Durham Area Stormwater Topics

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Prof. Will Wilson

PCB contamination in waterways with a case study of the Neuse River basin

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Polychlorinated biphenyls (PCBs) are anthropogenic compounds that contaminate ecosystems and serve as worldwide public health concerns. PCBs could be transported through the ecosphere through urban storm water runoff, erosion, and wastewater effluent. In North Carolina, tributaries and reservoirs of the Neuse River were contaminated with elevated levels of PCBs. The mechanisms of PCB transport through the ecosphere, the effects of PCBs on wildlife and humans, and a case study of PCB contamination in the Neuse River basin will be discussed in this paper.

PCBs: descriptions and production

As shown in Figure 1, PCBs are made up of two cyclohexenes connected with a single bond, substituted with one to ten chlorine atoms (Weintraub and Birnbaum 2008). To this date, 209 PCB congeners based on different arrangements of chlorine atoms have been discovered. Because of their ability to act as good insulators, PCBs were produced for use as transformer coolants and dielectric fluids in capacitors, transformers, and carbonless paper (Shiu and Mackay 1986; Longnecker *et al.* 1997). Due to their use in production of electrical equipment such as refrigerators and computers, elevated levels of PCBs have been detected in urban and developed areas. From 1930 to 1993, 1.3 million tons of PCBs were produced worldwide (Pozo *et al.* 2009). Prior to the federal ban on the production of PCBs in 1979, 1.2 billion pounds of PCBs were produced in the United States (Weintraub and Birnbaum 2008).

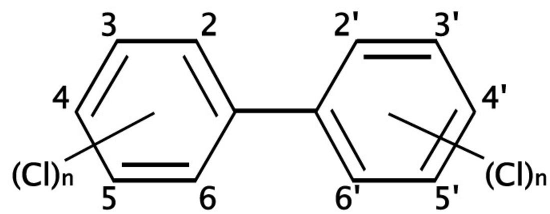

Figure 1- Atomic structure of polychlorinated biphenyls (PCBs). PCBs are made up of two six-carbon rings that are connected by a single bond and can be substituted with one to ten chlorine atoms. Photo credit to http://www.svend-erik.nu/Billeder/255.jpg

PCBs in the environment

Because PCBs have great hydrophobicity and lipophilicity, low vapor pressure, and chemical inactivity (Shiu and Mackay 1986), they persist in the ecosphere, accumulate in sediments, and are easily transported by runoff and atmospheric deposition. PCB contamination in the environment can also result from inadequate treatment of wastewater, oil spills, waste incineration in plants with inefficient gas filters, and sewage sludge contamination (Roiss *et al.* 2002). Elevated concentrations of PCBs had been found in sediments, water, and wildlife worldwide. The Great Lakes, San Francisco Bay, the Philippines, France, China, and Switzerland are a few examples of globally affected areas (Pozo *et al.* 2009; Rossi *et al.* 2002). Specifically, Japan and western Taiwan served as sites for epidemiological studies due to rice oil contamination with PCBs and other polychlorinated compounds, as illustrated below.

Pathways for PCBs to enter waterways

As described below, urban runoff, erosion of buried sediments, wastewater effluent, and atmospheric deposition are the major pathways for PCBs to enter waterways in the ecosystems (Davis *et al.* 2007; Rossi *et al.* 2002).

Urban runoff

Urban runoff through storm water could carry PCBs from urban pavements and impervious surfaces to storm drains and deposit them along the edge of the water bodies, oftentimes in areas with minimal flushing, resulting in elevated concentrations of PCBs in major waterways (Davis *et al.* 2007). Currently implemented practices to reduce urban runoff of PCBs include cleanup of contamination sites on land, vegetation buffers along contaminated pavements, and street sweeping. More data is needed to determine the effectiveness of these measures.

Erosion

Erosion of previously buried sediments contaminated with PCBs also serves as a major pathway of PCB introduction to the waterways (Davis *et al.* 2007). Active sediments are sediments that are interacting with the water columns through biological, chemical, and physical processes while buried sediments are located below active sediments and do not interact with the water columns. Erosion of buried sediments increases the amount of PCBs in the water column and the biosphere, effectively introducing PCBs to the waterways.

Wastewater effluent

Wastewater effluent from sewage treatment plants could introduce PCBs to waterways (Rossi *et al.* 2002). Due to their resistance to biodegradation, PCBs from treated wastewater and the residual PCBs in recycled toilet paper could be trapped and concentrated in the sludges of wastewater treatment plants. The subsequent use of sludges as fertilizers could introduce PCBs to the environment. In addition, insufficient treatment of wastewater could result in residual PCBs in treated wastewater. Subsequent to sewer cross-connections introducing wastewater in storm water pipes, urban storm water could transport the residual PCBs in treated wastewater to the receiving waterways, resulting in PCB contamination in the environment. Incineration of wastewater treatment plant sludges and separation of sewer systems could mitigate PCB contamination in wastewater effluent.

Atmospheric deposition

Because of their volatile nature, PCBs could enter the atmosphere and be transported across the ecosystems (Davis *et al.* 2007). PCBs could then deposit in the watershed and enter waterways through storm water runoff. Elevated levels of PCBs in air have been recorded at sites in India, China, and the Philippines among others (Pozo *et al.* 2009).

Effects of PCBs on wildlife and human health

Despite the worldwide ban on the sale and production of PCBs, blood levels of PCBs in humans remained constant with no signs of decrease (Consonni *et al.* 2012). This situation is alarming because PCBs were found to cause negative influences on human health, as illustrated below.

Mechanisms of toxic action: dioxin-like PCBs

PCBs can be classified as coplanar, with no chlorine atoms in *ortho* positions, or non-coplanar. Coplanar PCB congeners, also known as dioxin-like PCBs, can bind to aryl hydrocarbon receptor (AhR) in a dose-dependent manner. Binding of dioxin-like PCBs to AhR induces dimerization of AhR and stimulates expression of genes such as those coding for hepatic enzymes (Boix and Cauli 2012; Long and Bonefeld-Jorgensen 2012). Non dioxin-like PCB congeners act in a different mechanism that is not currently understood, and are found to constitute a greater portion of environmental samples.

Bio-magnification

With their high solubility in lipids, PCBs deposit and concentrate in the fat tissues of animal bodies (Weintraub and Birnbaum 2008). Elevated levels of PCBs in fish can be attributed to PCB uptake from water and PCB bio-magnification through the predation of contaminated benthic feeders. A study in Lake Michigan revealed that PCB uptake through bio-magnification could explain ninety-nine percent of the observed PCB contamination in lake trout (Thomman and Connolly 1984). Bio-magnification of PCBs might be congener-dependent. Dioxin-like PCBs showed no evidence for bio-magnifications but non-dioxin-like

PCBs were shown to bio-magnify (Long and Bonefeld-Jorgensen 2012).

Human epidemiology of PCBs

Occupational exposure served as the major source of PCB exposure for the U.S. population from 1940 to 1980 prior to the federal ban on the production of PCBs (Sinks *et al.* 1992). Occupational exposure to PCBs had been associated with melanoma, pancreatic cancer, and kidney cancer in some studies but the overall results remained inconclusive (reviewed in Longnecker *et al.* 1997). However, increased occupational exposure to PCBs was shown to result in altered liver function (Brown and Jones 1982). In addition, dermatologic conditions were observed in cohorts of manufacturing workers with elevated exposure to PCBs. Specifically, hyperpigmentation and chloracnes were found to be associated with occupational exposure to PCBs (reviewed in Longnecker *et al.* 1997).

Human populations can also be exposed to PCBs through diet and prenatal transfer of fluids from placenta to fetus (Rogan *et al.* 1986). The effects of PCBs on birth weight and spontaneous abortion remained inconclusive (Wasserman *et al.* 1982; Fein *et al.* 1984). However, prenatal exposure to PCBs has negative effects on human neurologic development. In two studies conducted in North Carolina (Rogan *et al.* 1986) and Michigan (Fein *et al.* 1984), elevated exposure to PCBs was associated with hypotonia (decreased muscle tone), hyporeflexia, and altered motor activity.

Recent human epidemiological studies of PCBs elucidated additional detrimental health effects of the chemicals. Increased exposure to PCBs was found to alter the concentrations of the neurotransmitter serotonin and its major metabolite 5-HIAA in the brain (Boix and Cauli 2012). In addition, a recent review by Crinnion (2011) revealed that PCB exposure is associated with symptoms of type-II diabetes such as high blood pressure and elevated triglycerides. Specifically, increased exposure to certain PCB congeners such as PCB-153 and PCB-118 had a statistically significant dose-response relationship with increasing risks of type-II diabetes.

Case studies of rice-oil contaminations in Asia

In Japan and Taiwan, where rice oil was contaminated with PCBs in 1968 and 1979, respectively, dermatologic conditions of chloracne and hyperpigmentation were frequently observed among the populations (reviewed in Longnecker *et al.* 1997). Slowed nerve conduction, altered immunoglobulin levels, and abnormal liver functions were also documented. Mortality due to liver cancer and disease was increased in both locations. Exposed children were reported to have lower body weights, cognitive deficiencies, and exhibit disordered behavior (Longnecker *et al.* 1997; Winneke *et al.* 2002; Hsu *et al.* 2007).

Fish consumption

Even though occupational exposure served as the major source of PCB exposure prior to the federal ban on PCBs, fish consumption is the current leading source of PCB exposure in the United States (Weintraub and Birnbaum 2008). Significantly higher levels of PCBs were found in non-Hispanic black populations due to their greater consumption of catfish and subsistence fishing habits in PCB contaminated waterways (Weintraub and Birnbaum 2008). The health effects of such racial discrepancy in consumption of PCB-contaminated catfish might further widen the poverty gap and exacerbate problems related to differential access to healthcare.

Local case study of PCB contamination—Lake Crabtree and other Neuse tributaries

Background

Located near the Raleigh-Durham International Airport, the Ward Transformer NPL (Figure 2) was a transformer reconditioning facility that operated from 1964 to 2006 (Neuse River Foundation 2013). Its business objectives included building, repairing, and storing transformers, which are devices that transfer electrical energy from one coil to another using magnetic flux. Ward was also responsible for the disposal of hazardous substances related to building transformers and other electrical devices for its business partners. Due to its indiscreet release of 30,000 gallons of hazardous PCB-contaminated oil along 200 miles of roadsides in fourteen counties in North Carolina, the EPA and the North Carolina Department of Environment and Natural Resources launched an investigation of the Ward Transformer headquarters in 1978 (EPA 2012). They found that the site was contaminated with PCBs and other pollutants. However, the investigation was not resumed until 1993, when the EPA collected soil and water samples from the site for further analysis.

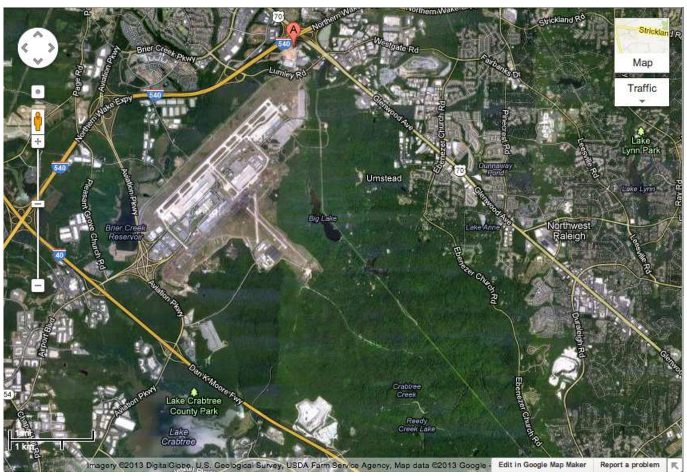

Figure 2- Map depicting the locations of the Ward facility (denoted by "A" in the figure) and the contaminated downstream waterways including Lake Crabtree, the Brier Creek Reservoir, and Crabtree Creek. Photo credited to Google Maps (2013).

Clean-up and mitigation of contamination

The EPA found elevated levels of PCBs on the site of the Ward Transformer headquarters and in the water bodies downstream including multiple tributaries of the Neuse River such as Lake Crabtree, the Brier Creek Reservoir, tributaries to Little Brier Creek, Brier Creek, and Crabtree Creek (locations depicted by Figure 2). As a result, the Ward Transformer facility was included as part of the National Priorities List of Superfund hazardous waste sites for remediation in 2003. In the cleanup of the Ward Transformer headquarters, which started in July 2007, the EPA and the responsible parties excavated 350,000 tons of contaminated sediment using low temperature thermal desorption and tested soil samples for contamination (EPA 2012).

Impacts on the downstream waterways

Elevated levels of PCBs were consistently observed in bottom feeders such as catfish and sunfish in Lake Crabtree and other waterways downstream of the Ward facility as shown in Table I (ATSDR 2008). As a result, starting in 2004, the North Carolina Department of Health and Human Services (NCDHHS) had issued multiple advisories to deter the public from consuming carp or catfish and limit all other fish consumption to 1 meal per month. In spite of the observed PCB contamination, the EPA decided against issuing an active cleanup of waterways downstream of the Ward facility. Instead, the EPA adopted the seemingly inefficient Monitored Natural Recovery program, which relies on natural processes to slowly degrade the contaminants (Neuse River Foundation 2013).

Table I- Levels of PCBs detected in fish in the tributaries of the Neuse River basin (data collected by ATSDR 2008).

Date of Sampling	Sampling Location	Average PCB Level per Fish (mg/kg) or ppm
November 2003	Brier Creek Reservoir	Catfish: 1.0 mg/kg (n=4) Largemouth bass: .65 mg/kg (n=4) Sunfish: .22mg/kg (n=5)
November 2003	Lake Crabtree	Catfish: 1.2mg/kg (n=4) Largemouth bass: .25mg/kg (n=6) Sunfish: .29 mg/kg (n=15)
November 2004	Crabtree Creek just below Lake Crabtree	Catfish: .18 mg/kg (n=3) Sunfish: not detected (n=6)
August and September 2005	Crabtree Creek at William Umstead State Park	Catfish: .340 mg/kg (n=5) Largemouth bass: .160 mg/kg (n=5)
July 2006	Crabtree Creek Near New Bern Avenue	Flathead fish: .05 mg/kg (n=1) Largemouth bass: not detected (n=1)
July 2007	Neuse River at mouth of Crabtree Creek	Channel Catfish: .130 mg/kg (n=8) Largemouth Bass: .0466 mg/kg (n=2)

Lake Crabtree: "catch and release"

Lake Crabtree (location depicted in Figure 2), one of the PCB contaminated tributaries to the Neuse River, was constructed by the Soil Conservation Services in 1989 for purposes of flood control (DWQ 2012). Lake Crabtree houses a park for recreational activities for the public (represented by Figure 3). The Lake Crabtree County Park is a popular venue for subsistence and recreational fishing as illustrated in Figure 5. Active fishermen of the park include women and children, who are vulnerable to the health effects of PCBs. As a result of the advisories issued by the NCDHHS, the Wake County government has implemented the "catch and release" policy for Lake Crabtree and Crabtree Creek, which advocates for people to release fish caught from the water (Wake 2008; depicted in Figure 4).

Figure 3- A photograph of the PCB contaminated Lake Crabtree County Park located in Wake County, North Carolina. Photo was taken in April, 2013.

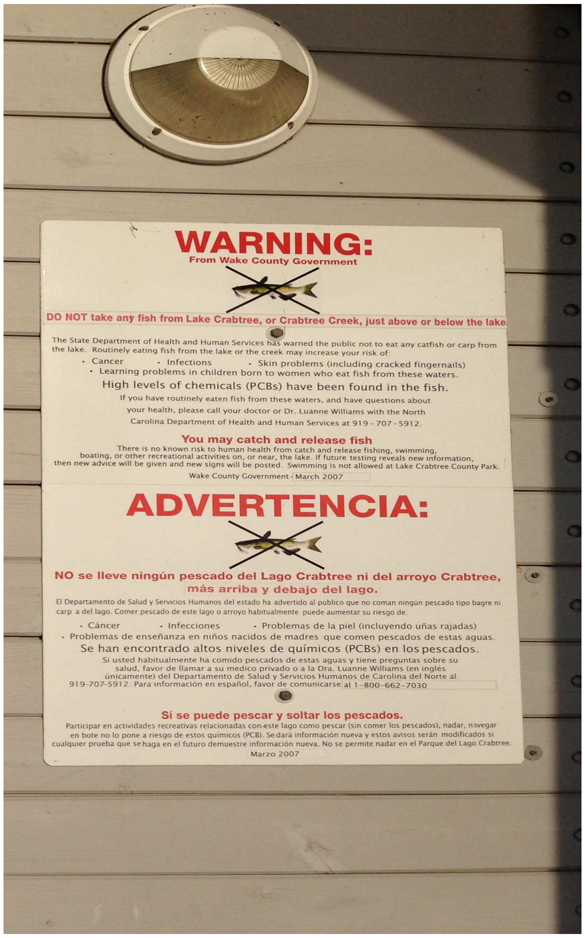

Figure 4- A photograph of the PCB contamination "catch and release" advisories at Lake Crabtree County Park. Photo was taken in April, 2013.

Figure 5- A photograph depicting two recreational fishers at the PCB contaminated Lake Crabtree County Park. Photo was taken in April, 2013.

Lessons learned and future directions for the local PCB contamination

This local case of PCB contamination in the Neuse River basin illustrates how unethical and indiscreet release of PCB contaminated material can pose lasting threats to the human population and the ecosystem. As evidenced by the persistence of PCBs in the environment, complete removal of the contaminant from the soil of the Neuse River basin will be a challenging task. In the future, further public education is needed to increase public awareness of the situation regarding PCB contamination. Greater community involvement and grassroots political action are required to advocate for the reversal of the Monitored Natural Recovery program and to launch an active cleanup of the downstream waterways.

Conclusions

As reviewed in this paper, PCBs are persistent anthropogenic compounds that enter the waterways by various means such as urban runoff and erosion of buried sediments, and pose threats to the health of wildlife and humans. PCBs were industrially useful as transformer coolants and dielectric fluids for the production of electrical equipment. The mass production of PCBs without a thorough investigation of their potential detrimental effects on the environment and human health indicates a need for the EPA to conduct exhaustive investigations of anthropogenic compounds prior to their release and careful scrutiny of such compounds subsequent to their manufacture in the future. In addition, in the local study of PCB contamination, the fifteen-year lapse between the EPA investigations of the Ward facility reveals a need for the EPA to increase efficiency in conducting investigations of the parties responsible for environmental contamination and to mitigate the environmental detriments with efficient active cleanup methods in a timely manner.■

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PBDEs: A Sampling of their Effect on Human Health & a Call for More Legislative Regulation

Natalie Skeiky

1. Introduction

Brominated flame retardants (BFRs) are chemical compounds that have been used in various commercial products throughout the years as an effective measure against potential fires. BFRs are fashioned in upholstery fabric, foam, and plastic; they are either covalently bonded to the polymer or additively mixed into the final product (Ward et al., 2008). Most often they are present in objects such as infant car seats and cribs, children's sleepwear, home furniture such as sofa cushions, carpet pads and paint, plastic products, kitchen appliances, and electronics from computers to televisions (Richardson, 2009). Due to their proximity to everyday human life, BFRs have begun to raise concerns among researchers in the environmental and health spheres. These chemicals have been found both near and far from their locations of production or use in the environment as well as in human and wildlife samples.

Brominated flame retardants operate on radical chemistry. They inhibit the combustion of organic material by scavenging the free radicals that normally encourage the spread of flames (Ward et al., 2008). Specifically, when BFRs are heated, they release bromine free radicals that can then scavenge other free radicals that are part of the flame propagation process, thereby countering fire escalation. (Richardson, 2009).

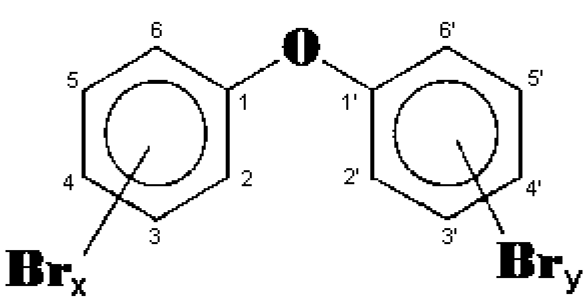

Figure 1: Basic molecular structure of a PBDE (IPCS)

Since the banning of polybrominated biphenyls in the late seventies, polybrominated diphenyl ethers (PBDEs) have been a popular component of flame retardants (Zota *et al*., 2011). However, this specific class of additive brominated flame retardants poses its own risks. First, PBDEs are made of 209 possible congeners—compounds with similar structures and chemical properties—that contain anywhere from 1 to 10 bromine atoms (Figure 1); of these congeners, 23 are of environmental significance (Ward et al., 2008). Second, not only are they environmentally persistent, but because they are lipophilic—can dissolve in fats and lipids—they also bioaccumulate in the tissues of animals and humans and have the potential to biomagnify up the food chain (Ward et al., 2008). Third, their recent correlation with developmental neurotoxicity in mice and their potential for hormonal disruption and even cancer pose great health concerns (Richardson, 2009). Therefore, there is a research-supported need for legislation that phases out the manufacturing of PBDE-containing products and regulates their use and disposal.

