded F	9	9	1.61	0.4864

Appendix J: Impact of System Age on Metal Concentrations in Forebay Sediments

Raw data indicated that age has no influence over the magnitude of metal concentrations in forebays. Graphs of age versus metal concentrations are shown in Figures J.1 through J.6. From a maintenance standpoint, the relationship between age and metal concentrations is important. Many current recommendations for deposited soil removal are based on time increments (others are volume-based). Age-based recommendations do not appear adequate. Perhaps stormwater wetlands and wetponds need to be monitored and sampled more frequently, to ensure excavation of sediments prior to violations of aquatic thresholds. If aquatic health protection becomes a driver for sediment removal, frequent sediment monitoring of stormwater BMPs is strongly encouraged.

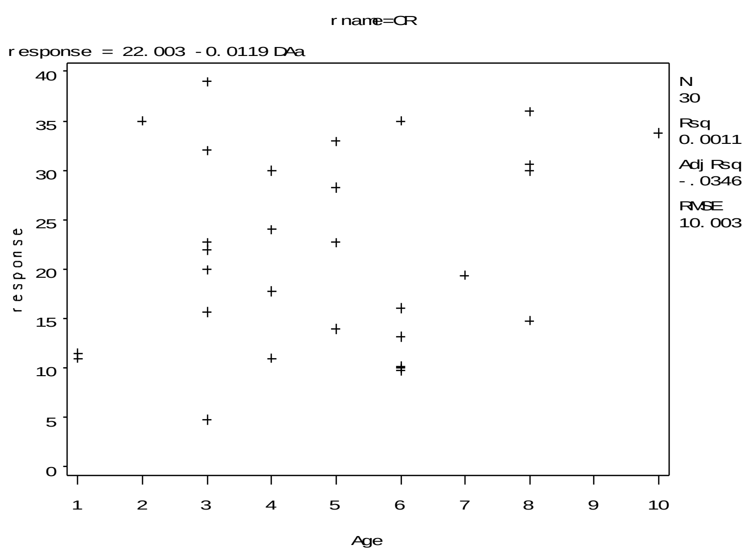


Figure J.1. Concentration of Chromium (Cr) at sites sampled. Age of facility appears to be a non-factor.

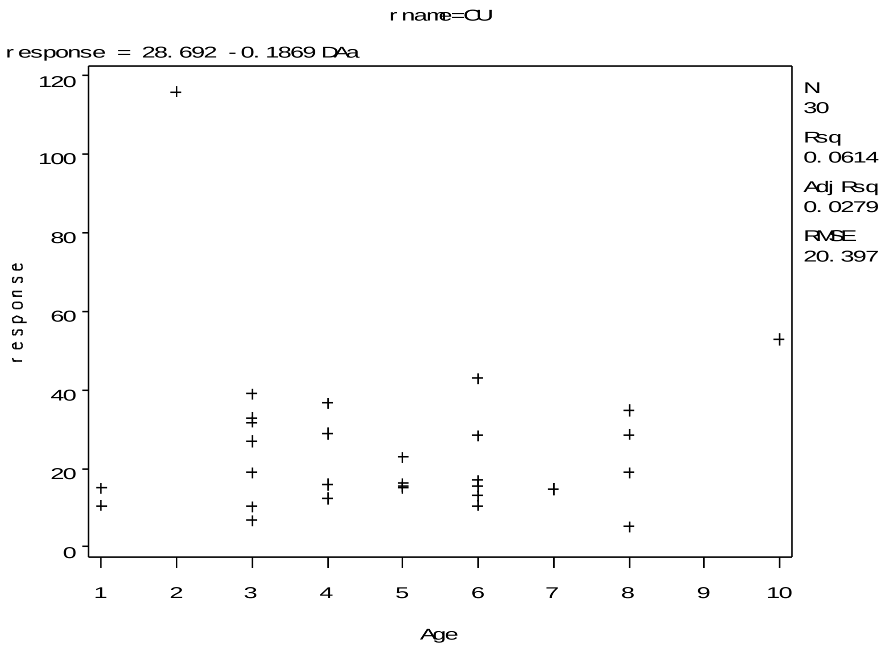


Figure J.2. Concentration of Copper (Cu) at sites sampled. Age of facility appears to be a non-factor.

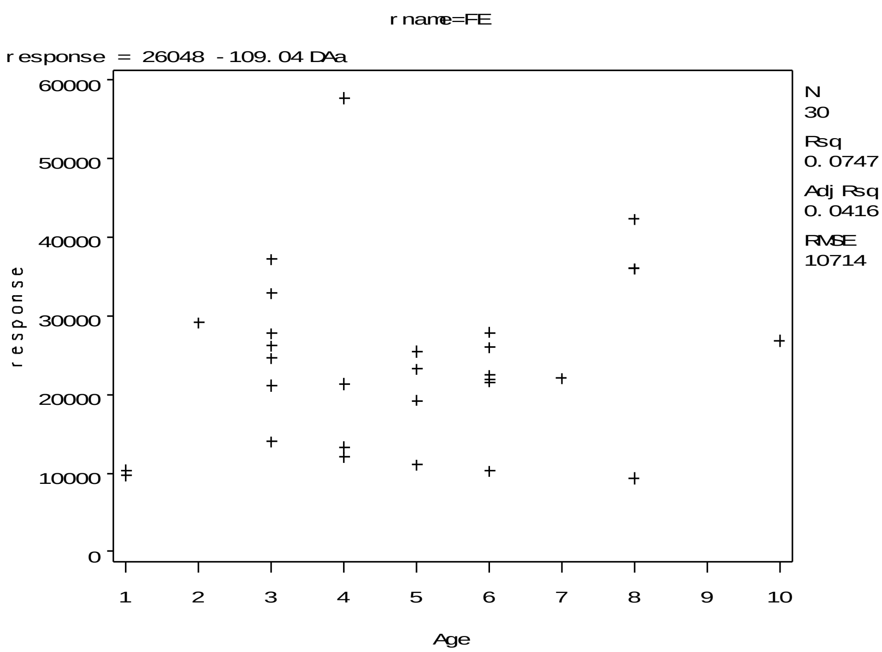


Figure J.3. Concentration of Iron (Fe) at sites sampled. Age of facility appears to be a non-factor.

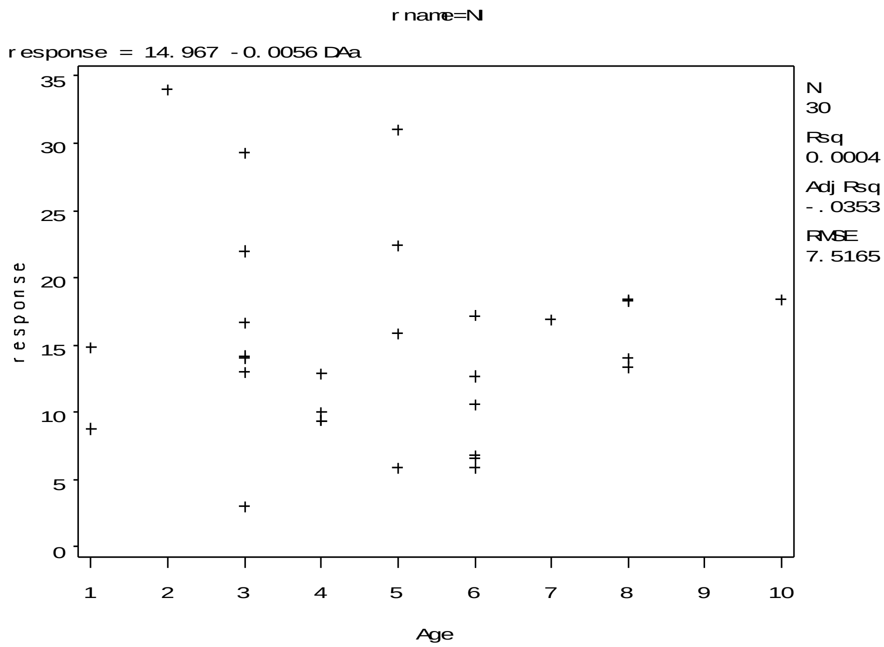


Figure J.4. Concentration of Nickel (Ni) at sites sampled. Age of facility appears to be a non-factor.