2. Exposure

High levels of PBDE have been detected worldwide in everything from soil and sediment to air, house dust, water, and even biological samples. By analyzing the global concentration data, scientists have shown that PBDE levels in humans have increased by a factor of 100 over the past three decades, with concentration loads doubling every 5 years (Ward et al., 2008). North Americans, whose average contaminant levels are 20 times that of Europeans, have the highest body burden in the world (Ward et al., 2008). Many 'market basket' studies, which correlate PBDE levels in both food and their consumers, indicate that total PBDE loads in North Americans (U.S. and Canada) are due to several simultaneous sources of exposure rather than a solely nutritional route (Ward et al., 2008).

2.1 Dietary

The consumption of contaminated food is predicted to be a major route to human PBDE exposure (Ward et al., 2008). It is possible that the application of PBDE-laden bio-solids to agricultural areas is a potential vector to food contamination (Ward et al., 2008).

PBDEs have been detected in a variety of meat, fish, and dairy products in North America, Europe, and throughout the world. A study done within the mid-Atlantic region of the U.S. (Maryland, Washington D.C., North Carolina) detected varying PBDE levels in both wild-caught and farm-raised fish fillets that are commercially sold (Hayward *et al*., 2007). The detectable PBDE concentrations in fish being sold for human consumption have implications for human health, especially since they continue to be present despite the 2004 phase out of 2 of the 3 commercial mixtures—the penta- and octa-BDEs (Davis *et al*., 2011). Although fish have the highest PBDE content, followed by meat and dairy, meat is the estimated major source of PBDEs in the diet of U.S. residents (Costa *et al*., 2007). Vegetarians tend to have lower plasma levels of PBDEs than the general U.S. population (Costa *et al*., 2007). In Hong Kong, China, the blood plasma of humans and fish that were analyzed had similar concentrations of PBDEs signaling that diet, specifically seafood, is a major source of these contaminants (Wang *et al.*, 2012). A study in South Korea identified fish and shellfish as comprising the largest portion of total human dietary PBDE intake (Na *et al*., 2013). PBDE levels were shown to be highest in infants and decrease with increasing age (Na *et al*., 2013).

2.2 Environmental

Soil, air, and water are additional sources of exposure. The World Health Organization (WHO) found that contact with current-use products containing PBDEs constitutes a viable source of exposure (Ward et al., 2008). The volatization of PBDEs from incinerated and landfill waste as well as PBDE release from wastewater treatment plants (WWTPs) are additional sources of exposure (Ward et al., 2008).

Figure 2: Aerial view of the North and South Durham water treatment facilities (Hazen and Sawyer)

Sewage sludge and biosolids, which often exceed 1 mg/kg dry weight, contain some of the highest PBDE concentrations in North America (Davis *et al*., 2011). It is possible that PBDE-containing products contaminate residential wastewater that flows to WWTPs and other bodies of water.

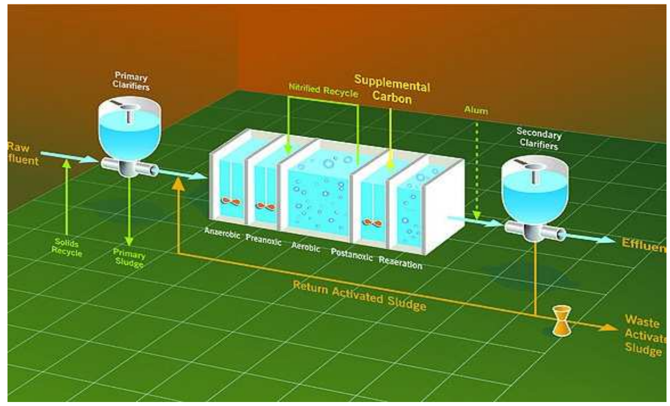

Figure 3: South Durham nitrogen removal process, which indicates the presence of sludge (Bilyk).

In a Durham, North Carolina WWTP where the penta- and deca-BDE commercial mixtures accounted for the PBDE contaminant burden in sewage sludge and biosolids, there seemed to be an overall decrease in PBDE concentration over the observed 4-year data period (Davis *et al*. 2011) (Figures 2 & 3). This same study indicated that indoor dust that infiltrates wastewater systems may be a valid source of PBDE wastewater contamination. Several studies have identified the inhalation of contaminated house dust as a risk factor; this medium accounts for 80-93% of PBDE exposure for toddlers in particular (Costa *et al*., 2007). Two studies determined that ambient concentrations of bioavailable PBDEs are much higher in urban areas than rural and remote areas in the Northeastern and East-Central U.S. (Strandberg *et al*., 2001; Hoh and Hites, 2005).

2.3 Mother-to-fetus transmission

The strongest correlations appear to be between PBDE levels in house dust, breast milk, and blood plasma (Ward et al., 2008). In fact, human serum and human milk are used as Standard Reference Materials (SRMs) for monitoring environmental contaminants like PBDEs (Schantz *et al.*, 2013). There is a positive relationship between ambient levels of PBDEs in the home and in the personal air of inhabitants (Ward et al., 2008). A study in Copenhagen, Denmark that sampled dust and indoor air in the homes of 43 pregnant women found a dominance of the BDE-209 congener, although it did not exceed pentaBDE concentration (Vorkamp *et al*., 2011). The placental data from the 43 women revealed a significant correlation with the dust concentrations of pentaBDE congeners indicating the importance of dust in pentaBDE exposure (Vorkamp *et al*., 2011). Another study showed that PBDEs can be found in both maternal serum and umbilical cord serum; however, whether the PBDE congener concentrations are higher in one sample medium versus the other depends on the stage of pregnancy (Foster *et al.*, 2011). Even the hair of newborns—as little as 10mg—is a useful matrix for examining *in utero* exposure to PBDEs (Aleksa *et al*., 2012).

3. Health Implications

PBDEs are global contaminants that have been detected in human adipose tissue, serum, and breast milk samples in North America, Europe, Asia, Oceania, and the Arctic (Chevrier *et al*., 2010). Over the last three decades, the concentration of PBDEs in human serum and breast milk has increased exponentially (Chevrier *et al*., 2010). The presence of these synthetic chemicals in human serum and breast milk is troubling because of the potential negative impacts this can have on childbearing women and children. Prenatal exposure to PBDEs has already been found to alter spontaneous motor behavior, memory, and learning in rats and mice (Chevrier *et al*., 2010). Recent studies on humans have reported that maternal serum PBDE levels are inversely related to birth weight and scores on tests of intelligence and attention (Chevrier, 2010). Since maternal thyroid hormones (THs) are essential for fetal brain development as well as modulation of the menstrual cycle, it has been suggested that PBDEs may affect neurodevelopment and fertility by disrupting THs. After all, during the first half of gestation, thyroid hormones are delivered to the developing fetus via the placenta, and though the fetus begins to produce its own later in the second trimester, it still depends on its mother's supply to stabilize its TH levels (Stapleton *et al*., 2011).

3.1 Thyroid

The negative health effects of PBDE may be attributed to their chemical structure similarity to thyroid hormones—thyroxine (T_4) above all (Stapleton *et al*., 2011). In laboratory studies done on animals, PBDEs have been shown to affect thyroid regulation in three ways: (1) decreasing circulating levels of thyroid hormones, (2) altering the expression of genes that encode thyroid-regulating proteins, and (3) reducing the activity of thyroid-regulating enzymes (Stapleton *et al*., 2011). *In vitro* studies have demonstrated that PBDEs and hydroxylated PBDEs—the PBDE metabolites hypothesized to have more potent effects on TH regulation than PBDEs—can compete for binding sites on thyroid hormone transporters in serum (Stapleton *et al*., 2011).

In a 2010 study done by Chevrier *et al*., the relationship between serum PBDE concentrations and thyroid function in 270 pregnant low-income Latina women living in California was determined. They measured the concentration of 10 PBDE congeners, free thyroxine (T_4), total T_4, and thyroid-stimulating hormone (TSH) around the 27^{th} week of gestation. They reported two major findings: (1) a significant inverse relationship between TSH concentrations and serum measurements of both the sum of all PBDE congeners (ΣPBDEs) and BDEs 28, 47, 99, 100, and 153 and (2) elevated odds of subclinical hyperthyroidism in relation to ΣPBDEs and BDEs 100 and 153. They attributed these associations to decreased TSH levels in participants receiving the highest doses of PBDE serum concentrations. There was a general lack of statistically significant relationships between PBDEs (summed or individual) and T_4 (free and total). It should be noted that whereas studies done with rodents yielded hypothyroxinemic effects, human studies yield hyperthyroidic effects. This may be explained by physiological differences and the fact that higher doses are administered in animal studies.

Stapleton *et al*. (2011) examined levels of PBDEs and PBDE metabolites in a pregnancy cohort of 140 African American women at or over 34 weeks of gestation living in North Carolina. They aimed to not only augment available data on the relationship between PBDE levels and THs, but also to determine whether PBDE and TH level associations previously noted in the second trimester were present during the third. They collected blood samples during routine prenatal visits and screened them for PBDE congeners, three –OH containing PBDE metabolites, and five THs. The researchers found PBDE congeners in all samples and two of the hydroxylated BDE metabolites in over 67% of the samples. The PBDE and OH-BDE levels were similar to those reported among the general U.S. population in a 2003-2004 National Health and Nutrition Examination Survey. This unexpected result suggests that PBDE levels are stable; otherwise the 4-to 7-year interim between the two studies, coupled with the phase out of PBDE use since 2004, would have led to a decline. Therefore, the long half-life of PBDEs in the human body contributes to their persistence and association with thyroid hormones throughout pregnancy, though the mechanisms of this are unclear.

3.2 Neurotoxicity

In 2011, Mireia *et al*. conducted a study that investigated the association between pre *and* postnatal PBDE concentrations and both neurodevelopment and thyroid hormone levels in 4-year-old children. They measured PBDE cord blood and serum concentrations and also assessed the motor and cognitive function, attention-deficit, hyperactivity and impulsivity, and social competence of 4-year-old children belonging to a cohort of pregnant mothers recruited between 1997 and 1998 in Menorca, Spain. PBDE 47 was the only quantifiable congener (≥0.002 ng/ml) found in a significant number of samples. Therefore the researchers assessed the children in two groups: those with concentrations above ("exposed") and below ("referents") 0.002 ng/ml. First, although, the associations were not statistically significant, they found that children pre-and post-natally exposed to PBDE 47 had lower cognitive and motor function scores than referents. Second, post-natal exposure to PBDE had three effects: (1) it increased risks of attention deficit symptoms of Attention Deficit Hyperactivity Disorder (ADHD) in a statistically significant way, (2) it did not increase risks of hyperactivity symptoms of ADHD in a statistically significant way, and (3) it led to a statistically significant higher risk of poor social competence symptoms. Oddly, there was no clear association between the neurological effects of post-natal PBDE 47 exposure and thyroid hormone levels at age 4. The scientists write that the relationship between PBDE exposure and THs has been inconsistent in animal studies as well and call for further research to understand the role THs play in the neurotoxic effects of PBDE exposure.

A study published by Hoffman *et al*. (2012) examined the association between the levels of five PBDE congeners—BDEs 28, 47, 99, 100, and 153—in breast milk and the social and emotional development of toddlers. The researchers used data from the Pregnancy Infection and Nutrition (PIN) Babies Study, which followed a cohort of pregnant women and their children at the University of North Carolina prenatal care clinic. These women gave breast milk samples for analysis at 3 months postpartum and completed the Infant-Toddler Social and Emotional Assessment (ITSEA) when their children reached 30 months of age. The researchers found a small, imprecise, but consistent association between increased levels of BDEs 47, 99, and 100 and increased externalizing behaviors, especially activity/impulsivity behaviors. This means these children were characterized as very active, fidgety, having trouble sitting still, or having difficulty inhibiting their actions. Compared with BDE-47 concentrations below the identified median value, externalizing behavior domain scores were 1.6 and 2.8 points higher for the children of women with breast milk BDE-47 concentrations in the 3rd and 4th quartiles, respectively. There was no association between PBDEs and other social and developmental domains. The data supports previous research indicating the neurodevelopmental impacts of PBDE exposure by suggesting a subtle association between early-life PBDE exposure and increased activity/impulsivity behaviors in early childhood. However, since the only comparable published study reported better behavior in 5- to 6-year-old children with pre-natal exposure to BDE-100, more comprehensive studies are needed.

4. Legislation & Policies

The growing knowledge of the negative environmental and health effects of PBDEs combined with their prevalence in a variety of commercial products throughout the world have initiated efforts to curb the use of these flame retardants.

4.1 International

More than 70,000 metric tons of PBDE have been produced worldwide; the United States and Canada account for the usage of about half of this amount, which includes almost all of the penta-BDEs manufactured (Ward et al., 2008). Until 2005, three commercial PBDE mixtures were distributed for use in North America: penta-bromodiphenyl ether (pentaBDE), octa-bromodiphenyl ether (octaBDE), and deca-bromodiphenyl ether (decaBDE) (Stapleton *et al*., 2011). These three congeners of PBDE have gained a lot of attention internationally due to the potential environmental and health risks they pose. Two international treaties on toxic pollutants have pointed out that PBDEs are a global concern that nations must work together to address. In November 2005, the Persistent Organic Pollutants Review Committee of the United Nations (UN) Stockholm Convention decided that pentaBDE met the criteria of a persistent organic pollutant and suggested that octaBDE be included as well (UN, 2005; UN, 2006). This was a significant step because the *Stockholm Convention on Persistent Organic Pollutants* (POPs) is a global UN agreement that outlines government obligations for the use, export, import, and disposal of POPs with the aim of reducing or eliminating the environmental release and effects of harmful chemicals (Ward et al., 2008). As a result and because of the ability of pentaBDE to be transported long distances from its point of use or production, the *Long-Range Transboundary Air Pollution* (LRTAP) convention of the United Nations Economic Commission for Europe (UNECE) determined that the pentaBDE commercial mixture should be classified as a POP under the 1998 Aarhus Protocol (EC, 2006b).

4.2 North America

In the U.S. and Canada, public concerns and economic factors finally encouraged the only North American producer of octaBDE and pentaBDE mixtures, Great Lakes Chemical Corporation (now Chemtura), to voluntarily stop producing these chemicals in December 2004 (Ward et al., 2008). This move effectively eliminated the use of penta- and octaBDE products in North America (Ward et al., 2008). Although these two PBDE congeners are no longer manufactured or imported into North America, decaBDE was continually used in consumer products until manufacturers agreed to phase out its use by 2012 (Stapleton *et al.*, 2011). All three PBDE congeners are still present in much of the furniture, mattresses, televisions, and office equipment currently in use (Ward et al., 2008). Moreover, PBDE-containing products are often incinerated, recycled, or dumped into landfills in happenstance manners without much thought for their impact on human health (Ward et al., 2008). The *International Association of Electronics Recyclers* estimates that 400 million consumer electronics are discarded each year (IAER, 2006). Stricter legislation is needed to regulate the fate of PBDE-containing products.

The only U.S. federal regulation that relates to the future restriction or prohibition of any PBDE category is the 2006 *Significant New Use Rule* (SNUR) under the Toxic Substances Control Act (Ward et al., 2008). It requires all persons and businesses to notify the Environmental Protection Agency (EPA) 90 days before intending to manufacture, use, or import pentaBDE or octaBDE products (EPA, 2005a). Another effort released in 2006, the *Polybrominated diphenyl ethers (PBDEs) project plan*, outlined the EPA's major initiatives toward PBDEs. Thankfully, beginning in 2006, individual states enacted their own regulations not only banning the use of penta- and octaBDEs but also restricting decaBDE use in certain manufactured products: California, in 2003, was the pioneer state in the case of the former and Washington and Maine, in 2008, were the two pioneer states with respect to the latter (Ward et al., 2008). Unfortunately, despite special interest groups lobbying their state representatives, House Bill 823/Senate Bill 993 to regulate PBDE use in commercial products was denied and North Carolina has yet to pass any PBDE legislation (National Conference of State Legislatures; Toxic Free North Carolina).

5. PBDE Alternatives

The policy restrictions and growing public concerns regarding the adverse health implications of PBDEs have caused the development of many new alternative flame retardants. Refer to the 2008 review paper by Ward *et al*. for an enumeration of the policy initiatives and various PBDE alternatives.

6. Concluding Remarks

The use of flame retardants in commercial products has been integral in the drastic drop in fire incidence over the past 30 years (Costa *et al*., 2007). However, recent research has substantiated the claims that the impact of PBDEs on the environment, wildlife, and human health outweighs the fire prevention properties of these synthetic chemicals. Due to the fact that PBDEs are not fixed in their polymer products through chemical binding, they can leak into the environment and are considered to be persistent organic pollutants (Costa *et al*., 2007). Through widespread environmental contamination and their lipophilic characteristics, PBDEs bioaccumulate in living tissues and biomagnify up the food chain. PBDEs have been associated with abnormal thyroid hormone functioning and neurotoxicity in rodents and humans, especially children.

The adverse effects of PBDEs on human health require a robust legislative effort to regulate, restrict, and outlaw their use. Although international conventions, a single U.S. federal regulation, and individual legislative actions by only a dozen U.S. states exist, this is merely a modest start. Of the 209 PBDE congeners, only two have been banned worldwide. Furthermore, the presence of PBDEs in products that have already been manufactured will continue contributing to environmental pollution by these contaminants. Future legislation needs to implement effective strategies that not only phase out the use of PBDEs in commercial products, but also target disposal procedures for PBDE-containing products that are already in use; no U.S. state should be excepted.

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A Comprehensive Look at Current Agricultural Practices and Eutrophication on the Tar-Pam and Neuse River Basins in North Carolina

Santosh Shanmuga

Introduction

Eutrophication of river basins has become a major problem over the past few decades.[1,2] Eutrophication is the ecological response of increased primary production to an increase in nutrients such as nitrates and phosphorus, and can lead to massive algal blooms and hypoxic water conditions. This hypoxia can have severe detrimental effects on local fish and invertebrate species, and can cause shifts in species composition and loss of biodiversity. Eutrophication can happen in any body of water, but are especially prevalent in coastal zones where rivers empty into the ocean. Currently, there have been 375 hypoxic coastal zones identified throughout the world and this number is constantly rising (Fig.1).[3] Eutrophication is driven by nutrient loading of the water primarily due to nitrogen and, to a lesser extent, phosphorus.[1,2] Specifically, nonpoint source pollution seems to be the biggest contributor of nitrogen and phosphorus loading in watersheds.[4,5] Examples of nonpoint source pollution include runoff from agriculture/irrigation, pastures and range, un-sewered areas, construction sites, as well as atmospheric deposition. These contributors of pollution are especially troublesome as nitrogen and phosphorus can be retained in the soil for decades long after human use has stopped. It can easily leach into groundwater or surface water, and is also easily volatilized (NH_4) and subsequently deposited.[6] Human land use has contributed to the majority of nutrient loading in river systems and results in eutrophication of these waters. This is universally perceived as a threat to biodiversity and human health. This paper will look at the problem of eutrophication in two North Carolina watersheds, the Tar-Pamlico and the Neuse, and examine its causes as well as current measures to mitigate this problem.

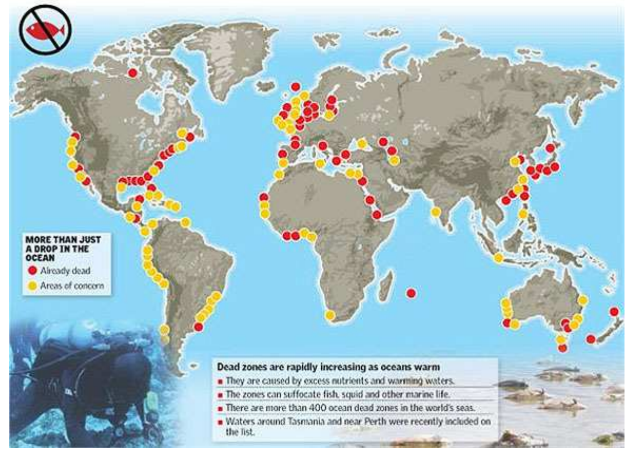


Figure 1. Global map of coastal hypoxic zones, indicating areas of concern and those that are already "dead".[7]

The Tar-Pamlico River is found in northeast North Carolina, originating in Person Country and travelling 215 miles before emptying into the Pamlico Sound in coastal North Carolina. The Tar-Pamlico encompasses 5,571 sq miles while flowing through 16 counties, the majority of which are primarily rural-agricultural. 55% of the land in the basin is forest or wetland, 25% is cultivated cropland or managed pastures, and about 1% is urban.[8] The Neuse River is also found in northeast North Carolina, originating in Durham Country and travelling 275 miles before emptying into the Pamlico Sound. The Neuse encompasses 6,235 sq miles while flowing through 18 counties. The land use for the Neuse watershed is also primarily agricultural; 56% of the land in the basin is forested/wetland, 23% is cultivated cropland and managed pastures, and only 8% is urban.[8]

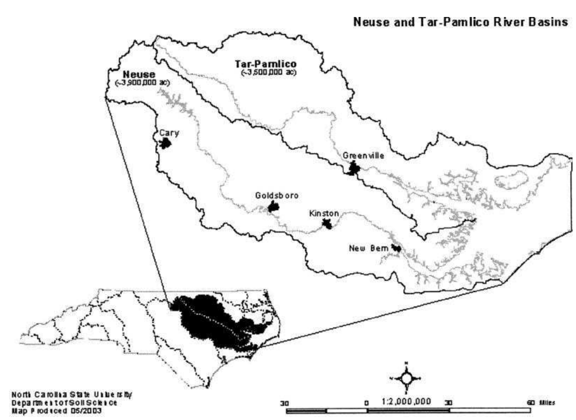


Figure 1: Map of watershed for Neuse and Tar-Pamlico rivers.