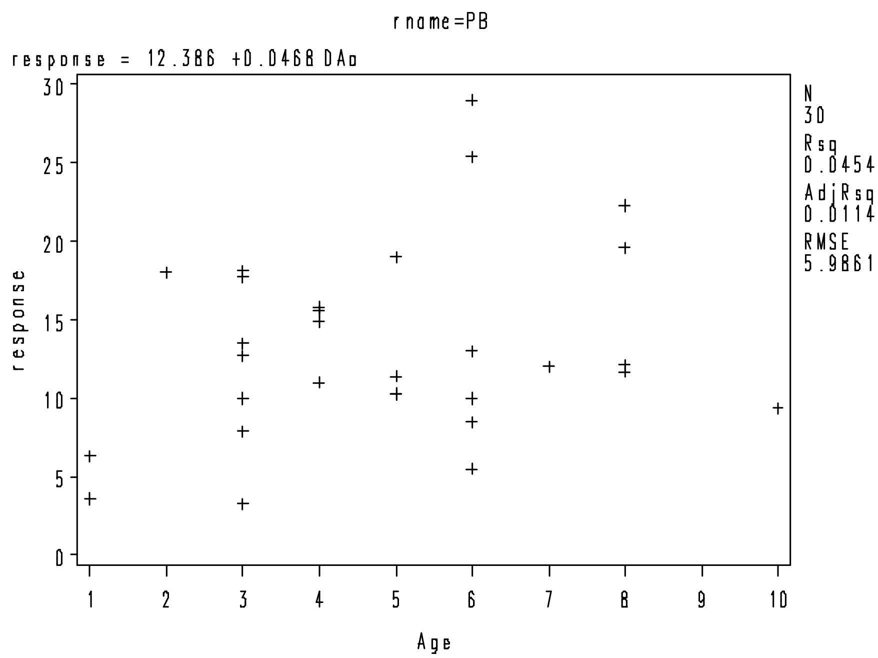


Figure J.5. Concentration of lead (Pb) at sites sampled. Age of facility appears to be a non-factor.

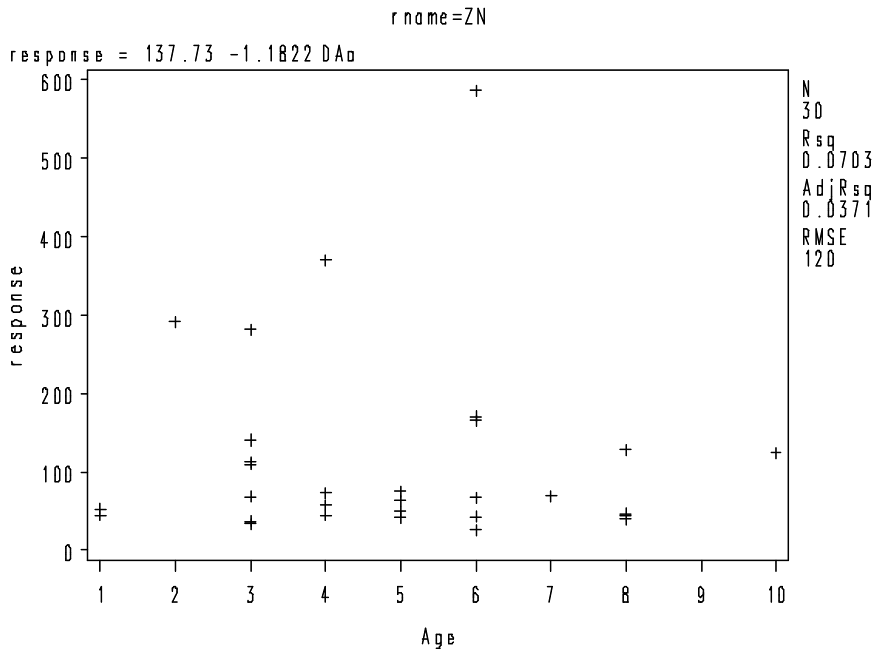


Figure J.6. Concentration of zinc (Zn) at sites sampled. Age of facility appears to be a non-factor.

Appendix K: Particle Size Analysis

Ten forebay sediment samples and ten sediment samples taken near the outlet structures were analyzed for particle size, using the Hydrometer method (Gee and Bauder, 1986). All samples were oven dried for no less than 24 hours in small metal cups (Figure K.1). Once samples were cooled, each was separately ground using a mortar and pestle (Figure K.1). In between each sample, the mortar and pestle was swept out using a small brush. Each sample was poured through a No. 10 (2 mm) sieve to remove gravel, as sediments were only analyzed for silt, sand, and clay portions (Figure K.1).

a.

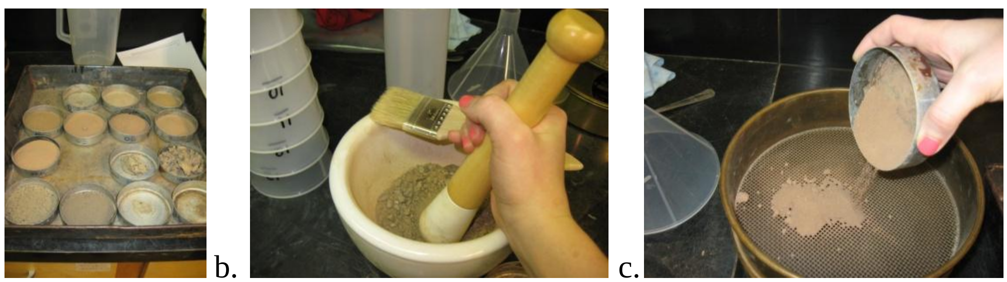


Figure K.1. (a) samples after oven drying, (b) grinding of samples using mortar and pestle, and (c) sieving of samples to remove gravel portion.

The thoroughly ground and sieved samples were then individually weighed using a Mettler AE200 precision weighing scale (Figure K.2), and transferred into 500 ml plastic beakers (Figure K.2). Fifty mL of 10% sodium metaphosphate solution was added to each sample, and then just enough deionized water was added to each so that the beakers were filled to the 300 mL marker-line (~ 250 ml of H_2O). All samples were covered with paraffin and left overnight to settle (Figure K.2).

a.  b.  c. 

Figure K.2. (a) scale used to weigh samples (b) plastic beakers used for holding samples, and (c) samples covered in parafilm prior to being placed in graduated cylinders for analysis.

Next, samples were shaken for 5 minutes each, using a milkshake-type machine and metal shaker tins (Figure K.3). Drops of Ethyl alcohol were added to samples that appeared "foamy" (which results from the presence of organic matter) after shaking to remove organics from solution. Samples were then transferred to 1-liter graduated cylinder (Figure K.3). Deionized water was used to rinse the metal shakers in between "shaking" runs, and to aid in the removal of samples from the metal shakers (Figure K.3).

a.  b.  c. 

Figure K.3. (a) graduated cylinders used in analysis (b) transferring agitated samples to graduated cyclinders and (c) mixing apparatus used to agitate samples before transferring to graduated cylinders.

Samples were allowed to settle for 24 hours, and then were plunged for 30 seconds using a plastic stirring rod. A hydrometer reading was taken 30 and 90 seconds after plunging. Twenty-four hours later, the samples were plunged again, and hydrometer readings were taken 6 hours afterward. The same procedure was then repeated using a 16 hour time increment.

Stokes Law was used to analyze the hydrometer readings in combination with sample masses to determine the percent sand, silt, and clay of each sample (Gee and Bauder, 1986) (Table K.1.). Then, using the USDA soil classification triangle, each sample was assigned a soil texture (Table K.1 and Figure K.4).

Table K.1. Fraction of sand, silt, and clay in forebay sediments and sediments taken near outlet structures at 10 stormwater wetlands and wetponds.