The Problem

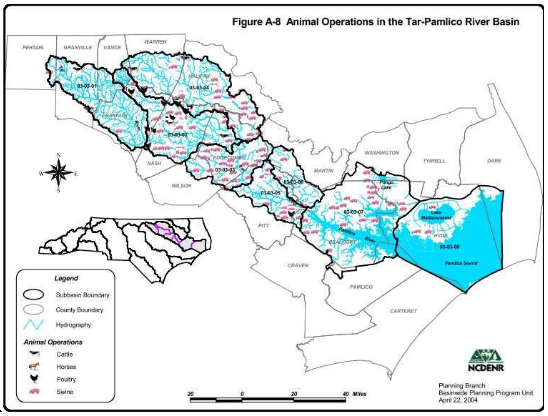
Eutrophication of the Pamlico Sound has been documented in several cases.[9,10,11] This has been linked directly to nutrient loading from agricultural practices (primarily fertilization and livestock waste) as well as episodic events such as hurricanes.[9,12] In addition, both the Tar-Pamlico river and Neuse river experience periodic algal blooms, mostly the dinoflagellate *Pfiesteria piscicida*.[13,14,15,16] This is problematic because *P. piscicida* in numbers can cause toxicity and is the major cause for what is commonly known as red tide.[17] Pamlico sound, and especially the Neuse River, have experienced a large number of fish kills every year (Fig. 3) for over the past two decades as a direct result of hypoxic conditions and toxic algal blooms.[18, 19, 20] This poses a real threat to local fisheries and already has caused severe damage to local stocks of fish like the Atlantic Menhaden (*Brevoortia tyrannus*), which is a staple of the ecosystem and serves as the main source of food for many other species of predatory fish, marine mammals, and birds. In addition to ecosystem repercussions, these effects of eutrophication also have negative effects on commercial fisheries as well as recreation of all sorts. Red tide makes it unsafe to swim, and massive fish kills give off an unpleasant, rotten odor that permeates with ease.

Figure 3: Massive 2009 fish kill in the Trent River (Neuse tributary) near New Bern, NC

Causes

The largest contributor to these hypoxic conditions and the toxic algal blooms are nutrient loading of nitrogen and, to a lesser extent, phosphorus. Excessive nitrogen and phosphorus cause a spike in algal primary productivity. As a result, algal biomass explodes while the dissolved oxygen content of the water decreases. This low O_2 content, coupled with the fact that algal blooms can be highly toxic, is what causes most of the damage and creates these hypoxic "dead zones". As the Neuse and Tar-Pamlico river watersheds are primarily agricultural, it is easy to see what can be the largest contributor to excessive nitrogen and phosphorus loading.[21] The main cause of the excessive loading is the overuse of nitrogen-rich fertilizers on crops. Excess fertilizer not used by the crops can then either leach into groundwater, get washed into streams via runoff, or be retained in the soil over a period of decades.

In looking at the Tar River (Table 1), we can see that about a quarter of land in the watershed (and the majority of anthropogenic land use) is cultivated cropland, uncultivated cropland, and pasture. It should also be noted that the total percent of cultivated cropland had decreased from 1982 to 1997 by 16%. In addition, the Neuse river exhibits similar land cover statistics (Table 2) with over a quarter of land in the watershed being cultivated cropland, uncultivated cropland, and pasture. In addition to the higher percentage of agricultural land use in the Neuse watershed, there is also more urban land use as well. This might help explain why the eutrophication and fish kills are so severe in and around New Bern, NC (the mouth of the Neuse River). It should also be noted that the total percent of cultivated cropland had decreased from 1982 to 1997 by 17%. Although these decreases in cropland are seen in both the Tar-Pamlico and Neuse rivers, eutrophication still happens. This could possibly be explained by the ability of nitrogen to be retained in the soil with a slow turnover rate, and thus have negative consequences decades after anthropogenic usage of the land had stopped.[22] In both the Tar-Pamlico and Neuse rivers, the number of animal operations from 1990-1997 had increased substantially, especially swine and poultry (Figure 4).[23,24] The Neuse river watershed also has much larger animal operations compared to the Tar-Pamlico watershed (for example, over 2 million swine vs. 500,000 swine in 1998). Increase in animal operations and livestock have been directly linked to an increase in nitrogen loading and eutrophication of streams and waters.[25,26]

A 

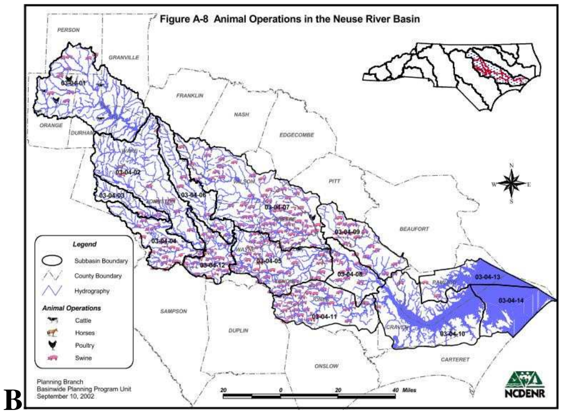
B 

Figure 4: Animal operations in the A) Tar-Pamlico basin and B) Neuse Basin in 2004.[23,24]

	MAJOR WATERSHED AREAS														
	Upper Tar River		Fishing River		Lower Tar River		Pamlico River		Pamlico Sound		1997 TOTALS		1982 TOTALS		% change since 1982
LAND COVER	Acres (1000s)	%	Acres (1000s)	%	Acres (1000s)	%	Acres (1000s)	%	Acres (1000s)	%	Acres (1000s)	% of TOTAL	Acres (1000s)	% of TOTAL	
Cult. Crop	151.4	18.7	126.8	22.4	262.9	39.8	173.4	25.3	55.0	4.5	769.5	19.5	923.2	23.3	-16.6
Uncult. Crop	23.8	2.9	3.9	0.7	0.0	0.0	0.0	0.0	0.0	0.0	27.7	0.7	5.8	0.1	377.6
Pasture	90.5	11.2	17.6	3.1	9.5	1.4	1.3	0.2	0.0	0.0	118.9	3.0	94.9	2.4	25.3
Forest	419.0	51.6	379.1	66.9	286.8	43.4	305.1	44.5	118.1	9.7	1508.1	38.2	1565.1	39.5	-3.6
Urban & Built-Up	66.9	8.2	12.1	2.1	63.3	9.6	27.7	4.0	13.1	1.1	183.1	4.6	96.3	2.4	90.1
Federal	0.0	0.0	0.0	0.0	6.2	0.9	19.9	2.9	98.7	8.1	124.8	3.2	80.7	2.0	54.6
Other	59.7	7.4	27.4	4.8	32.5	4.9	158.1	23.1	937.2	76.7	1214.9	30.8	1196.4	30.2	1.5
Totals	811.3	100.0	566.9	100.0	661.2	100.0	685.5	100.0	1222.1	100.0	3947.0	100.0	3962.4	100.0	
% of Total Basin		20.6		14.4		16.8		17.4		31.0		100.0			
SUBBASINS	03-03-01 03-03-02		03-03-04		03-03-03 03-03-05 03-03-06		03-03-07		03-03-07 03-03-08						
8-Digit Hydraulic Units	03020101		03020102		03020103		03020104		3020105 **						

* = Watershed areas as defined by the 8-Digit Hydraulic Units do not necessarily coincide with subbasin titles used by DWQ.
Source: USDA, Soil Conservation Service - 1982 and 1997 NRI, updated June 2001
** Pasquotank River Subbasin 03-01-55 is contained in hydraulic unit 03020105.
Neuse River Subbasin 03-04-13 is contained in hydraulic unit 03020105.
The hydraulic unit 03020105 is discussed in the Tar-Pamlico River Basin Water Quality Plan.

Table 1: Land Cover in the Tar-Pamlico River Basin by Major Watersheds – 1982 vs. 1997.[23]

LAND COVER	MAJOR WATERSHED AREAS								1997 TOTALS		1982 TOTALS		% change since 1982
	Upper Neuse		Lower Neuse		Contentnea		Lower Neuse						
	Acres (1000s)	%	Acres (1000s)	%	Acres (1000s)	%	Acres (1000s)	%	Acres (1000s)	% TOTAL	Acres (1000s)	% TOTAL	
Cult. Crop	296.7	19.3	208.7	30.7	240.0	38.6	129.3	15.7	874.7	23.9	1054.4	28.8	-17.0
Uncult. Crop	25.4	1.7	16.3	2.4	8.8	1.4	3.4	0.4	53.9	1.5	13.1	0.4	311.5
Pasture	73.2	4.8	44.0	6.5	13.6	2.2	5.4	0.7	136.2	3.7	116.7	3.2	16.7
Forest	684.1	44.6	330.8	48.7	269.7	43.3	356.9	43.4	1641.5	44.9	1769.4	48.3	-7.2
Urban & Built-Up	349.7	22.8	47.7	7.0	48.1	7.7	35.5	4.3	481.0	13.1	254.1	6.9	89.3
Federal	5.8	0.4	2.9	0.4	0.0	0.0	75.1	9.1	83.8	2.3	75.1	2.0	11.6
Other	99.4	6.5	29.2	4.3	42.3	6.8	216.0	26.3	386.9	10.6	381.0	10.4	1.5
Totals	1534.3	100.0	679.6	100.0	622.5	100.0	821.6	100.0	3658.0	100.0	3663.8	100.0	
% of Total Basin		41.9		18.5		17.0		22.4		99.8			
SUBBASINS	03-04-01 03-04-02 03-04-03 03-04-04 03-04-06 03-04-12		03-04-05 03-04-08 03-04-09		03-04-07		03-04-10 03-04-11						
8-Digit Hydraulic Units	03020201		03020202		03020203		03020204						

* = Watershed areas as defined by the 8-Digit Hydraulic Units do not necessarily coincide with subbasin titles used by DWQ.

Source: USDA, Soil Conservation Service - 1982 and 1997 NRI

** 270 square miles of Neuse River subbasin 03-04-13 is contained in hydrologic unit 03020105. The hydrologic unit 03020105 is discussed in the Tar-Pamlico River Basin Water Quality Plan. 336 square miles of Neuse River subbasin 03-04-14 is contained in hydrologic unit 03020106. The hydrologic unit 03020106 is discussed in the White Oak River Basin Water Quality Plan. It is not currently feasible to estimate the land use in these portions to include the Neuse land cover estimates.

Table 2: Land Cover in the Neuse River Basin by Major Watersheds – 1982 vs. 1997.[24]

Current Measures

There are several current measures, both at the federal and state levels, to help mitigate and/or eliminate the effects of nutrient loading in the Tar-Pamlico and Neuse river basins. The Neuse River and Tar-Pam River were deemed Nutrient Sensitive Waters (NSW) in 1989 by the NC Environmental Management Commission (EMC). A NSW has exceeding high nutrient and *chlorophyll a* levels. In 1997, the EMC adopted a new Agriculture Rule to help the nutrient sensitive water management programs in place. The rule stated that all persons engaging in agricultural procedures most participate in a local nitrogen reduction strategy program and implement standard best management practices (BMPs), with the target goal of a 30% reduction in N loss from agricultural lands and no increase in P loss from agricultural lands with respect to 1991 levels. Local Advisory Committees (LACs) and a watershed Basin Oversight Committee (Neuse and Tar-Pam BOCs) were also formed to help implement and monitor/report nutrient sensitive water management programs. Since reported starting in 2001, the30% reduction goal target has been far surpassed in both the Neuse and Tar-Pam rivers. According to the 2010 crop year Annual Progress Reports on the Neuse and Tar-Pam from the BOCs, the Neuse achieved a 49% decrease in N loss and the Tar-Pam achieved a 52% decrease in N loss (Table 3). A reduction in P loss risk in the Tar-Pam is also seen. [27,28]

So what exactly is being done to achieve this reduction? As stated earlier, the Agriculture Rule helped vastly reduce the amount of N loss from agricultural lands through BMP implementation and nitrogen reduction strategies. Furthermore, the State of North Carolina passed Senate Bill 1217 in 1997 in an effort to put stricter regulation on animal waste disposal. All animal operations were required to have animal waste management plans by January 1998. The plan must address three specific items: 1) periodic testing of soils where waste is applied; 2) development of waste utilization plans; and 3) completion and maintenance of records on-site for three years. In addition, House Bill 515 passed in 1997 and placed a moratorium on the construction of new swine farms. These efforts at the state level were put in place to help curtail the effects of animal (especially swine) waste on the watershed.

Estimated Reductions in Agricultural Nitrogen Loss from Baseline (1991-1995) for 2008 (NLEW v5.32) and 2009 (NLEW v5.32), Neuse River Basin

County	Baseline N Loss (lb)* NLEW v5.30	Proposed %	2009 N Loss (lb)* NLEW v5.32	2009 Reported N Loss (%) NLEW v5.32	2010 N Loss (lb)* NLEW v5.33	2010 Reported N Loss (%) NLEW v5.33
Carteret	1,274,541	>30%	837,941	34%	855,718	33%
Craven	3,893,265	>30%	2,005,428	48%	1,466,600	62%
Durham	167,317	>30%	66,257	60%	73,080	56%
Franklin	149,640	30%	54,578	64%	38,054	75%
Granville	156,004	21%	69,876	55%	73,302	53%
Greene	4,046,251	30%	1,883,835	53%	1,550,864	62%
Johnston	6,303,200	30%	4,030,436	36%	2,997,037	53%
Jones	2,898,537	>30%	1,731,973	40%	1,436,983	50%
Lenoir	4,008,721	>30%	3,232,002	17%	2,815,832	30%
Nash	*1,115,518*	30%	418,876	62%	516,595	54%
Orange	436,356	18%	172,860	60%	163,429	63%
Pamlico	2,393,849	>30%	1,507,440	37%	1,564,266	35%
Person	534,043	26%	337,362	37%	155,579	71%
Pitt	3,053,543	30%	1,167,907	62%	1,229,109	60%
Wake	1,219,036	30%	237,917	80%	236,284	81%
Wayne	7,786,992	30%	3,921,165	50%	4,538,240	42%
Wilson	2,614,048	>30%	1,693,192	35%	1,560,653	40%
Total	42,050,861		23,369,045	44%	21,271,625	49%

**Nitrogen loss values are for comparative purposes. They represent nitrogen that was applied to agricultural lands in the basin and neither used by crops nor intercepted by BMPs in a Soil Management Unit, based on NLEW calculations. This is not an in-stream loading value.*

Estimated Reductions in Agricultural Nitrogen Loss from Baseline (1991) for CY2010 (NLEW v5.52) and CY2010 (NLEW v5.53), Tar-Pamlico River Basin

County	Baseline N Loss (lb)* NLEW v5.51	CY2009 N Loss (lb)* NLEW v5.52	2009 Reported N Loss (%) NLEW v5.52	CY2010 N Loss (lb)* NLEW v5.53	2010 Reported N Loss (%) NLEW v5.53
Beaufort	8,811,875	4,944,627	44%	5,081,141	42%
Edgecombe	5,103,502	3,332,444	35%	3,053,849	40%
Franklin	1,993,925	639,206	68%	556,448	72%
Granville	971,365	344,791	65%	418,580	57%
Halifax	2,819,301	1,449,612	49%	1,634,622	42%
Hyde	4,861,387	2,850,975	41%	2,822,212	42%
Martin	825,278	485,331	41%	470,744	43%
Nash	4,658,164	1,488,684	68%	1,640,068	65%
Person	168,038	82,829	51%	38,208	77%
Pitt	5,966,245	2,650,499	56%	1,946,405	67%
Vance	449,753	107,094	76%	123,570	73%
Warren	610,045	116,501	81%	146,126	76%
Washington	898,346	487,115	46%	548,183	39%
Wilson	780,741	379,478	51%	386,832	50%
Total	38,917,965	19,359,186	50%	18,866,988	52%

**Nitrogen loss values are for comparative purposes. They represent nitrogen that was applied to agricultural lands in the basin and neither used by crops nor intercepted by BMPs in a Soil Management Unit, based on NLEW calculations. This is not an in-stream loading value.*

Table 3: CY2010 and CY2009 reductions in N loss for the Tar-Pam and Neuse basins by county[27, 28]

The BOCs also characterized the factors that influenced N loss by percentage for the Neuse and Tar-Pam basins (Table 4). For the Tar-Pam basin, fertilization management proved to be the biggest factor, followed by cropping shifts, BMP implementation, and finally loss of cropland (various reasons). For the Neuse basin, the biggest factor was cropping shifts, then followed by fertilization management, loss of cropland, and finally BMP implementation. Fertilization management can be characterized by increased cost of fertilizer ($180 per ton in 1989 to $783 per ton in 2012 for anhydrous ammonia), nutrient management training, and crop rotation. Like the Agriculture Rule, the EMC adopted a

Nutrient Management Rule, which applied to all persons applying fertilizer on land greater than 50 acres. The Nutrient Management Rules states that the person must complete training in a nutrient management program and develop a nutrient management plan for their land. Overall, this lead to a reduction in fertilization rates (N lbs/acre) from 2007 to 2010. [27,28] Cropping shifts can be characterized by increased commodity prices along with crop rotations.

Factors That Influence Nitrogen Reduction by Percentage on Agricultural Lands, Neuse River Basin

	2007 NLEW v5.30	2008 NLEW v5.32	2009 NLEW v5.32	2010 NLEW v5.33
BMP implementation	6%	5%	7%	9%
Fertilization management	10%	12%	14%	12%
Cropping shift	7%	10%	8%	13%
Cropland converted to grass/trees	2%	1%	1.5%	2%
Cropland lost to idle land	7%	6%	6.5%	6%
Cropland lost to development	7%	7%	7%	7%
Total	39%	41%	44%	49%

**Percentages are based on a total of the reduction, not a year to year comparison.*

Factors that Influence Nitrogen Reduction by Percentage on Agricultural Lands, Tar-Pamlico River Basin

	CY2007 NLEW V5.51	CY2008 NLEW V5.52	CY2009 NLEW V5.52	CY2010 NLEW V5.53
BMP implementation	10%	10%	11%	12%
Fertilization Management	20%	21%	20%	20%
Cropping shifts	8%	10%	11%	13%
Reduction in cropland due to idle land	3%	4%	3.5%	3%
Reduction in cropland due to cropland conversion	2%	4%	3.5%	3%
Reduction in cropland due to development	1%	1%	1%	1%
TOTAL	**44%**	**50%**	**50%**	**52%**

Table 4: Factors that influence nitrogen reduction by percentage on agricultural lands for the Neuse and Tar-Pam basins. [27, 28]

BMPs, as seen in Table 4, also contribute greatly to the reduction in N and P loss from agricultural practices. The most common BMPs utilized are riparian buffers (20' and 100' mostly, however 30', 50', and 70' are also used), water control structures, and scavenger crops. There are several other BMPs that are either not utilized or utilized but not recorded by the BOCs (diversion, livestock exclusion, conservation tillage, fencing, sod-based rotation, etc). These all contribute to N and P loss reduction, but it is not exactly sure how much they contribute. Thus N and P loss from agricultural lands has greatly diminished due to several factors, most importantly reduction/management of fertilization, farmer education/training, cropping shifts, BMP implementation, and livestock waste management.

Further Measures

Biosolids serve as an attractive option for fertilizing agricultural fields due to skyrocketing fertilizer prices and the ability to cut down on adding new N to the environment via commercial fertilizers. Using recycled biosolids would definitely help in reducing N and P TMDLs in river systems such as the Neuse and Tar-Pam. The US EPA provides educational information about the use and recycling of biosolids, as well as a program called the National Biosolids Partnership that promotes effective management and use of biosolids.

As stated earlier, BMPs proved to be a valuable and efficient means of reducing N and P in the Tar-Pam and Neuse basins. However, not all farmers utilize BMPs or utilize the most efficient BMPs. There is a huge range of different BMPs, and knowing which ones are the best and/or most efficient for a certain agriculture practice or procedure is critical. There are several programs at the state and national levels that help farmers determine what BMPs are best for them. North Carolina has an Agriculture Cost Share Program (ACSP). The ACSP works with farmers at the district level by identifying the best BMPs suited for that agricultural operation, help develop conservation plans, and subsequently offering the proper technical assistance for installation, operation, and maintenance. At the state and national level, there is the USDA Natural Resources Conservation Service (NRCS). The NRCS helps farmers develop and implement complete conservation plans by conducting thorough resource inventory, geology and watershed data analysis, crop production potential, etc.

Conclusions

Large steps have been taken in agriculture to help reduce N and P loading in the Tar-Pam and Neuse river basins and impressive results have been noted. However, these two watersheds are still at extremely high levels of nutrients and still exhibit high levels of eutrophication. Not all methods and practices for reducing N and P loss in agriculture have been implemented and/or explored, thus leaving much room for future improvement in management and conservation techniques. Furthermore, there are a myriad of ways in which excessive nutrients can be introduced into these river systems, and more research needs to be done on the percentage and proportions of nutrient loading that trace back to wastewater, urban runoff, agriculture and other non-point and point sources; and whether or not some of these sources are lagging behind in management practices. Clearly more must be done to help stem the influx of nutrients into the Tar-Pam, Neuse, and other river systems found throughout the country.

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Impediments to Innovative Green Stormwater Practices: A Review of North Carolina's Policies and Suggestions for the Future

Amanda Santoni

As the United States has become more urbanized, there has been an increase in the amount of stormwater runoff reaching the nation's water bodies. Stormwater discharge is a large source of various types of pollution such as thermal pollution (Foulquier et al, 2009), mercury (Evers et al, 2005), sediment (Kim et al, 2006) and pesticides (Hageman et al, 2006). Recently there has also been an increase in studies examining the presence of polybrominated diphenyl ethers (PBDEs) and polychlorinated biphenyl (PCBs) from runoff, which may have large negative effects on the human population (Harrad and Hunter, 2006; Hwang and Foster, 2008). Traditionally, stormwater has been discharged directly into water bodies such as streams or lakes through the widespread use of pipes. Only recently has society begun to notice the repercussions of these actions.

In response to the consequences of discharging stormwater directly into streams, innovative practices known as green stormwater infrastructure (GSI) have been developed. GSI is any technique that deals with stormwater at the source, through methods such as infiltration, retention, uptake by plants, or other methods rather than treating the water after it has entered a sewer or stormwater system (Valentine, 2007). This is generally achieved through simulating natural processes, rather than building extensive metal or other impermeable infrastructure. Types of GSI include green roofs (Fig. 1A), living shorelines (Fig, 1B), permeable pavement (Fig. 1C), bio-swales (Fig. 1D), infiltration basins (Fig. 1E), wetland construction (Fig. 1F), rain gardens (Fig. 1G), and stormwater ponds (Fig. 1H). The benefits of GSI are vast and include economic, ecological, and community advantages. GSI may reduce water treatment needs, improve water quality, decrease flooding, recharge groundwater supplies, reduce urban heat islands, as well as improve habitat for wildlife and humans alike (CNT, 2010; Fig. 2). Different types of GSI have varying functions, costs, and effectiveness. One of the most frequently implemented forms of GSI is the stormwater retention pond because it is simple and inexpensive. However, stormwater ponds often do not reduce sediments and nutrients to the extent needed (Harper & Baker, 2007), and it is therefore necessary to implement more innovative GSI solutions to stormwater runoff.

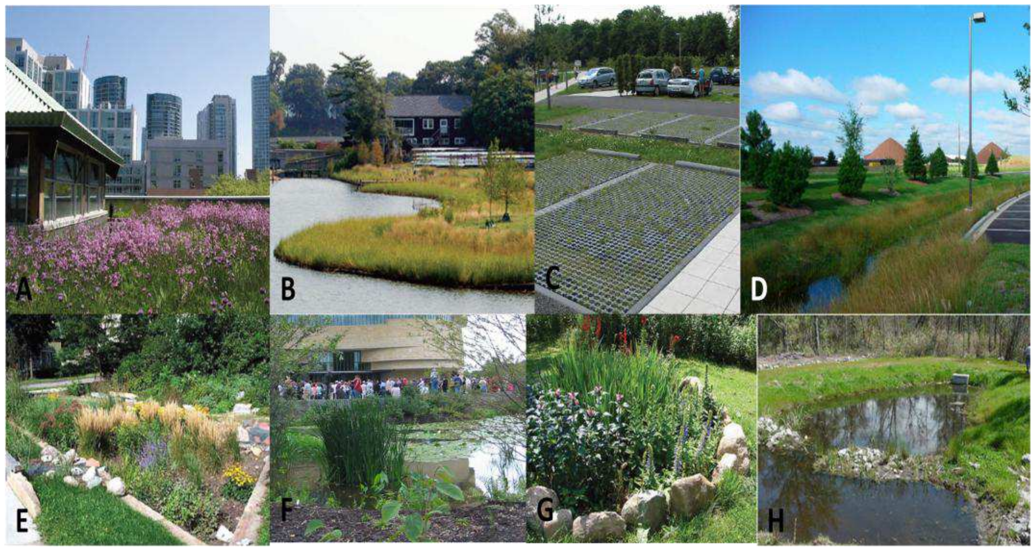

Figure 1 - A. Green roof (CNT, 2010) B. Living shoreline (CBF, 2007) C. Permeable pavers (CNT, 2010) D. Bio-swale (Lake County Stormwater Commission) E. Infiltration basin (CNT, 2010) F. Wetland Construction (NCDENR, 2007) G. Rain gardens (CNT, 2010) H. Stormwater ponds (NCDENR, 2007)

Although a myriad of green infrastructure choices are available, they are rarely implemented (Thurston, 2011). The United States Environmental Protection Agency (EPA) oversees federal laws such as the Clean

Water Act (CWA), Phase I & II Stormwater Permit Regulations, and the National Pollution Discharge Elimination System (NPDES) to deal with water quality issues. However, these laws only regulate point sources of pollution, or those that are attributed to a single source. Point sources are generally those that use pipes to discharge water directly into a body of water. This leaves the regulation of non-point source pollution such as stormwater runoff to individual states and municipalities (National Resource Council, 2009). The states face a multitude of barriers to the implementation of GSI, and some have been more successful than others. Despite the generally environmentally friendly consciousness in North Carolina, there is still a lack of innovative GSI in the state. This paper offers an overview of general barriers to innovative GSI implementation, an examination of Durham, North Carolina's downfalls in implementation, examples of how some states have overcome these barriers, and suggestions for North Carolina policy to increase the use of GSI in the future.

Common Barriers to Green Stormwater Infrastructure (GSI)

Each state has different ways of adhering to the federal Clean Water Act and Stormwater Permit Regulations, while advocating for GSI to varying degrees. Some states opt for more "command and control" style regulations, while others offer market-based solutions such as implementing tax breaks or grants and subsidies to decrease the initial costs (Carter and Fowler, 2008). A study by Valentine (2007) found that the main inhibitors of GSI are the cost, the deficiency in studies on the actual performance of these structures, as well as the detached nature of these tactics as opposed to a more centralized and city-wide stormwater system.

Benefit	Reduces Stormwater Runoff											Improves Community Livability						
	Reduces Water Treatment Needs	Improves Water Quality	Reduces Grey Infrastructure Needs	Reduces Flooding	Increases Available Water Supply	Increases Groundwater Recharge	Reduces Salt Use	Reduces Energy Use	Improves Air Quality	Reduces Atmospheric CO_2	Reduces Urban Heat Island	Improves Aesthetics	Increases Recreational Opportunity	Reduces Noise Pollution	Improves Community Cohesion	Urban Agriculture	Improves Habitat	Cultivates Public Education Opportunities
Practice										CO_2								
Green Roofs	Yes	Yes	Yes	Yes	No	No	No	Yes	Yes	Yes	Yes	Yes	Maybe	Yes	Maybe	Maybe	Yes	Yes
Tree Planting	Yes	Yes	Yes	Yes	No	Maybe	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Maybe	Yes	Yes
Bioretention & Infiltration	Yes	Yes	Yes	Yes	Maybe	Maybe	No	No	Yes	Yes	Yes	Yes	Yes	Maybe	Maybe	No	Yes	Yes
Permeable Pavement	Yes	Yes	Yes	Yes	No	Maybe	Yes	Maybe	Yes	Yes	Yes	No	No	Yes	No	No	No	Yes
Water Harvesting	Yes	Yes	Yes	Yes	Yes	Maybe	No	Maybe	Maybe	Maybe	No	No	No	No	No	No	No	Yes

Yes Maybe No

Figure 2 - The potential benefits from various forms of green stormwater infrastructure (From CNT, 2010).

Another study by Roy et al (2008) identified seven main reasons why GSI is not implemented regularly. Roy et al maintain that GSI is a large risk when it comes to the performance and cost of these structures. Additionally, in many cases there are not existing manuals describing the standards and guidelines for implementation. Often, the regulation is disjointed, or lacks "institutional capacity."Additionally, there are few laws requiring GSI, few incentives, and a large-scale "resistance to change."

Because each state, as well as each municipality within each state can enact their own policies, the laws may vary at different locations within a state. This becomes exceedingly complex, and problematic when attempting to develop land while being environmentally friendly (Jaffe et al, 2010). Therefore, some states and cities have been more successful than others at implementing GSI. Despite the fact that North Carolina is a generally environmentally friendly state, it has not been able to implement GSI to its fullest extent statewide, or even citywide.

North Carolina GSI Implementation

In North Carolina, the main barriers to implementation of GSI are the permitting process, the lack of legislative mandate and incentives, as well as insufficient guidelines and standards. Permitting is by far the most obvious process that inhibits the ability to implement GSI on a larger scale.

As Roy et al (2008) suggest, the lack of institutional capacity is a problem and is seen in the permitting process in North Carolina. In NC, there are many types of permits required for different activities, with various approval processes for each. In order to install a traditional sediment control or shore stabilization technique on the NC coast, such as a bulkhead or rip-rap, a general CAMA (Coastal Area Management Act) permit is required. A general CAMA permit is relatively easy to file, and may be issued on site by the North Carolina Department of the Environment and Natural Resources (NCDENR). Conversely, for an innovative green design such as a living shoreline, a major permit is needed because it involves some supplementation of sand or substrate and the addition of a rock sill slightly off shore, which will alter the shoreline. A major permit may take up to 150 days (75 days plus a 75-day extension. This permit must be reviewed by 10 different state and federal agencies. Another major obstruction in the permitting process is the cost of a permit, beyond the price of actually implementing the design, as Valentine (2007) mentioned. The minor CAMA permit for structures such as rip-rap or a bulkhead cost $200-400, but is usually $200 for sediment and erosion control structures (NCDENR, 2007). A major permit for a structure that restores natural processes to protect shorelines from erosion, such as a living shoreline can be $250-$475. The price difference between the major and minor permits may not seem large, but in the planning process, the extra cost of the permit further exasperates the initial cost of GSI as opposed to a more traditional technique. The increase in cost, as well as lost time and hassle involved with executing an innovative green solution is a major deterrent of green infrastructure for those who wish to develop or redevelop land in NC.

The second main barrier to implementing GSI in North Carolina is the lack of legislative mandate and a shortage of incentives for developers. Currently, there are very few laws in North Carolina that require that developers implement GSI. Under NPDES Phase I and II, there are six cities in North Carolina that must follow a stormwater management plan including Raleigh, Durham, Fayetteville/ Cumberland County, Charlotte, Winston-Salem, and Greensboro (NCDENR, 2007). Each of these cities has implemented their own stormwater management plan, which includes stormwater runoff control during construction, as well as after development. The stipulations for controlling runoff emphasize the use of vegetated conveyances "to the maximum extent practicable" (NCDENR, 2007) and to deal with excess runoff created by the development on-site. This is rather open to interpretation where vegetated conveyances are practicable, and do not necessarily require the use of GSI. When GSI is implemented under this system, it is also generally in the form of a stormwater pond, due to low cost and not functionality. These six cities do have a stormwater fee, which charges land owners for the amount of impervious surface on their land (McDonough, 2007). While this may seem as though it would significantly decrease the amount of impervious surface in new developments, the fee is rather low, and many businesses can afford to pay for large areas of imperviousness.

In other areas, not under NPDES, there are long-term watershed restoration projects, setting limits on nutrient and sediment loading from runoff for four watersheds in North Carolina: Jordan, Falls, Tar-Pamlico, and Neuse (NC Session Law 2009-486, Senate Bill 1020). Durham, NC spans two watersheds, the Falls Lake (Fig. 3) watershed to the north, and the Jordan Lake (Fig. 4) watershed to the south. Both of these watersheds are have limits on nutrients and sediment loading, Falls Lake under 15A NCAC 02B.0277 and Jordan Lake under 15A NCAC 02B.0265. These laws place local governments in charge of developing stormwater management programs for these areas. The Falls Lake watershed has a goal of reducing nitrogen by 40%, and phosphorus by 70% of the 2006 baseline. This will be achieved by requiring plans to be approved for new development areas more than 0.5 acres for residential properties and 12,000 square feet (0.275 acres) for commercial properties and implementing BMPs at these sites. The Jordan Lake watershed has different nutrient reduction targets for each of the main rivers flowing into the basin. The Jordan Lake watershed requires stormwater management plans to be established for new development affecting one acre or more of residential, and 0.5 acres for commercial property. Both watershed laws mandate that stormwater systems

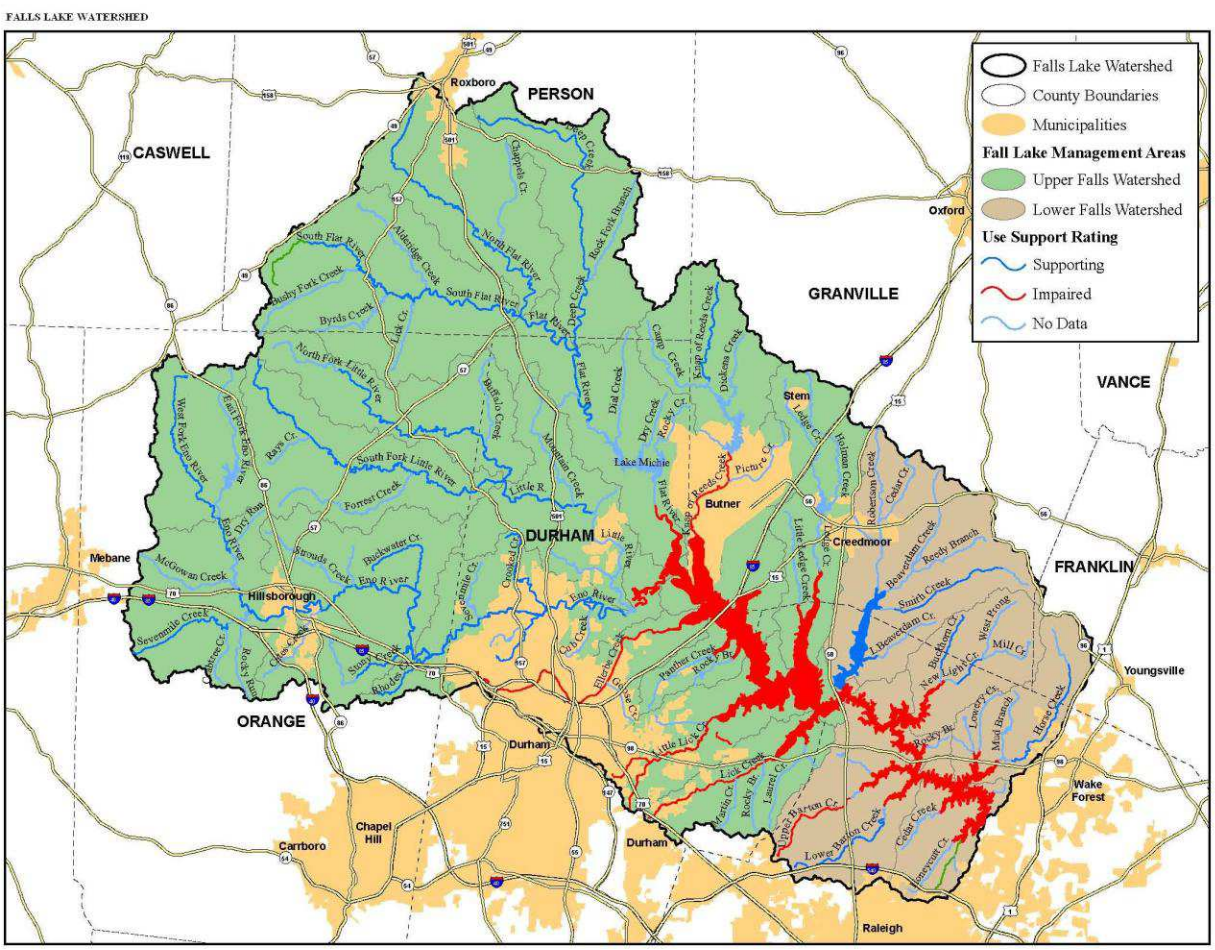

Figure 3. Falls Lake watershed. (NCDENR.org)

retain and treat runoff generated by an inch of rainfall. However, there are no mandates on the exact methods of stormwater, nutrient, and sediment reduction, and there is the option to offset any pollution with mitigation elsewhere.

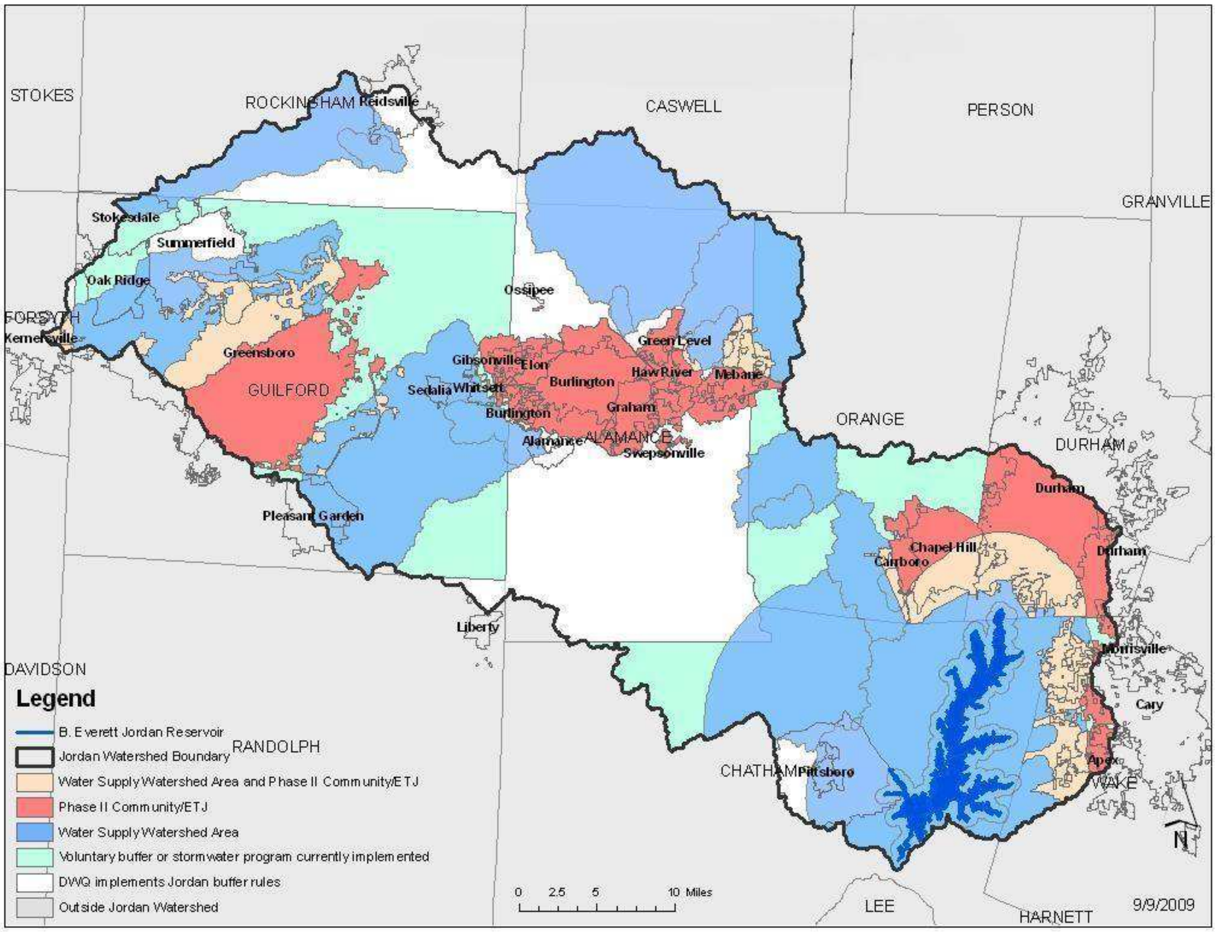

Figure 4. Jordan Lake watershed. (NCDENR.org)

Despite the laws that are in place, there are still over 900 streams in North Carolina that qualify as impaired, and only 125 of them are monitored by the state total maximum daily load (TMDL) (Berg and BenDor, 2010). The lack of incentives for developers to use green systems may partially account for this difference. Incentives may be in the form of direct aid, such as a grant or subsidy for green infrastructure, or may be in the form of an indirect aid such as a tax break or decrease in some fee (Carter and Fowler, 2008). In North Carolina, direct incentives for GSIs are scarce. One from the Community Conservation Assistance Program (CCAP) will pay for up to 75% reimbursement of a stormwater Best Management Practice (BMP) in NC (NCDENR, 2007). Additionally, the NCDENR 319 grant program offers funding for structures that reduce non-point source pollution. As far as indirect aid, the state of North Carolina does not offer any options. However, it is not all bad news for North Carolina, as some cities such as Durham do have a reduced stormwater fee for decreasing the amount of stormwater from new development.

The last major barrier that North Carolina has that is inhibiting the development of green solutions to stormwater is the insufficient guidelines and standards of the state. The North Carolina Division of Water Quality had published a Stormwater Best Management Practices (BMP) Manual (2007). This helps to establish guidelines for the state procedures for implementing green infrastructure, but the manual is incomplete. The chapter focusing on BMP construction techniques has not yet been written. The manual does include chapters on stormwater wetlands, wet detention basins, sand filters, bio-retention, grassed swales, restored riparian buffers, infiltration devices, dry detention basins, permeable pavement, rooftop runoff, and proprietary systems. While all of these prevent stormwater and associated pollutants from reaching bodies of water, not all of them qualify as GSI. The manual still included many structural solutions using metal conveyance and treatment that does not mimic natural processes such as proprietary systems. Although the manual does ignore some types of green infrastructure completely, such as living shorelines, it is effective at highlighting examples of BMPs.