			Forebay Sediments				**Outlet Sediments**			
Sample ID	**Landuse**	**Age**	Sand **%**	Silt **%**	Clay **%**	USDA **Class.**	Sand **%**	Silt **%**	Clay **%**	USDA **Class.**
R	COM	1	66.4	23.9	9.7	sandy loam	29.0	46.0	25.0	loam
T *	COM	3	54.8	27.0	18.2	sandy loam	77.0	13.7	9.3	sandy loam
U*	COM	4	92.5	3.2	4.3	sand	38.9	48.3	12.8	loam
A	COM	4	57.4	21.7	20.9	sandy clay loam	28.8	41.8	29.4	clay loam
B	COM	4	33.7	35.9	30.4	clay loam	28.8	41.8	29.4	clay loam
EF *	RES	1	86.6	10.1	3.4	sand	37.1	40.5	22.4	loam
V *	RES	5	57.4	28.0	14.6	sandy loam	41.0	38.6	20.3	loam
CD *	RES	5	41.7	37.0	21.3	loam	27.7	41.4	30.9	clay loam
GH	RES	6	85.6	9.6	4.7	loamy sand	46.9	33.0	20.1	loam
AB*	RES	8	90.2	5.8	3.9	sand	51.0	32.8	16.1	loam

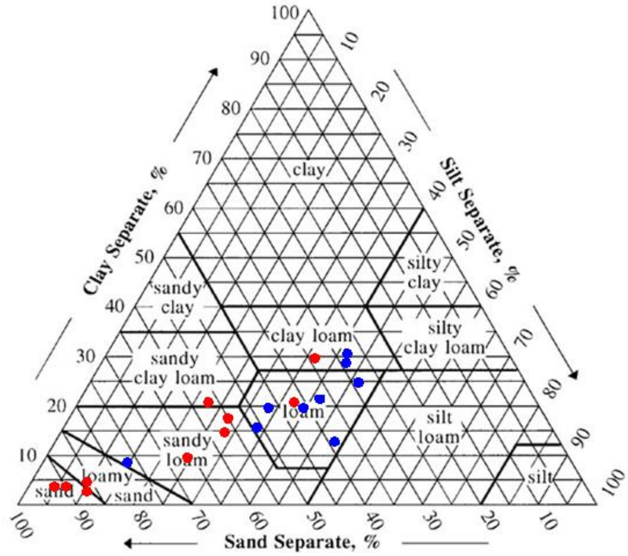


Figure K.4. Soil Classification Tool showing soil classifications of all sites analyzed (blue indicates outlet sediments, red indicates forebay sediments).
(http://soils.usda.gov/technical/aids/investigations/texture/)

iV

Soil Texture Analysis The Hydrometer Method

1. Weigh 40 grams (60 grams if course texture) of air-dried soil. That has been ground to pass through a 2mm sieve into a 400ml beaker.
2. Add 50ml of 10% Hexametaphosphate solution (dispersing Agent) and approximately 250ml of water.
3. Let stand for 30 minutes, stirring at 15-minute intervals. overnight
4. Transfer sample from beaker to metal mixing cup and mix for 5 minutes on milkshake mixer.
5. Transfer this solution to a 1-liter hydrometer jar and complete to volume.
6. The sample is now ready for the hydrometer measurements.
7. Four measurements will be taken at the following times:
   0.5 minutes
   1.5 minutes
   360 minutes (6 hours)
   960 minutes (16 hours)
8. The sample is mixed for 30 seconds with a brass plunger. A few seconds before the first reading the hydrometer is slowly inserted into the hydrometer jar and a measurement taken at the 0.5 minute. Let the hydrometer stay in the solution until the 1.5 reading is finished. After this remove the hydrometer, stir the sample again for 30 seconds and wait until the 360-minute reading. Slowly insert the hydrometer a few seconds prior to the 360-minute. This is repeated for the 960-minute reading.

   1/2 1 1/2
   6
   16
   6
9. A blank solution with the same amount of dispersing agent will have to be prepared to correct for the weight of the dispersing agent will have to be prepared to correct for the Weight of the dispersing agent. (RL in the procedure)
10. Input: Each record should contain the following information:

Columns	Data
1-6	Sample I.D. (May use numbers and symbols)
7-8	Soil weight (to nearest gram)
9-10	Temperature in degrees C at 0.5 minutes (to nearest degree)
11-12	RL at 0.5 minutes (to nearest half)
13-15	R at 0.5 minutes (to nearest half)
16-17	Temperature at 1.5 minutes
18-19	RL at 1.5 minutes
20-22	R at 1.5 minutes
23-24	Temperature at 360 minutes
25-26	RL at 360 minutes
27-29	R at 360 minutes
30-31	Temperature at 960 minutes
32-33	RL at 960 minutes
34-36	R at 960 minutes
37-39	Per cent moisture by weight (to nearest tenth %) If moisture is <1% leave blank

All input data should be written without decimal points. Data should be saved as a "Text" file in Microsoft Word or another word processing program.