How Other States and Municipalities Have Overcome Barriers

GSI has been more successful in Maryland, Portland Oregon (Kulkarni, 2012), Maine (Jaffe et al, 2010), Chicago Illinois, Philadelphia Pennsylvania, Milwaukee Wisconsin, and Seattle Washington (Valentine, 2007). The lack of institutional capacity (Roy et al, 2008) has been addressed by various states and cities throughout the U.S. For example, Portland Oregon has established a simplified permitting process for green initiatives (Kulkarmi, 2012) that allows for faster permitting. Maryland has also launched a joint permitting initiative with the Federal permitting system in order to process permits that require both state and federal approval before issuing a permit (MDE). Also in Maryland, the permitting costs of an erosion control infrastructure, green or not, even a living shoreline is $250 (MDE). Chicago Illinois has streamlined their green roof permitting process, and currently has more than 400 green roof projects underway (CNT, 2010)

Other cities and states have been able to overcome the lack of legislative mandates and incentives that are frequently found throughout the nation. Portland Oregon requires the use of green infrastructure techniques if development of land is more than 500 square feet (Kulkarmi, 2012). Additionally, new buildings that are city owned must have a green roof with no less than 70% vegetation (Carter and Fowler, 2008). Chicago also has a similar law that new buildings receiving public money must have some green roof or Leadership in Energy and Environmental Design (LEED) aspects (Valentine, 2007). Chicago also requires any building that is more than 15,000 square feet, or any parking lot that is more than 75,000 square feet to detain the first ½ inch of rain on site. Philadelphia requires buildings of this size to detain the first inch. Maryland has gone beyond this ½ inch of rain, and requires that water from a one year 24 hour storm be retained onsite (Cappuccitti and Page, 2000). Currently, the four North Carolina nutrient and sediment reduction watersheds are on par with Philadelphia, but slightly behind Maryland.

As far as incentives, Maryland has a great deal of grants available through the Chesapeake Bay Foundation, the Chesapeake Bay Trust, the Maryland Department of the Environment (MDE) Water and Management Administration, MDE Tidal Wetlands Division, and interest free loans from the MDE Water Quality Financing Administration. Maine has put four million dollars toward their state revolving fund for green projects (Jaffe et al, 2010). The city of Seattle set aside $7.4 million for green infrastructure and The Milwaukee Metropolitan Sewerage District earmarked $8.8 million for green infrastructure in 2007 alone (Valentine, 2007). Indirect incentives are an important resource as well. For example, Portland Oregon offers to reduce the stormwater fee for those buildings that install green roofs or extra green space beyond the required 70% green roof coverage (Carter and Fowler, 2008).

Maryland has also addressed the issue of guidelines and standards well. Like NC, Maryland has a Stormwater Design Manual (2000), but also and Environmental Site Design (ESD) Process and Computation Manual (2010) and has collaborated with the conservation fund and the state of Virginia to produce a guide entitled *A Sustainable Chesapeake* (2010). Maryland's Stormwater Design Manual is complete, and has fourteen standards on what needs to be addressed when constructing a BMP. It also includes sections on selecting the best sites and type of BMP for the project. The manual includes specific requirements for conveyance, landscaping, and treatment. The manual promotes what they call Environmental Site Design (ESD), which is the use or mimicking of natural processes. The guide, *A Sustainable Chesapeake* (2010) is a book that includes many pictures of BMPs and BMP construction as well as successful BMPs from which to model other systems. Portland Oregon and Chicago Illinois each have their own manuals that are much stricter than their state regulations.

Recommendations for the State of North Carolina and the City of Durham

The state of North Carolina needs to make it easier for developers in the state to apply for and receive permits for green infrastructure. It should follow in the steps of Maryland and Portland Oregon in designing a streamlined permitting system for those that are applying to build GSI. Permits for these systems should not cost more money than those that are detrimental to the environment, or the simplest option. If green infrastructure is made to cost less in the permitting system, more developers might be interested in installing these systems. Additionally, a Triple bottom line assessment between GSI and traditional infrastructure should be conducted for large cities in North Carolina such as Durham. This tactic was useful for launching GSI initiatives in Philadelphia, PA because GSI was found to provide $2,724.4 million more in benefits than traditional infrastructure (CNT, 2010).

North Carolina should also institute a balance of requirements for GSI and incentives for these systems. This balance has worked well in Oregon, where large cities such as Portland are required to have green infrastructure such as green roofs if they are over a certain size. However, setting requirements alone cannot solve the problem. These requirements are very difficult to pass, and while requiring GSI will indeed increase the amount of green infrastructure, it may not ensure quality of design if it is not regulated properly (Carter and Fowler, 2008). Incentives, especially direct incentives are needed as well for voluntary implementation of green infrastructure. The reduced cost will break through the price barrier that is often the limiting factor when designing a green stormwater control system. Additionally, making green infrastructure voluntary outside of city centers will ensure that the green infrastructure being built will be considered thoroughly and be a quality model for other projects. North Carolina should work hard to set up more grant systems such as the state of Maryland and Portland, OR for green infrastructure.

The state of North Carolina should also work to create more resources for the guidelines and standards of how GSI should be constructed. The NC Stormwater Best Management Practices (BMP) Manual (2007) should be completed to include information on the BMP construction techniques. It should also be updated to include standards to which the implementation of BMPs should be held as well as more figures that show the different types of BMPs. It should also advocate for green solutions over structural solutions such as proprietary systems and water filtration. It would also be useful to include information and graphics showing successful implementation of each type of BMP like the guide *A Sustainable Chesapeake* in Maryland and Virginia.

Research and education on green stormwater infiltration should also be continued in order to more fully advocate for these systems. However, change will come most rapidly through the implementation of requirements, incentives, permitting reform, and more clear engineering standards and guidelines on GSI. With these considerations, North Carolina and the city of Durham can become leaders in stormwater management and continue to leave a legacy of environmental stewardship for future generations. ■

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Using Constructed Wetlands to Emphasize the Importance of Outreach and Education in Stormwater Management

Faith M. M. Middleton

1. Introduction & Background

As population size continues to increase, so does the amount of urbanization. In 2010, the World Health Organization (WHO) indicated that more than half of all people live in urban areas and projections suggest that by 2030, 6 out of every 10 people will live in a city (WHO, 2013). The problem with urbanization is that it introduces a wide variety of environmental problems; the most prominent of these being water pollution due to stormwater runoff from impervious surfaces. Stormwater can be defined as the water from either rainfall or snowfall events that flows across land into rivers, creeks, lakes, ditches, canals, etc. (Adams County Government, 2013). As a result, debris, sediment, pollution, bacteria, and various nutrients can end up in nearby bodies of water. Part of this runoff is usually absorbed by infiltration, but increases in urbanization lead to more impervious surface which decreases infiltration.

Traditional stormwater planning has a tendency to focus on moving large amounts of runoff from an area with the use of ditches and pipes that divert it into nearby streams. This approach can lead to issues such as flooding, erosion, sedimentation, and contamination. These are most likely the result of altered stream hydrology, temperature, and water quality. Previous studies have shown that stormwater can alter stream hydrology by increasing flood volume and flood peak (Leopold, 1968) and that increases in stream temperature can alter ecosystems. For example, it can lead to habitat degradation for cold water fish (Herb et. al., 2008). Studies also show that runoff leads to decreased water quality (Leopold, 1968). Runoff may decrease water quality because it can cause the accumulation of mercury (Evers et. al., 2005), pesticides and semi-volatile organic compounds (Hageman et. al., 2006), various metals (i. e. Zn, Cu, and Pb) (Sansalone et. al., 1997), and dissolved solids (Leopold, 1968). It can even decrease dissolved oxygen concentrations (Leopold, 1968).

Stormwater management practices generally try to reduce and control the negative effects of stormwater with the help of either remediation strategies or prevention strategies. Prevention strategies are designed to prevent stormwater runoff based on the idea that natural environments produce less runoff than artificial environments (UW-Madison, 2009). Techniques for prevention include the use of rain gardens, green roofs, rooftop gardens, buffer zones, parking lot and road designs, and traditional zoning (UW-Madison, 2009). Remediation strategies are designed to reduce problems associated with stormwater runoff (i.e. pollutant levels) before it reaches body of water (UW-Madison, 2009). Remediation techniques are most effective when combined and they include conveyance, street cleaning, catch basins, infiltration trenches, constructed wetlands, and wet ponds (UW-Madison, 2009).

This paper focuses on the use of constructed wetlands as a stormwater management practice to help remove pollutants, control flow, as well as educate citizens about stormwater and influence their attitudes towards it. The proposed South Ellerbe constructed wetland could be the first step towards achieving this objective in Durham, NC.

2. Constructed Wetlands

Constructed wetlands are currently used in Durham, but the existing wetlands are relatively small in scale and tend to be located on private property. For instance, a few well known wetlands in Durham are located at a local Hampton Inn, the Duke Smith Warehouse, the Ellerbe Creek Preserve, and Southpoint Terrace [See Figure 1] (Durham Government, 2013).

Figure 1 - Examples of small privately owned constructed wetlands in Durham (Durham Government 2013)

2.1 Types of Constructed Wetlands -

While the proposed wetland concept is a good start, there is room for improvement. To get a better idea of where improvements can be made, it is important to look at the types of wetlands that can be constructed. Wetlands can be divided into the following major categories: Free water surface constructed wetlands, subsurface constructed wetlands, and hybrid constructed wetlands [See Figure 2].

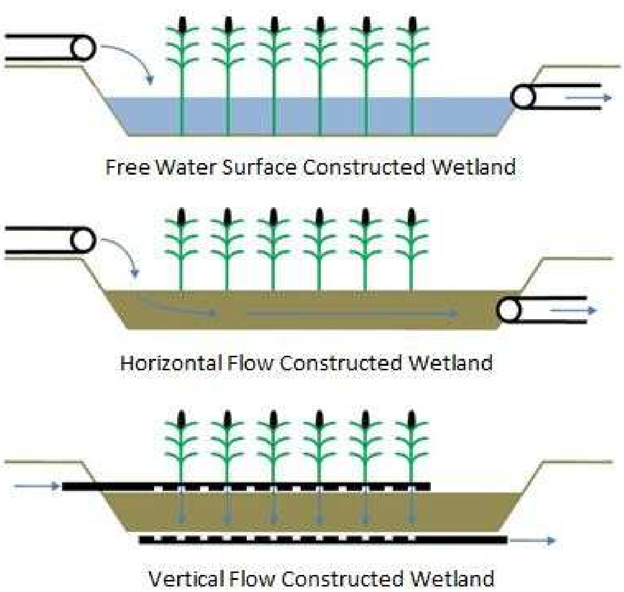

Figure 2 - Basic diagram of the three main wetland types (Wikipedia 2013)

2.1.1 Free water surface constructed wetlands (FWS CW)

FWS CWs contain areas of open water and floating, submerged, and emergent plants. As water flows through the wetland, it is treated by physical, chemical, and biological processes (Vymazal, 2011). For example, organics are removed through microbial degradation and suspended solids via sedimentation. Nitrogen removal is variable and depends on a variety of factors such as the chemical form of nitrogen, water temperature, dissolved oxygen concentration, and the availability of organic carbon. Ammonia is best removed using nitrification and phosphorus removal occurs through adsorption and precipitation (Vymazal, 2011). While phosphorus removal can be substantial, it occurs at a slow rate and is limited by the contact between the water column and the soil. FWS CWs are used most commonly for tertiary treatment of municipal wastewater and stormwater runoff and they can provide great storm/surge capacity (Vymazal, 2011).

2.1.2 Subsurface constructed wetlands

Horizontal flow constructed wetlands (HF CW):

In HF CWs, incoming wastewater flows through porous medium under the surface of the bed planted with emergent vegetation towards an outlet where it is collected before leaving via a water level control structure (Vymazal, 2011). Macrophytes are an integral part of this wetland because they help with filtration bed insulation, provide substrate for bacteria to grow on, release oxygen into the rhizophere, uptake and store nutrients, and release substances from their roots with antimicrobial properties (Vymazal, 2011). HF CWs are commonly used for secondary treatment of wastewater and they effectively remove organics, suspended solids, microbial pollution, and heavy metals as the water flows through aerobic, anoxic, and anaerobic zones (Vymazal, 2011). Organic compounds are degraded by bacteria in anaerobic and aerobic zones, while suspended solids settle into the spaces of the filtration bed to be filtered out. The filtration media used plays a key role in the removal of ammonia-N and phosphorus. Ammonia-N removal is limited by the lack of oxygen in the media and although phosphorus removal is usually low it depends on whether or not special media with high sorption capacity is used (Vymazal, 2011). The benefit of HF CWs is their low operation and maintenance costs. However, there may be problems with filtration bed clogging and subsequent surface flow (Vymazal, 2011).

Vertical flow constructed wetlands (VF CW):

VF CWs are similar to HF CWs as far as the purpose they serve, however VF CWs consist of a flat bed of gravel topped with sand and planted with macrophytes. They are designed so that wastewater flows down through the bed where it is connected to a drainage network and outlet at the bottom. Once the bed drains completely, air is allowed to pass through which provides a good oxygen supply to the system and helps improve nitrification (Vymazal, 2011). One of the main benefits of VF CWs is that they require less land than HF CWs to treat water. Nevertheless, VF CWs do not create suitable conditions for denitrification to produce gaseous nitrogen forms which can escape into the atmosphere (Vymazal, 2011). Another downside of VF CWs is that they can be more complicated to design and more expensive to build.

2.1.3 Hybrid constructed wetlands

Hybrid constructed wetlands combine the previously mentioned constructed wetland types to achieve greater efficiency in wastewater treatment (Vymazal, 2011). Common hybrids involve the combination of HF CWs and VF CWs. For example, several vertical flow beds can be installed parallel to one another in order to remove organics, suspended solid, and enhance denitrification while two to three horizontal flow beds are installed in series to remove solids and nitrogen (Vymazal, 2011). Hybrids constructed wetlands can be extremely beneficial, but they can be more expensive to build and more complicated to operate.

2.2 Proposed South Ellerbe Creek Wetland -

According to background information in Wise (2012), Durham city engineers recommend converting the abandoned Duke Diet & Fitness Center (located near the intersection of Duke Street and Trinity Avenue [See Figure 3]) into a large constructed wetland to help with stormwater cleansing. The abandoned site is roughly nine acres and collects runoff from 485 acres of highly developed land near downtown Durham. Local residents hope the city will buy the abandoned building

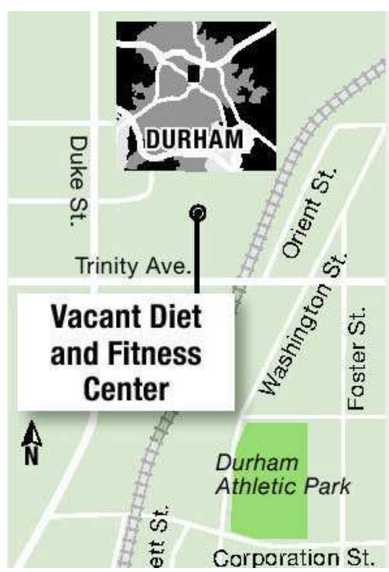

Figure 3 - Map of the area where the proposed wetland is to be built (Wise 2012)

and turn it into a community center, but engineering studies show that converting the building would cost more than the building is worth.

Council members have reservations about pursuing this project due to concerns with the budget and objections from citizens. Despite estimated costs for building the wetland being close to $8 million, engineering studies predict that it could save city taxpayers between $7 million to $20 million in expenses needed to comply with the new Falls Lake water quality regulations (Wise, 2012). Without the wetland, the city may have to buy land and build approximately 25 smaller filter sites to accomplish the same results as the proposed South Ellerbe Creek wetland. In addition to benefiting Falls Lake, it would also improve the water quality of Ellerbe Creek. Residents nearby the prospective wetland site have reservation about the project because previous city pollution mitigation projects have caused aesthetic problems, increases in crime, and increases in nuisance wildlife (Wise, 2012).

Preliminary plans show the wetland as a park-like feature that increases habitat for wildlife, filters stormwater, and provides opportunities for environmental education [See Figure 4] (Wise, 2012). A few of the key features listed on the concept plan include seating, a trial, a wetland outlet, a Shallow Marsh, and a Deep Marsh (Durham Government, 2013). Plants listed for use in the Shallow Marsh include Swamp Rose Mallow, Virginia Sweetspire, Cardinal Flower, Inkberry, and Swamp Milkweed (Durham Government, 2013). Plants listed for use in the Deep Marsh include Button Bush, Pickerel Weed, Woolgrass, Lizards Tail, Blue Flag, Iris, and Common Rush (Durham Government, 2013).

Figure 4 - Concept plan for the proposed South Ellerbe Creek wetland (Durham Government 2013)

3. Enhancing the Proposed South Ellerbe Creek Wetland:

The previously discussed constructed wetland types offer many options for creating functional wetlands. This information suggests that an ideal constructed wetland should incorporate multiple wetland types in order to enhance water quality. If the design of the Ellerbe Creek wetland were altered and modeled after the wetland system of the Alexandra Canal located in Singapore [See Figure 5], the overall management of stormwater runoff in this area would be optimized.

3.1 The Alexandra Canal Wetlands -

The Alexandra Canal is a 1.2 km concrete canal that was transformed into a beautiful functional waterway and community park. The project cost $34 million and was completed in 23 months with the primary motivation behind the project being to transform attitudes and behaviors of people towards stormwater and waterways in general (Kondepudi, 2012). Key features of the canal are four educational wetlands, a rain garden, and a bioretention swale. The wetlands used consist of a sedimentation bay (with a retention time of 3 days), a surface flow wetland (with a retention time of 2.9 days), a floating aquatic wetland (with a retention time of 7 days), and a subsurface wetland (with a retention time of 2 days) [See Figure 6] (Kondepudi, 2012). One major accomplishment of the project is that it showed how sustainable stormwater designs can be incorporated with minimal negative impact to the hydrologic cycle and aquatic ecology (Kondepudi, 2012).

Figure 5 - Overview of the Alexandra Canal in Singapore (Greatnewplaces.com 2011)

While the proposed wetland in Durham does not need to be on as large of a scale as the Alexandra Canal, there are many aspects of that project which can be used to improve the original concept of the proposed South Ellerbe Creek wetland. For example, the wetlands at Alexandra Canal are arranged so that water enters the system at the sedimentation bay and then flows to the surface flow wetland, floating aquatic wetland, and subsurface wetland (Kondepudi, 2012). It then exits the wetland system after flowing through the subsurface wetland. Having a system in place where stormwater passes through multiple wetlands can increase the efficiency of pollution removal from wastewater (Kondepudi, 2012). That is because there is a better chance of removing more pollutants if multiple purification steps are used. Therefore, Durham should consider incorporating a similar system with multiple wetlands connected to each other. Another aspect of the Alexandra Canal that can be used to make the proposed wetland better is the use of learning trails and educational huts with posters developed by local students to help those that visit the canal learn about the various stormwater management techniques used (PUB's ABC Waters Program, 2011).

Figure 6 - Wetlands featured at Alexandra Canal in Singapore (Kondepudi 2012)

4. Stormwater Education

If a wetland system like the one at the Alexandra Canal is implemented, it will only be a success if people start to change their mindset and attitudes towards stormwater runoff. The best way to make this happen is by educating individuals, starting with local citizens, about runoff to develop community support. The idea behind this is that if local decision makers, elected officials, and citizens have more than a basic grasp of stormwater pollution concerns, they will be more likely to make effective decisions that will have a positive impact on stormwater issues.

4.1 Singapore's Active Beautiful Clean (ABC) Waters Program -

A major component of the Alexandra Canal project is public outreach and education. Managed by the ABC Waters Program, the goal of education and public outreach is to change attitudes towards water and the management of wastewater/stormwater. One way the program has sought to do this is by creating interactive out-of-classroom learning experiences with the use of educational huts and posters scattered throughout the site along a learning trail (PUB's ABC Waters Program, 2011). This allows visitors to interact with the space and learn about how the garden-like park serves a dual purpose as a gathering space for the community and a water purification site. The ABC Waters Program even held a wetland planting day as the Alexandra Canal was being transformed to give local students a greater appreciation for stormwater management, water quality, and the relationship between nature and water (PUB's ABC Waters Program, 2011).

4.2 Elements of Successful Stormwater Outreach and Education Programs -

The U.S. Environmental Protection Agency also realizes the importance of public education in improving stormwater management and they have established what is known as the Phase II Stormwater Rules. The Phase II Stormwater Rules require stormwater management programs to include public education and outreach in addition to public participation/involvement (EPA, 2005). However, what does it take to make outreach and education programs successful? According to an article by Neiswender et. al. (2010), the six key elements that must be considered when developing education programs are (1) going beyond awareness with out-come based education, (2) audience targeting, (3) public participation, (4) incorporating stormwater into other natural resource and land use planning, (5) coordinating multi-jurisdictional efforts, and (6) having educators with technical expertise (Neiswender 2010).

To go beyond awareness with out-come based education means focusing on desired outcomes (i.e. changing individual behavior). A few bad habits that need to be changed are littering, improper disposal of trash and recyclables, application of lawn chemicals, changing motor oil on impervious surfaces, etc (EPA 2012). If individuals were to cut down on these negative behaviors, it would decrease the amount of pollution that ends up in runoff. Social marketing concepts that make out-come based education successful are placing specific behavior prompts near where the behavior is most likely to occur, removing barriers to desired behavior, and asking for a commitment from the intended audience (Dane County 2003). Audience targeting can help figure out who the intended audience is. According to a stormwater management plan created for Dane County, WI, there are three main types of audiences to target: those who must act (i.e. elected officials, homeowners, developers, business owners), those who must support change (i.e. conservation groups, civic organizations, the media), and those who must act and support in the future (i.e. youth) (Dane County 2003). Public participation is an essential part of education programs because it helps build support needed to implement changes (Neiswender, 2010).

Incorporating stormwater into other natural resource and land use planning should be considered when developing education plans because many of the steps taken to protect natural resources and manage sprawl also help protect stormwater infiltration areas, foster onsite treatment and infiltration, and reduce runoff caused by traditional curb and gutter designs (Neiswender, 2010). Coordinating multi-jurisdictional efforts can be beneficial because the efficiency of educational programs can be increased if resources from various agencies are combined (Neiswender, 2010). Also, it is important to have educators with technical expertise because if educators work with technical experts, they can devise ways to communicate technical messages to a variety of audiences in understandable ways (Neiswender, 2010).

5. Conclusion

Despite all of the regulatory requirements designed to reduce stormwater runoff and the amount of contamination therein, stormwater still reaches and pollutes a high percentage of waterways. Many of these contaminants result from everyday activities in and around urban communities and the best way to reduce them is by educating the public. It takes individual behavior change and proper practices to control such pollution, so the public must be sufficiently aware and concerned about the significance of their behavior for stormwater pollution. Through information and education, improper behavior can be changed.

As for the proposed Ellerbe Creek Wetland, the current design plan can be improved by incorporating a multi-step purification system with the various constructed wetland types. A previous study by Kondepudi (2012) proved that the multi-step wetland system in Singapore at the Alexandra Canal can efficiently remove unwanted waste in the waterway due to pollutants, so it can serve as a model for other wetland systems. The concept plan for the proposed wetland mentioned including a wetland outlet, shallow marsh, and deep marsh. While these wetlands can help remove pollutants and moderate stream flow by reducing peak and flood flows, it would be better if officials creating the plan included a variety of wetland types. If the wetlands at Alexandra Canal were used an a model system to design the proposed Ellerbe Creek Wetland after, the best approach for creating an efficient wetland system would be to include at least one HF CW, one VF CW, and one hybrid constructed wetland in a multi-step sequence; so, stormwater can be purified at multiple stages. The more purification stages the water passes through, the less contaminated it should be.

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Stormwater at Raleigh-Durham International Airport

Heather McGee

Introduction

Urbanized areas can have diverse impacts on both hydrology and surface water quality. This is usually attributed to large amounts of impervious surface. Impervious surfaces, such as roads and parking lots, do not allow water to infiltrate the soil and enter the groundwater. This can cause a decrease in the height of the water table in urban areas. Another effect of impervious surfaces is that rain runs across them and often picks up an array of chemicals and compounds that degrade surface water quality.

Raleigh Durham International Airport (RDU) is a 5,000 square acre facility through which 9.2 million passengers traveled in 2012 (RDU Airport Authority, 2012). Airports are prime examples of urbanization on the landscape, requiring both large amounts of impervious surface for runways, terminals, and extensive parking lots. Airport runoff also has high concentrations of many pollutants including polycyclic aromatic hydrocarbons (PAHs), pesticides, polychlorinated biphenyls (PCBs), heavy metals, grease and oil (Sulej et al., 2011).

However, current technology has provided many ways to capture and treat stormwater runoff via 'green infrastructure' (Ahiblame et al., 2012). For example, rain cisterns collect rainwater from roofs and prevent the runoff from flowing rapidly into surface water bodies. Detention ponds even allow for recharge of groundwater by increasing filtration. This paper examines what the RDU Airport Authority has implemented to reduce the effects of stormwater, and explains the known environmental effects of airports. Then the challenge of determining the impact of RDU on water quality data is discussed and finally recommendations are provided on how RDU could improve its stormwater practices.

Current Practices at RDU

The main stormwater management practice currently implemented at RDU is the use of Brier Creek Reservoir as a detention pond for stormwater runoff (RDU Airport Authority, 2012). This man-made lake holds runoff from the facility and the water is used to irrigate 80% of the grounds (RDU Airport Authority, 2012). The term "reservoir" is a bit misleading as this water is not potable, nor is it used for drinking water (RDU Airport Authority, 2012). This water will eventually re-enter the surface or groundwater system.

Figure 1 - Construction of detention pond next to reservoir (Parsons 2005)

Figure 2 - Completed detention pond next to reservoir (from Google Maps)

Detention ponds help to reduce the impact of stormwater by decreasing the large surge of water that runs off impervious surfaces during rain storms. They also allow pollutants to settle out because the water is slowed. At RDU, a 5-acre storm water treatment pond is fed by a 2,700-foot-long, 84-inch-wide storm drain that is buried 45 feet under runway 5L-23R (Parsons, 2005)(Appendix i). This then drains through a 60 inch pipe, complete with an overflow channel (Parsons, 2005).

Figure 3. **Another example of an RDU Stormwater Detention Pond (from Google Maps)**

RDU also samples its stormwater for pollutants to ensure that it complies with the Environmental Protection Agency's (EPA) standards (RDU Airport Authority, 2012). In addition, RDU tests its drinking water, to ensure it is safe to drink. Weirs have been placed in ponds on the property to control any fuel spills that may occur (RDU Airport Authority, 2012). The property is also inspected for pollutants that are exposed to rainfall, so they will not be washed away during a storm event(RDU Airport Authority, 2012).

Environmental Concerns Regarding Airports

Atmospheric Pollution

The most commonly researched aspect of air travel's effect on the environment is airborne pollutants and noise pollution (Nunes et al., 2011). The burning of jet fuel, fumes that escape during fueling, and atmospheric deposition of particulate matter are common concerns (Nunes et al., 2011). Air travel research has often focused on air pollution because it originates both from the centralized region of the airport, and from airplanes themselves as they fly from city to city across the earth. Nunes et al. (2011) recognized this focus on atmospheric and noise pollution, and argues that more research should focus on soil and groundwater contamination from airports. Most atmospheric pollutants eventually settle out as wet or dry deposition. With either possibility, the pollutants can then be transported by stormwater into the soil, groundwater, or surface water (Nunes et al., 2011). Therefore, atmospheric pollutants, especially those originating from the airport itself, have impacts on surface and ground water quality.

Hydrology

Another potential impact of RDU airport on its surrounding environment is the changes of hydrology due to impervious surfaces. Instead of a natural surface, covered with vegetation, the airport comprises approximately 700 acres of paved or roofed surface (ESRI, 2012). While much of this water is diverted to the Brier Creek Reservoir (Parsons, 2005), and therefore is able to infiltrate to groundwater, the spatial distribution of infiltration is changed. Depending upon how the airport stormwater sewer is arranged, all the water that falls as rain with the Sycamore watershed boundary may be piped to the Little Briar watershed (Figure 4). This changes the amount of water available to human and animal populations downstream in the Sycamore watershed.

Deicing

In winter months, especially in cold climates, it is a major undertaking to keep airplanes and runways clear of ice. In order to keep safely continue air traffic during cold weather, deicing fluids are sprayed on both aircraft and runways (Switzenbaum et al., 2001). Common deicing compounds for aircraft are propylene glycol and ethylene glycol based fluids, while ethylene glycol and urea are used to deice runways (Koryak et al., 1998). These compounds have been shown to increase microbial activity, therefore decreasing oxygen supply, when they enter local water bodies (Koryak et al., 1998). In the winter of 1995-1996, 3 million liters of ethylene glycol and a half a million liters of urea were spread on the runways of Pittsburgh International Airport (Koryak et al., 1998). While data is available for RDU, it is likely that much lower quantities of deicing fluids are used due to its warmer climate.

Thermal Impacts

Thermal changes to ground and surface waters are likely more pronounced at RDU than at airports in cooler climates. It has been shown that rain which falls on paved surfaces often 'picks up' heat from the pavement before flowing into ground or surface water (Herb et al., 2008). These changes in temperature can affect the overall temperature of water body, and limit the variety of biota that can survive there (Kaushal et al., 2010). The large amount of paved surface at RDU has the potential to significantly increase the temperature of rain from when it hits the ground, to when it joins the surface water.

Challenges to Determining Potential Effects of RDU runoff on local waterways

Superfund Site

Unfortunately, due to the proximity of an unrelated Superfund hazardous waste site, it is difficult to tease out any negative effects of RDU's stormwater runoff on local water bodies. Just upstream of the Brier Creek Reservoir is the former site of the Ward Transformer. This was a facility which manufactured, repaired and rebuilt electrical equipment (EPA, 2012). The facility has high levels of PCBs, which have also been found in the water and soil downstream of the facility (EPA, 2012). Because of these contaminants, it is unsafe to consume fish from Brier Creek Reservoir, or its tributaries (NC Wildlife Commission, 2012). While the concentration of PCBs decreases further downstream, the EPA still states that it is unsafe to eat carp or catfish from Lake Crabtree, which is located approximately 4 miles from the original site (EPA, 2012). Because one of the main pollutants from airports is also PCBs, it is very hard to separate the PCBs inputs from the Ward Transformer Superfund site, and the inputs from RDU airport.

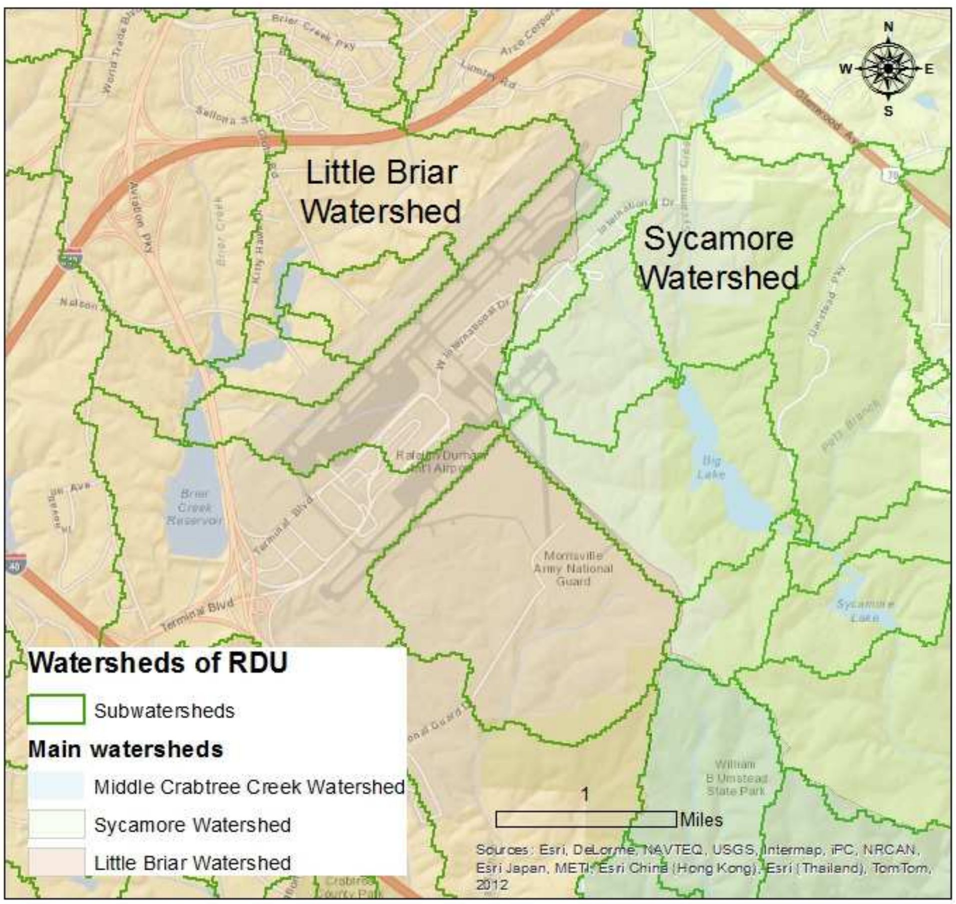

Figure 4 - Watersheds which include RDU airport. Note the straight-line divides between subwatersheds, caused by artificial drainage.

Watershed Divides

It is worth noting that RDU airport sits on the watershed boundary between Little Briar and Sycamore watersheds (Figure 4). Because the area surrounding runway 5L-23R (if not more) is artificially drained to Brier Creek Reservoir, it is unclear how much of the RDU property drains into the other side (Appendix i). Usually, an analysis of topography using high resolution digital elevation models (DEMs) would show which areas drain to which watershed. However, the use of stormwater conveyance infrastructure undermines this analysis.

NDPES Permits

Possible pollutants from RDU airport which may enter surrounding surface waters are diverse, from PCBs, deicing chemicals, fuel, and thermal pollution (Sulej et al., 2011). Some evidence that RDU does emit some sort of water pollutant is that they have been issued two individual wastewater National Pollutant Discharge Elimination System (NDPES) permits (NC Division of Water Quality, 2013a). They have also been issued a general Groundwater Remediation NDPES Permit, and a Stormwater Runoff NDPES permit (NC Division of Water Quality, 2013b). These permits are issued to quantify, control and reduce point-source pollutants across the country (NC Division of Water Quality, 2013a).

Lack of Water Quality Data

While no study could be found that specifically tested the effects of RDU on water quality, some information can be obtained from past North Carolina Department of Environment and Natural Resources monitoring sites. Sites were located both upstream and downstream of the airport in both Sycamore and Little Briar watersheds. However, the parameters measured at each station varied, and only a few parameters were measured both up and down stream of RDU (Table 1). Any spatial changes in water quality cannot be inferred from these data, because they were often taken years apart. The means of several measurements at each site, number of measures taken, and the years they were measured are included purely to provide an idea of the data available (Table 1). Additional data on the water quality around RDU is needed in order to determine the effectiveness of their stormwater management measures, and to isolate any impact of the airport on downstream waters.

Parameter	Watershed	Upstream	Downstream	Unit	Years of Upstream Sampling	Number of Upstream Samples	Years of Downstream Sampling	Number of Downstream Samples
Ammonia-nitrogen as N	Sycamore	0	0.061	mg/L	1973	2	1981, 1991,2000, 2010	10
Ammonia-nitrogen as N	Little Briar	0.05	0.1375	mg/L	1973	1	1990, 1991, 1995, 1996	4
Kjeldahl nitrogen	Little Briar	0.4	0.825	mg/L	1973	1	1990, 1991, 1995, 1996	4
Kjeldahl nitrogen	Sycamore	0.1	0.44	mg/L	1973	2	1981, 1991, 1995, 2000, 2010	5
Dissolved Oxygen	Little Briar	6.8	6.827	mg/L	1973	1	1990, 1991, 1995, 1996	4
Dissolved Oxygen	Sycamore	7.88	67.15	mg/L	1973 - 1975	5	1981, 1991, 1995, 1996, 2000,	27
Dissolved Oxygen Saturation	Little Briar	81	85.509	%	1973	1	1990, 1991, 1995, 1996	4
Dissolved Oxygen Saturation	Sycamore	84.8	67.15	%	1973-1975	5	1981, 1991, 1995, 1996	12
Fecal Coliform	Sycamore	98	15.714	cfu/100ml	1973-1975	5	1996, 2000, 2010	7
Phosphate-phorphorus as P	Little Briar	0.06	0.1375	mg/L	1973	1	1990, 1991, 1995, 1996	4
Phosphate-phorphorus as P	Sycamore	0.05	0.06	mg/L	1973	2	1981, 1991, 1995	6
Temperature	Sycamore	19.8	27.474	Degrees C	1973-1975	10	1981, 1991, 1995, 1996, 2000,	39
Temperature	Little Briar	25	23.164	Degrees C	1973	1	1990, 1991, 1995, 1996	5

Table 1 - Available water quality monitoring around RDU

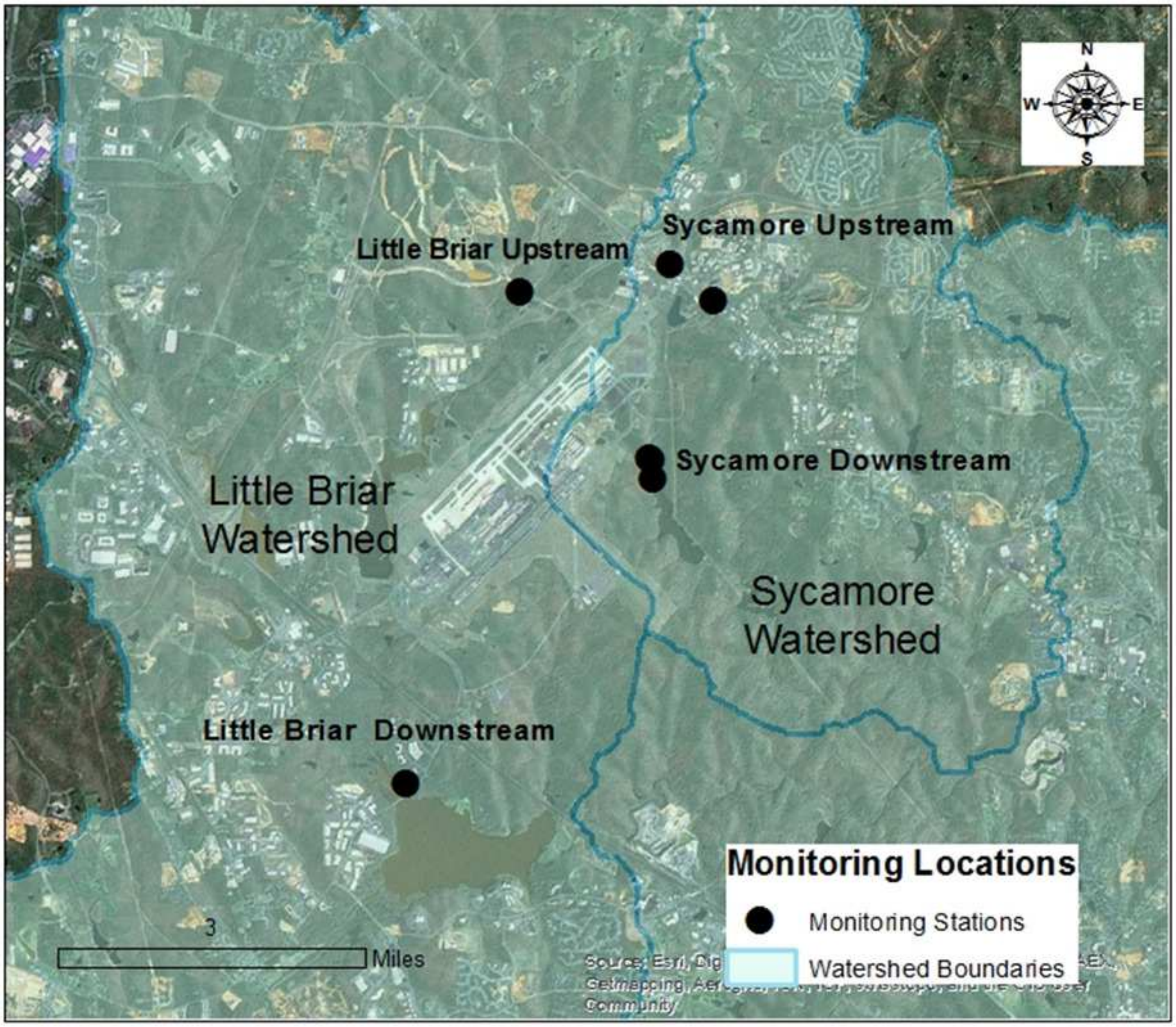

Figure 5 - Map of monitoring locations around RDU Airport

Recommendations for Improvement

Although there are many potentially harmful impacts of airports, there are many techniques that can be used to mediate environmental harm. While RDU already has multiple programs in place to recycle, control spills, and protect wildlife, several ways in which they could improve their stormwater management are proposed below.

Constructed surface wetlands

One possible method of preventing pollutants from entering waterways is by intercepting them with constructed surface wetlands. Wetlands have been shown to be capable of removing various pollutants from the waters that flow through them (Kadlec and Wallace, 2008). These constructed treatment wetlands vary from vegetated bio-retention swales between runways to large scale swamps requiring altered hydrology and new plantings. These structures can filter out excess nutrients and heavy metals (Calijuri et al., 2011). A natural wetland situated downstream of a pollution source will likely filter out excess nutrients and heavy metals. If natural wetlands are not present, they can be constructed at low cost, and designed specifically to increase their filtering capacities (Calijuri et al., 2011).

However, a problem with these structures is that they often attract waterfowl. While in most restoration projects this is a positive side effect, for airports this poses a major problem. Airports actively try to keep birds away from runways, because of the damage that can occur to both planes and birds when they meet unexpectedly. At RDU, nets are placed over all the lakes and ponds on the facilities to prevent birds from using them, and frequenting airport property (RDU Airport Authority, 2012). Therefore, if constructed surface wetlands are used, they should be placed well away from runways, and possibly be covered as well. Since RDU owns nearly 5,000 acres and only about 1,500 are currently used for terminals and runways, adequate distance from runoffs could be achieved. As shown in Figure 5, there are several natural wetlands located near RDU. It is usually more effective to restore impaired wetlands than to build new ones. Therefore, these already wet sites are good candidates for new and restored treatment wetlands.