Figure K.5. Sheet used for guidance in sampling, courtesy of NCSU Soil Science Department.

Appendix L: Plant Growth Potential.

Table L.1. Analytical Testing Procedures and Instruments of Analysis.

Metal	Form	Analytical Testing Procedure	Analysis Instrumentation
Cr, Cu, Fe, and Zn	Total	EPA Method 3050B, strong acid digestion	ICP-OES
Pb, Cd, and Ni	Total	EPA Method 3050B, strong acid digestion	ICP-MS
Cr, Cu, Fe, and Zn	Plant Available	extraction with weak acid (0.05 N + 0.025 N H2S04)	direct aspiration atomic absorption spectroscopy
Pb, Cd, and Ni	Plant Available	extraction with weak acid (0.05 N + 0.025 N H2S04)	direct aspiration atomic absorption spectroscopy
TKN	Total	EPA Manual 351.2 (1979)	Automated
$NO_3N + N0_2$-N	Total	EPA Method 353.2 (1979)	Automated
TP	Total	EPA Method 365.4 (1979)	Automated

ICP-OES = Inductively Coupled Plasma Optical Emission Spectroscopy
ICP- MS= Inductively Coupled Mass Spectroscopy

While the toxicity of metals in sediment was not problematic by 40 CFR 530 standards, it is important to evaluate the potential nutrient availability for plant uptake and consequent growth. If plants cannot establish on applied sediments, stabilization will not occur and spoils, or excavated sediments, will erode, creating toxicity problems elsewhere. To assess nutrient availability, total mass of phosphorus and nitrogen were calculated for each forebay, assuming the theoretical excavation would occur to a depth of 2 feet. The mass of phosphorus and nitrogen per excavation period ranged from 12.5 lbs to 500 lbs and 16.3 lbs to 1600 lbs, respectively (Table L.2).

Table L.2. Theoretical calculated amount of TN and TP removed per excavation period (lbs)

	pH (pre-excavation)	TP (mg/kg) ww	* lbs of TP per excavation period	TN (mg/kg) ww	*lbs of TN per excavation period
MEAN	6.1	169.3	117	447.8	283.4
MEDIAN	6.3	143.8	66	360.1	158.3
MIN	5.2	39.5	12.5	86.9	16.3
MAX	7.2	571.7	500.2	2602	1596.6

* excavation period = time at which forebay sediments are excavated. Time of excavation varies depending on sedimentation rates and individual forebay characteristics. ww= wet weight.

Ideal nitrogen fertilization loads for four low-maintenance grasses (Table L.3) were obtained from Hardy et al. (2009). Recommended phosphorus loads were not provided. All sediments sampled have potential to meet minimum nitrogen fertilization recommendations for all 4 grasses; however, this is dependent upon the dimensions of application including depth and area to be covered. Reasonably, a minimum sediment application depth of 6 inches would allow for the establishment of plants. Overall, authors believe that the amount of nitrogen in forebay sediments will be adequate for grass growth. Still, it is recommended that a soil analysis be completed prior to land-application to identify types and quantities of essential elements of plant growth. At the very least, sediments should be assessed for bioavailable nitrogen, phosphorus, calcium, and potassium, near the time of excavation.