Constructed subsurface wetlands

A more suitable form of remediation for RDU airport may be subsurface constructed wetlands. These systems are similar to the

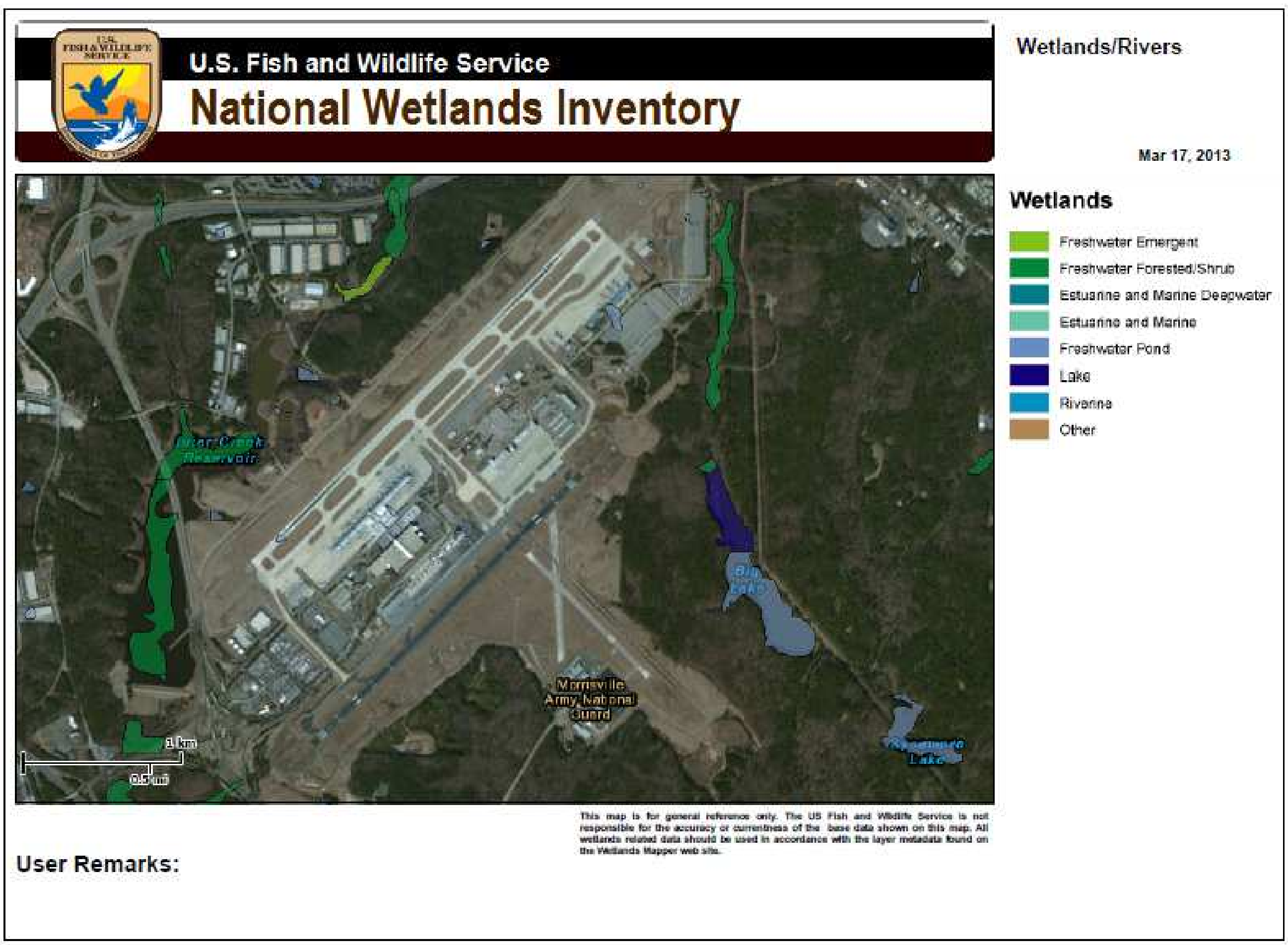

Figure 6 - Map of current wetland locations near RDU airport

constructed surface wetlands mentioned above, only the water is moved through the system entirely underground. This avoids attracting waterfowl, while still providing the filtering benefits of wetlands. Subsurface wetlands are particularly useful at airports, where plenty of space is available (Higgins and McLean, 2002). Higgins and McLean (2002) demonstrated that a constructed subsurface wetland was effective at removing de-icing glycols from stormwater.

Pervious surface addition

For some uses, such as parking lots and driveways, permeable surfaces can replace typical impervious paving materials like cement and asphalt (Ahiablame et al., 2012). These surfaces have been shown to reduce runoff by 50% to 93% (Ahiablame et al., 2012). Unfortunately, due to the high-impact nature of airport runways, they need to meet strict guidelines and regulations (Guistozzi et al., 2012). Unfortunately, permeable surfaces are simply not durable enough to use in this case. While RDU may not be able to reduce the runoff from their runways, they may be able to make them 'greener' in a different way. Guistozza et al. (2012) have shown that the use of recyclable materials in airport runways is an effective way to meet the required standards while also increasing sustainability. At an airport in Italy, the use of 85% recycled materials in runway pavement resulted in 35% reduction in emissions from paving (Guistozza et al., 2012).

One thing that RDU has already done well is building parking garages rather than large parking lots. This reduces the impervious surface area, and allows for more natural infiltration rates. However, RDU's long term parking areas are lots, not garages, and they are made from impervious surfaces (RDU Airport Authority, 2012). While it would be expensive to tear up and re-pave these lots with permeable methods, it would be more cost effective to use permeable pavements in any new parking lots that may be built in the future.

Bio-retention/Swales/Rain Gardens

Another method which could be used to reduce runoff is to create bio-retention cells in the small areas of grass between runways and around the terminal. These could allow stormwater to infiltrate into the groundwater instead of being piped into Brier Creek Reservoir. Bio-retention cells and rain gardens have been shown to reduce peak runoff by 45% (Ahiablame et al., 2012). This would be another low cost option.

Green Roofs

One of the most expensive methods of reducing runoff is by installing green roofs (Szalay, 2011). A green roof is a roof that is covered with a lining and rocks, and soil, on which plants are grown (Ahiablame et al., 2012). Rainwater is collected on the roof, instead of running off. Small, resilient plants are raised on the roof, to absorb the water, and reduce net carbon dioxide emissions (Ahiablame et al., 2012). While green roofs require major renovations, they could also be used for public outreach if they are open to visitors. RDU already has an observation deck, for citizens to watch airport operations. If a section of a terminal, or any building was converted to a green roof, it would make a very pleasant observation deck, which doubles as positive marketing for the airport. Many peoples opinion of the airport would increase if they knew its managers were acting sustainably.

Green roofs have successfully been installed in several airports in Europe, and at O'Hare airport in Chicago (Fernandez-Canero and Gonzalez-Redondo, 2010). A concern with green roofs at airports is that they may attract birds. As stated above, this increases the chance to aircraft accidents. Other airports have addressed this problem by carefully choosing the vegetation on the roofs to ensure no food plants are included (Fernandez-Canero and Gonzalez-Redondo, 2010). The Schiphol International Airport in Amsterdam even uses a team of trained dogs to chase birds off the green roof (Fernandez-Canero and Gonzalez-Redondo, 2010).

Offsets

If none of the above strategies are deemed feasible by the RDU Airport Authority, the harm caused by the airport facilities could be offset by preserving land elsewhere. Instead of installing stormwater control measures at the airport itself, RDU could contribute to efforts to install them in other places within the watershed. In this way, a positive impact could be made while minimizing costs.

There is also a certification process airports can participation in to show their commitment to sustainability. It is called the ISO14001 Certification, and only 3 airports in the United States have earned it (Shane, 2007). It is a voluntary process which requires organizations wishing to obtain certification to uphold certain standards of sustainability (ISO14001Certification.com). Obtaining the ISO 14001 Certification would be another way in which RDU could both reduce runoff pollution, and increase their visibility and public image.

Conclusion

Like urban areas in general, RDU airport has the potential to impact the hydrology and water quality of surrounding areas. The impervious surfaces associated with the airport decrease rainwater infiltration, increase the flashiness of streams, and add pollutants to watersheds. However, RDU has made attempts to remedy this impact by creating several detention ponds which increase infiltration, allow sediment to settle out, and reduce stream flashiness. This may be the stormwater control method of choice for RDU because it is one of the cheapest stormwater control methods.

Further research is needed to isolate the impacts of the RDU's stormwater from the impacts of nearby industry. This data would also allow the RDU Airport Authority to determine how much mitigation is needed to counteract the negative environment effects it may cause. If RDU would like to further reduce their impact on local watersheds, they could install permeable parking areas, green roofs and/or create wetlands or small swales to filter runoff. The increased use of stormwater control measures at RDU International Airport would help reduce the ecological impact of the airport facilities. ■

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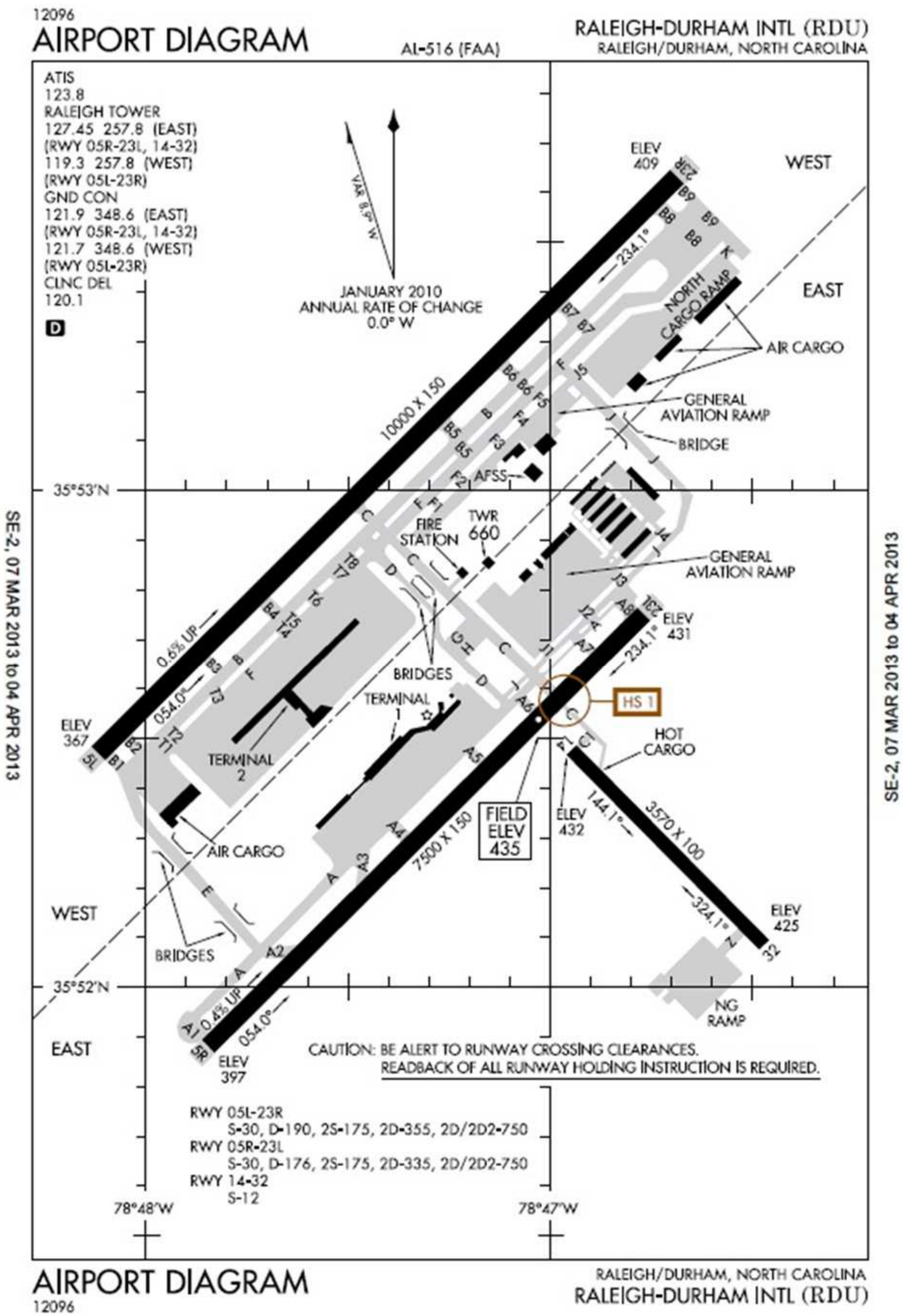
Appendix i From RDU Airport Authority's Noise Reduction Program



Stormwater Mitigation Banking

Andrew Kondash

Introduction:

Stormwater mitigation banking is a means for accounting for and minimizing the growing effects of urbanization. Before understanding the process of stormwater mitigation banking, it is important to develop an understanding of the process. Mitigation banking is the process of making, fixing, or increasing protection of ecosystems to offset anticipated impacts. (Osmond, et al., 1995) In other words, it is a process for incentivizing the adoption of ecosystem services. Adding a banker as the middle man cuts out the time for contractors and builders to get permits, estimates, and draw up mitigation plans, while it allows the banker to sell the credits it produces for a profit. (Osmond, et al., 1995) Stormwater mitigation banking comes from increased pressure on lawmakers and developers to limit discharges from impervious surface into river systems.

The problem with stormwater flow over impervious surface is that the quick flows, coupled with nitrogen, heavy metals, and other pollutants that have a tendency to accumulate on those surfaces are easily concentrated and focused on individual watersheds. The most notable instance of this occurs when pervious surfaces, such as fields are paved to build a road. The Environmental Protection Agency (EPA) has proposed the idea of adding a new bill to define stormwater mitigation practices to complement its recent setting of discharge limits on highway and other construction projects. (Heltman, John, 2010) While this bill would be implemented for construction on interstate and other federal projects, mitigation banking and best management practices (BMPs) are already taking place at the state and local level. This paper will focus on mitigation banking and BMP processes of some states and compare those to current North Carolina practices before summarizing potential directions in which the state and national plans.

State Practices:

Delaware:

The issue of stormwater management first came to light in Delaware in 1991 when the legislation, 7 Del. C., Ch. 40, was passed subjecting the state to regulations requiring the treatment of stormwater pollution and the flooding attributed to runoff from development. (Davis V, 2008) After several failed attempts at a regulatory structure, the Delaware Department of Transportation (DelDOT) developed the idea to self-regulate and began building stormwater retention ponds. While this process worked to prompt the building of several ponds, developers began having problems with this system. The main problem was that many of the roads under construction crossed multiple watersheds, requires small individual projects in each watershed. To solve this problem, DelDOT and the Delaware Natural Resources and Environmental Control (DNREC) collaborated to form a memorandum of agreement (MOA), which helped create a framework structure for the development of stormwater banking practices. (Davis V, 2008) The banking system worked to allow for deficits in one watershed to be made up in another in greater need of water quality control. This helped to alleviate stresses from minor roadway widening projects and urban development where nearby projects may not be available. Thus far, the plan has been a sweeping success with 39.15 credit acres and only 7 of the 45 watersheds in the state with a deficit (no deficit is less than 0.55 acres). (Davis V, 2008) The system itself is set up on a per-project basis and audits are conducted every 3 years to gauge effectiveness. On top of saving time, DelDOT also argues that safety and environmental protection are greatly improved because small projects such as adding a 1-2 foot shoulder to the road do not pose the bureaucratic hurdle they once did, while smaller projects can be lumped together into larger, more meaningful retrofits and rehabilitation. Funds collected from the banking process have also gone towards standalone projects such as stream restoration and parking lot retrofits.

New Jersey:

In New Jersey, on site treatment of storm water runoff is required in all New Jersey Department of Transportation (NJDOT) projects. Similar to Delaware, when the right of way is not readily established or a road project cuts across watershed boundaries, mitigation is required in the form of an excessive number of small BMP projects. (Aboobaker N, 2009) These small projects are difficult to manage, expensive, and time consuming for the state. As of 2009, the idea of developing a credit and debit system was in its infancy and several reports on the feasibility and development of a banking system have only now been filed. Current NJ laws require 80% total suspended solids removal from new pavement and 50% removal from reconstructed pavement. (Fekete A, 2011) Current protocol requires the state to mitigate on site and when not possible, up stream of the proposed site. As of 2011, no further work has been done to implement a system.

Washington (State):

The state of Washington, like New Jersey, is currently in the process of examining the feasibility of a mitigation banking system. The Washington State Department of Transportation (WSDOT) and Washington State Department of Ecology (WSDOE) have identified advanced mitigation (banking) or environmental credit trading as the main vessels through which large scale mitigation can be carried out. (Girts M, et al., 2001) Currently the implementation of any sort of program is still under review, with lessons being taken from other states that have successfully implemented mitigation banking programs, along with lessons from current wetlands mitigation banking in state. (Landry M, et al., 2005) No consensus has yet emerged as the favorite for program adoption.

North Carolina Mitigation:

General Information:

In North Carolina, mitigation is required by the North Carolina Department of Natural Resources' (NCDENR) Ecosystem Enhancement Program (EEP). When a development project is first proposed, the developer is required to draft a stormwater plan for the project and calculate nutrient loading from the proposed site. If the proposal has explored all possible alternatives and no viable way to avoid impacts is determined, mitigation is required. The program requires two different types of mitigation to occur. Under 15A NCAC 02H .0506(h) and 15A NCAC 02H .1305(g) the NC Division of Water Quality (DWQ) requires compensatory mitigation for losses of streams and wetlands under section 404 of the clean water act. The other required offset is for nutrient loads in areas that have established Nutrient Management Strategies (NMS). There are four NMS in North Carolina, shown in the map below, and are for Falls Lake, Jordan Lake, the Neuse River basin, and the Tar-Pamlico basin. (Anon., n.d.)

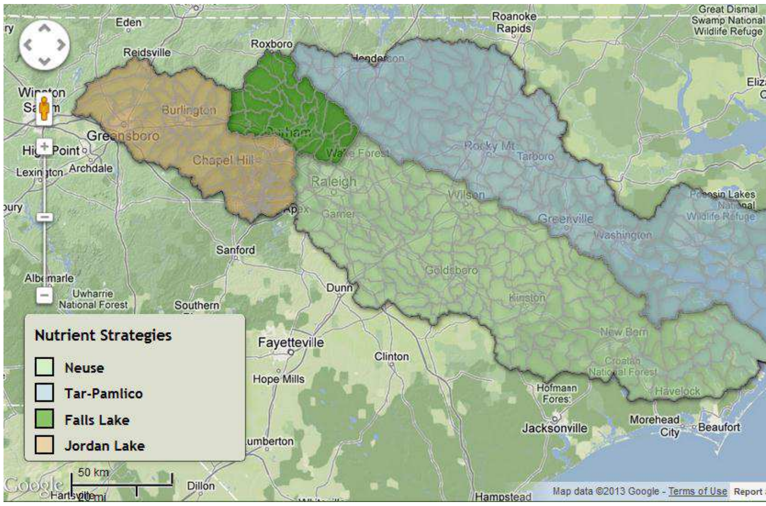


Figure 1: North Carolina Major River Basins

Upper Neuse River Basin (Falls Lake) and why urban development and mitigation is so import to these areas.