Table L.3. Ideal nitrogen requirements for 4 grasses (Hardy et al., 2009)

Example Crop type	Recommended Fertilizer Rate of N (lbs/acre) for optimal plant growth
Bluegrass (pH=6.0)	100-200
Clover, grass establishment (pH=6.5)	10-30
Fescue/Orchard/Timothy establishment (pH=6.5)	50-70
Switchgrass (pH=5/5)	120-160
(Hardy et al., 2009)	

WET WEIGHT TO DRY WEIGHT EXAMPLE CALCULATION, USING TN FOR SITE A

Forebay Surface Area (SA), = 4356 ft^2
Reported Moisture Content (%MC) = 49.1 %
Reported TN, Dry Weight (WW) = 473 mg/kg

$$Dry\ Weight\ (DW) = WW * \left(\frac{100}{100 - \%MC}\right)$$

$$= 473 * \left(\frac{100}{100 - 49.1}\right) \approx \mathbf{928\ \frac{mg}{kg}\ DW}$$

TN PER EXCAVATION PERIOD, FOR SITE A (SAME METHOD USED FOR TP CALCULATIONS)

Assume 2 ft. excavation depth, therefore:

$$\text{Volume Excavated} = 4346\ ft^2 * 2\ ft$$

$$\approx \mathbf{8700\ ft^3}$$

It was assumed that the forebay sediment was mostly sand particles, therefore a specific gravity (SG) of a sandy, wet-packed soil of 2.68, or 130 lbs per cubic foot was used in calculations.

$$TN, WW * SG * Volume\ excavated = lbs\ of\ TN\ excavated\ from\ Site\ A$$

$$\frac{473\ mg\ TN}{kg\ sediment} * \frac{1\ kg\ sed.}{2.205\ lbs\ sed.} * \frac{130\ lbs\ sed.}{ft^3\ sed.} * \frac{1\ g\ TN}{1000\ mg\ TN} * \frac{1\ kg}{1000g\ TN} * \frac{2.205\ lbs\ TN}{1\ kg\ TN}$$

$$\approx 0.06\ lbs\ TN\ per\ ft^3 sediment\ excavated$$

$$8700\ ft^3 * \frac{0.06\ lbs\ TN}{ft^3 sediment\ excavated} \approx \mathbf{534\ lbs}\ of\ TN\ result\ from\ excavating\ forebay\ A$$

TN PER ACRE OF FOREBAY SURFACE AREA, FOR SITE A:

$$\text{TN per acre SA} = \frac{\text{Total TN in excavated sediments}}{\text{forebay SA}}$$

$$= \frac{534\ \text{lbs TN}}{\frac{4356\ \text{ft}^2}{43{,}560\ \frac{\text{ft}^2}{\text{acre}}}}$$

$$\approx \mathbf{2550} \frac{\textbf{lbs TN}}{\textbf{SA in acres}}$$

AREA POTENTIAL FOR LANDSPREADING OF SEDIMENTS FROM SITE A, USING BLUE GRASS TO SEED:

Blue Grass requires 100-200 lbs of nitrogen per seeded acre. The plant used to seed sediments is dependent on the sediment nutrient content; conversely, the area of land that may be spread with sediments is based on the plant nutrient requirements.

Sediments from site A must be spread on a *minimum* of:

$$\frac{\text{534 lbs TN per excavation period}}{\frac{\text{200 lbs TN}}{\text{acre}}} \approx \boldsymbol{2.7\ acres}$$

Sediments from site A must be spread on a *maximum* of:

$$\frac{\text{534 lbs TN per excavation period}}{\frac{\text{100 lbs TN}}{\text{acre}}} \approx \boldsymbol{5.4\ acres}$$

Similar Calculations were completed with 3 other grass species.

Appendix M: Brief NCSU Site Descriptions (non-confidential sites)

RA-VARS1

This wetland is located on the south-west corner of the Varsity Drive and Capability Drive intersection of NCSU campus near the visitor information booth and a 4-story parking structure. The watershed consisted of areas with high traffic flow, and could be classified as an industrial/commercial area. Three pooled areas were connected with a small stream flow. Inlets were present in the first two regions only; therefore, samples were taken evenly throughout the first two regions and along the flow path connecting the two pools. Small fish were present as well as plentiful wetland plants, however a highly potent sulfur smell was apparent as well as an oily film over the water surface. This undersized wetland was studied by Mr. Robert Tucker as part of his thesis project completed in 2007.

Figure M.1. Profile View of Ra-Vars Wetland Forebay on NCSU Campus

RA-VPD1

This wetpond is located near Ra-Vars1 and consequently near the four story parking structure. The vegetation was more dense at this site than that of Ra-Vars1, with abundant tall grasses and weeds. The heavy plant life made sampling difficult, still all samples were suitably taken. Tadpoles were present and other small unidentified insects.

Figure M.2. Profile View of Ra-VDP Wetland Forebay on NCSU Campus

The following three sites are located in close proximity on NCSU Campus near the College of Textiles Building.

Figure M.3. Profile View of 3 Wetland Forebays on NCSU Campus

RA-TXTBRD_1

This wetpond was located on NCSU campus near the textile building. Unfortunately the inlet was dry. A Sampling was taken at a pool of ponded water, acting as a forebay, approximately 10 feet downstream of the inlet. Samples were also taken along a path of water leading to the pool. Samples were divided with the intent of equal location representation. The series of wetlands contained abundant plants, insects, and algae. Trees and thicker vegetation surrounded the pond. Heavy brush spread throughout the wetpond.

Figure M.4. Profile View of Ra-TxtBrd_1 Wetland Forebay on NCSU Campus

RA-TXTBR_2

Similar to Ra-TxtBr_1, however this wetpond was larger and deeper that Ra-TxtBr_1. Similar wildlife species were identified, however there were less surrounding trees.

Figure M.5. Profile View of Ra-TxtBrd_2 Wetland Forebay on NCSU Campus

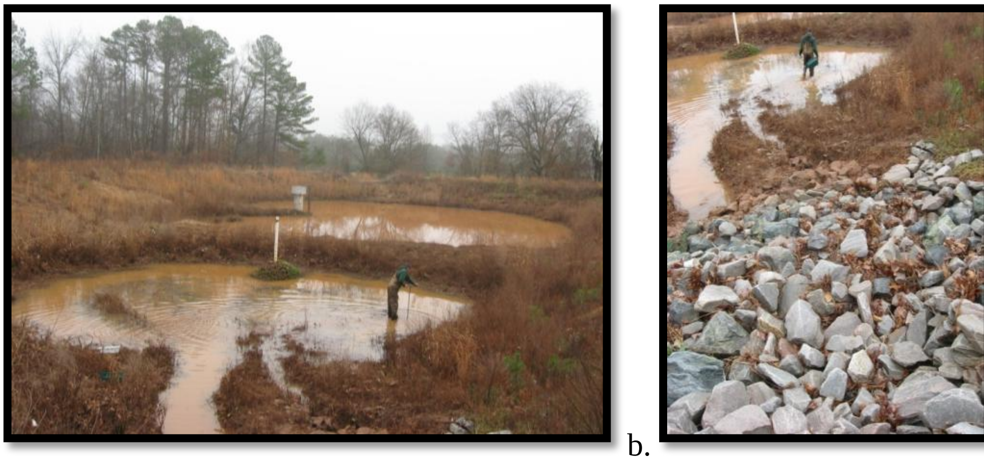
RA-TXTBRD_3

This wetpond was located adjacent to Ra-TxtBrd_2, therefore holds an identical site description. This pond was indeed smaller in surface area than Ra-TxtBrd_2, yet exhibited a comparable water depth. Both sites were located near the textile bridge, along the NCSU campus greenway.

Figure M.6. Profile View of Ra-TxtBrd_3 Wetland Forebay on NCSU Campus

RA-ALUMRD_1

This wetpond was located near NCSU campus, adjacent to the alumni association building. The inlet consisted of an inclined rock bed approximately 60 ft long, which ultimately led to the pond forebay. Construction was apparent, and as a result, surrounding vegetation had been removed. Red clay was abundant throughout the pond and on surrounding land. There were no insects or shrubs of any kind visible.



b.

Figure M.7. (a) Profile View of Ra-AlumRd_1 Wetpond Forebay and (b) rock bed inlet

RA-MOTPL_1

This wetpond is located on Sullivan Drive, on NCSU campus, adjacent to the University motor pool, and near a large electrical sub-station and university tennis courts. The pond was surrounded by heavy shrubs and pine trees. There were no plants growing on the surface of the pond itself, most likely due to the elevated water depth. The hooded merganser (*Lophodytes cucullatus)*, a native duck, was observed, which gives reason to believe fish species are also present.

a. b.

Figure M.8. Profile View of Ra-MotPl_1 Wetpond Forebay on NCSU Campus