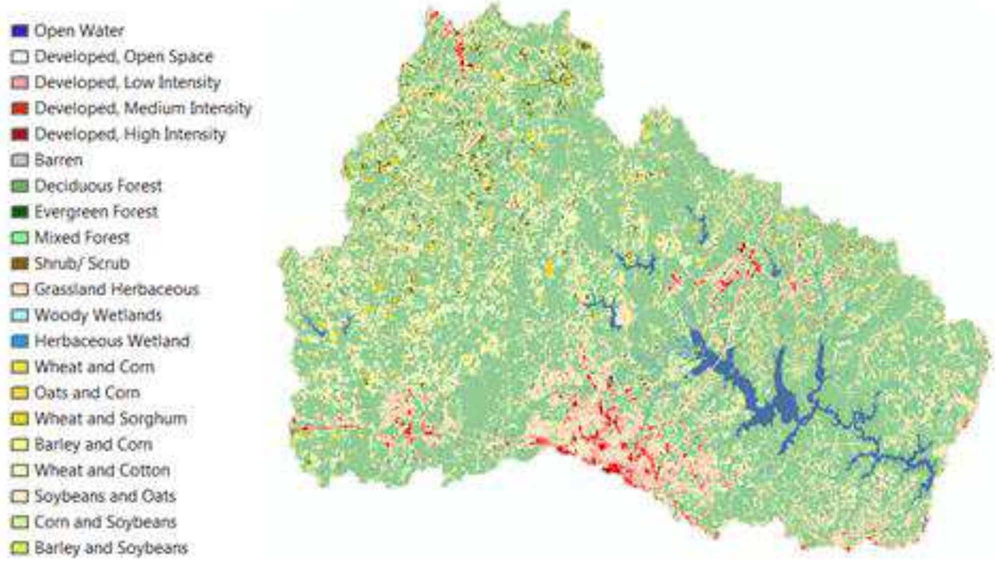
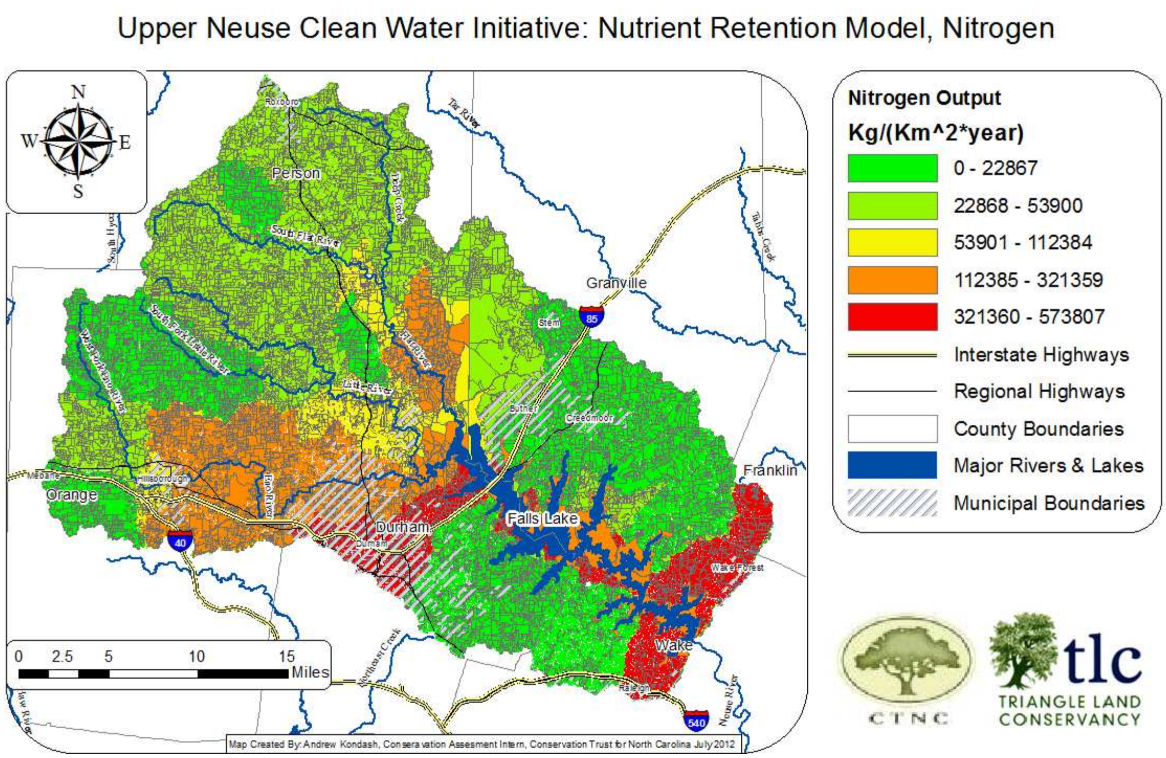


Figure 2:Upper Neuse River Basin Landuse/Landcover

Nutrient Rules: A Closer Look

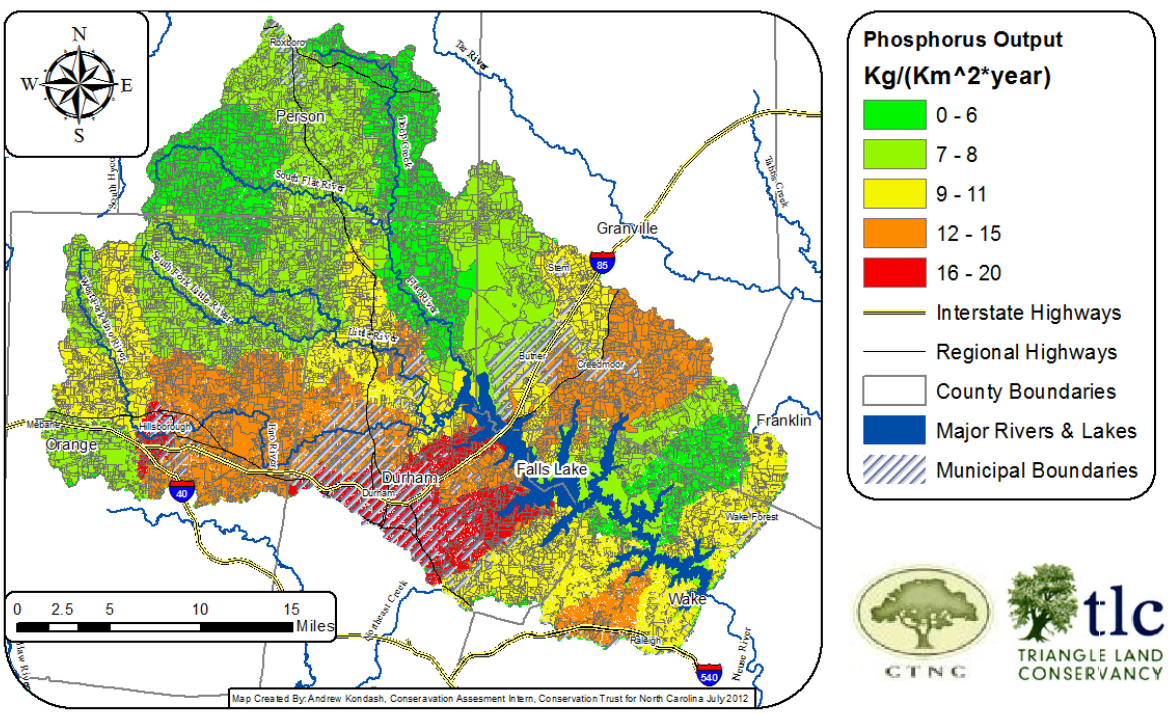
Since there are four different nutrient management strategies in North Carolina, the Falls Lake rules are used as the guideline for this section. The rules were used because they were the most readily available and have been proven to be effective. This rule system sets up a two part goal for nutrient and nitrogen mass load removal by 40% and phosphorus load removal by 77% from 2010 levels by 2021. (NC Rules Review Commission (RRC), 2011) New development projects are limited to 2.2 pounds per acre of nitrogen and 0.33 pounds per acre per year of

Any development project within these regions must have reports on how they will meet nitrogen and or phosphorus nutrient load target rates. Whenever development is unable to reach these goals onsite, either off-site mitigation or the purchase of nutrient offset credits from private mitigation banks or from NCEEP. The following maps show how impervious surface, nitrogen, and phosphorus loads are correlated in the



phosphorus. When these goals cannot be reached, offsite mitigation can be carried out or mitigation credits can be purchased in accordance with 15A NCAC 02B .0277 4(b). In accordance with 15A NCAC 02B .0277 4(a), existing sites are tasked with retrofitting to meet predevelopment rates and where unavailable to limit discharge to 2.89 pounds nitrogen per acre per year and 0.63 pounds phosphorus per acre per year. (NC Rules Review Commission (RRC), 2011) The rules also try to develop an accounting system for the measurement of discharges. These programs are set to begin in July of 2013.

Upper Neuse Clean Water Initiative: Nutrient Retention Model, Phosphorus



The Banking System:

The EEP works with private mitigation banks to purchase credits and is currently working on developing a database of regulatory agencies and their inventories so developers can be effectively matched with their best possible offset projects. 15A NCAC 02B .0279 8(a)(ii) of the Falls Lake Rules states that developers may fund offset measures when unable to meet discharge requirements by paying NCEEP after filing claims with the state or through another seller of offset credits are approved by the DWQ. 15A NCAC 02B .0278 6 and .0277 4(b), state that municipalities can work with counties and municipalities within the same subwatersheds to meet reduction goals. Additionally, it says that those municipalities have the option to work with private or third party sellers such as mitigation banks. (NC Rules Review Commission (RRC), 2011)

The EEP is primarily responsible for the banking system and in itself, purchases credits from established mitigation banks. There are two mechanisms through with EEP purchases credits. (Anon., n.d.) First, they can file a waiver of competition, which can be used to purchase (through the NCDENR Purchase and Services Division) a small quantity of credit in a service location monopolized by one bank. (Anon., n.d.) Alternatively and more frequently, a “Full-Delivery Procurement Process” can be undertaken through which EEP will compete with other mitigation banks to fund projects of larger quantities. (Anon., n.d.)

Conclusion:

Looking towards the future, there could be a push away from traditional retention ponds and towards newer, greener technologies such as biorentention areas, bioswales, infiltration trenches, and filter strips. (Davis

V, 2008) While programs such as DelDOT's mitigation banking initiative and the Falls Rules are currently working to improve and facilitate stormwater management, several hurdles and problems still need to be addressed. These include the movement of measuring metrics away from total suspended solid (TSS) load mitigation and towards individual nutrient measures on a watershed bases. (Davis V, 2008) The success of systems such as DelDOT and Falls Rules are facilitating the movement away from site-specific restoration and towards crediting and debiting within entire watersheds, taking the pressure off of developers. These successes are also spurring the adoption of similar programs in states such as New Jersey and Washington.

As technology becomes more advanced, the tools with which to combat increased pollution loads will also develop. It is clear that the state of North Carolina is ahead of the curve with the development and implementation of the Falls Lake Rules and other NMS projects which already used a pollutant based system and have proven the successful adoption of a banking system. ■

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The Effectiveness of Urban Tree Planting in Reducing Atmospheric Particulate Matter

Christopher Hewitt

In urban areas, the planting of appropriate tree species in sufficient numbers can improve the urban environment. The planting of trees and other vegetation can reduce stormwater runoff, lower the effects of urban heat islands by reducing temperatures and ultraviolet radiation through shading and evapotranspiration (US EPA, 2006), remove CO_2 and other gaseous pollutants from the atmosphere by sequestering them into tissues and wood production, and filter ultra-fine particles and aerosols from city air (Manning, 2008) that could become pollutants within stormwater systems and urban watersheds.

Particulate matter (PM) can be classified into two categories, primary or secondary. Primary particulates are carbon particles formed from such activities as the incomplete combustion of fossil fuels, mining, quarrying, and tire wear. Secondary PM forms from chemical reactions in the atmosphere (Becket et al., 2007), such as the formation of O_3 from photochemical reactions involving volatile organic compounds (VOCs) released by plants (Bealey et al., 2007).

In order to quantify pollution concentrations, PM is generally categorized into three sizes: PM_{10}, $PM_{2.5}$, PM_1, or $PM_{0.2}$. PM_X is the abbreviation for particulate matter with a diameter less than X µm and is measured in μgm^{-3} of air. PM_{10} are highest in urban centers characterized by heavy industry and road traffic (Bealey et al., 2007; Harrad and Hunter, 2006). Finer particulates (< $PM_{2.5}$), however, can travel great distances from their source and have extended residence times in the atmosphere (Beckett et al., 2000a). Regardless of size, particulates suspended in the air have adverse effects on human health and greatly impact those with preexisting respiratory and cardiac problems; finer particulates have been linked to asthma formation and alveolar inflammation (DoH, 1995).

In addition to increasing amenity values (Donovan and Butry, 2010; Sander and Haight, 2012; Sander et al., 2010), improving quality of life, decreasing ambient air temperatures, and trapping gaseous pollutants (Beckett et al., 2007), trees are an excellent form of vegetation for airborne particulate interception because of their large leaf surface areas and the turbulent mixing of air passing through their leaves (Beckett et al., 2000b). The planting of trees in urban areas has been shown to effectively remove airborne particulates, and can directly lead to a decrease in the number of deaths and hospital admissions every year in regards to adverse health effects from poor air quality (Tiwary et al., 2009).

The purpose of this paper is to provide a literary review on the effectiveness of trees in the removal of airborne particulates that both decrease urban air quality and can become contaminants in stormwater systems. The objectives of this paper are to : (1) provide an overview of how trees remove airborne particulates from the atmosphere, (2) examine the physiology of trees in order to determine whether a species is effective at particulate capturing, (3) survey ways and locations of tree planting that maximize particulate capture, (4) discuss several negative side effects associated with the use of trees for improving air quality, and (5) examine measures that can and are being taken in Durham, North Carolina in regards to urban planting.

Deposition

PM is removed from the atmosphere through either wet or dry deposition. Wet deposition is when rain, snow, mist, and washout collect particles in the atmosphere, and then deposit them on the ground or other surfaces. Dry deposition occurs when particles are directly intercepted through gravity, sedimentation, and turbulence. Figure 1 demonstrates that a variety of impaction and interception processes can lead to particulate removal. Beckett et al. (1998) demonstrated that particulates are most easily captured on leaf surfaces that are moist and rough and that magnetic and electrically charged surfaces are also effective at removing PM (Kardel et al., 2011). Leaves will continue to collect particulate matter until equilibrium is reached, and upwards of 50% urban tree PM_{10} interception can be resuspended (McPherson et al., 1994). ■

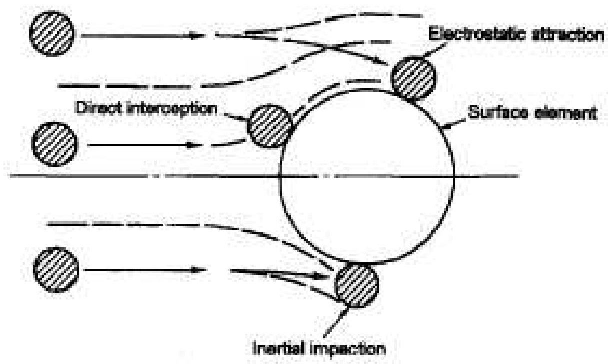

Figure 1 – Mechanisms of airborne particulate removal. From Boubel et al. (1994).

Effective Tree Types and Species

Tree species choice, planting design, and location relative to pollution source are critical in determining the effectiveness in PM capture by trees. Numerous studies have examined how tree type (conifer/deciduous broadleaf) and leaf physiology (presence of waxy cuticle, trichomes, rough/smooth surface) effect deposition rates (Beckett et al., 1998; Beckett et al., 2000a; Donovan et al., 2005; Freer-Smith et al., 2004; Hwang et al., 2011; Kardel et al, 2011; Nowak et al., 2006; Peachey et al., 2009; and Saebo et al., 2012). In addition to meteorological field sampling techniques, tree capture ability has been studied through the use of wind tunnels examining capture efficiencies and deposition velocities (Fig. 2).

Numerous studies have demonstrated that atmospheric deposition and retention is greater for coniferous species than deciduous, broadleaf species (Beckett et al., 2000a; Donovan et al., 2005; Freer-Smith et al., 2004; Hwang et al., 2011; Nowak et al., 2006; Saebo et al., 2012). Conifer needle physiology and life strategies provide benefits that other species of trees cannot. The retaining of foliage (Becket et al., 2000a) gives conifers a long in leaf season and allows for greater deposition rates in the winter months when particulate accumulation is greatest (Nowak et al., 2006). Saebo et al. (2012) suggest that the long and narrow nature of pine (*Pinus spp.)* needles makes it easier for particles to be deposited from the air and that conifers should be utilized in the greatest numbers in urban tree planting projects. Conifers have also demonstrated greater velocities of deposition with increases in wind speed, which has been attributed to smaller leaves and a more complex shoot structure in comparison to broadleaf trees (Freer-Smith et al., 2004).

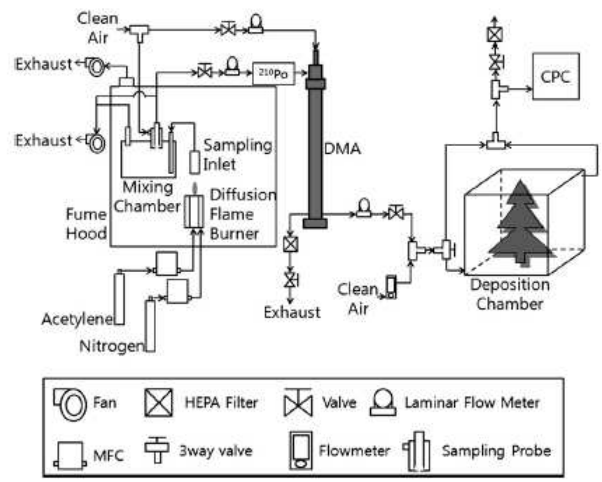

Figure 2 – A schematic representation of Hwang et al.'s (2011) deposition chamber utilized to simulate the flow of particulate matter associated with the combustion of gasoline in automobiles

When comparing broadleaf tree species, surface texture is of the utmost importance when determining particle capture effectiveness. Rough and hairy leaf surfaces have higher capture efficiencies than smooth, leathery, and waxy leaves (Beckett et al., 1998; Becket et al., 2000a; Becket et al., 2000b; Hwang et al., 2011; Kardel et al., 2011). The presence of trichomes gives broadleaves a hairy texture and leads to an increased surface area that increases dry deposition rates (Kardel et al., 2011). In addition to leaf texture, the bark of broadleaf trees plays an important role in the capturing of particulate matter with rough and sticky bark from sap excretions leading to increased deposition (Beckett et al., 1998).

The role of a waxy cuticle in terms of capture efficiency is a debatable topic in the literature. Many studies support the idea that leaves with waxy surfaces have decreased deposition rates, as particles in an air stream are likely to flow over these hard and smooth surfaces (Beckett et al., 2000a; Freer-Smith et al., 2004). The hydrophobic behavior of waxes would also lead to a decrease in wet deposition (Freer-Smith et al, 2004). Saebo et al. (2012), however, compared PM accumulation on 22 species of trees and found that those with waxy leaves demonstrated the highest levels of accumulation. In conifers, waxy cuticle development has also been shown to be a significant factor in the retention of PM. During cuticle development, PM can become permanently trapped in the wax, thus reducing the amount of resuspended particulates through normal meteorological events (Kardel et al., 2011).

In order to quantify capture effectiveness of different species widely used in urban planting, Donovan et al. (2005) developed the Urban Tree Air Quality Score (UTAQS) system which was based on both detrimental VOC emission rates and beneficial deposition rates on a per species basis. Air parcels were measured using the Cambridge tropospheric trajectory model of chemistry and transport (CiTTyCAT) and species were given scores and divided into one of three categories: species that greatly improves air quality, species that moderately improves air quality, and species that worsen air quality. The results of the study can be seen in Table 1 and Figure 3. Beneficial species reduced O_3 and NO_2 photochemical formation and reduced HNO_3 concentrations through deposition. The UTAQS system can help urban planters and other professionals select species for planting that can both mitigate the effects of VOCs and increase PM capture. ■

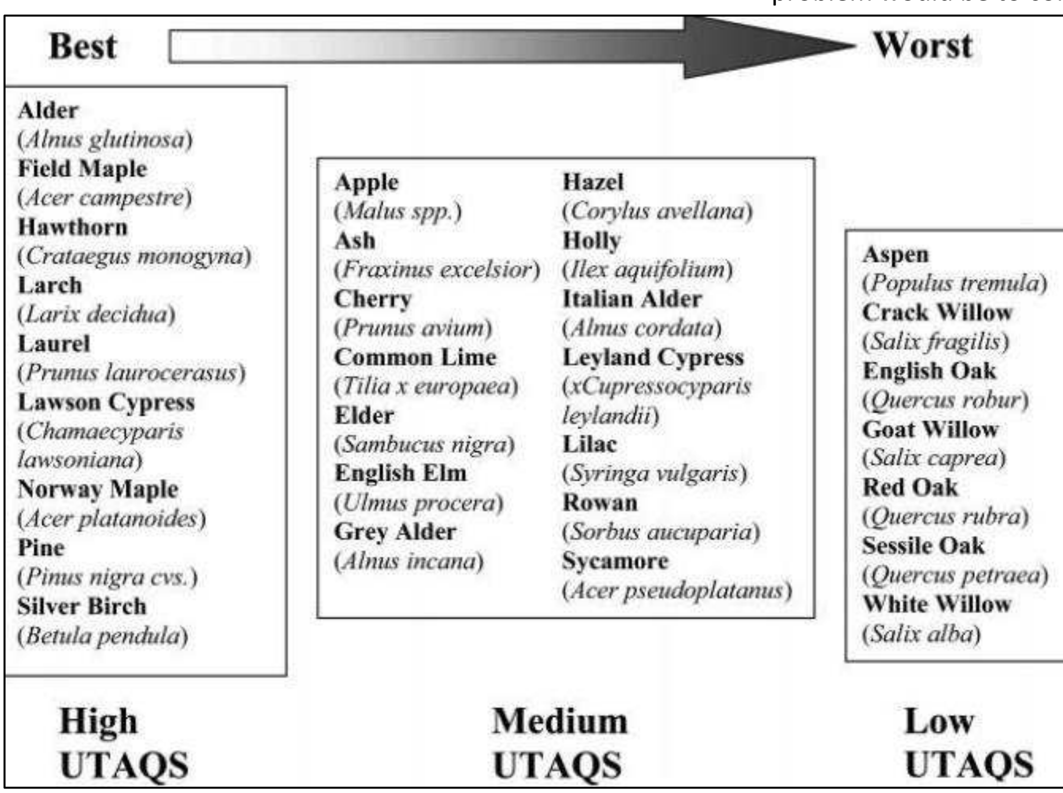

Figure 3 – Urban tree air quality scores of the 30 most common species of trees in the West Midlands, UK. From Donovan et al. (2005).

Planting Techniques and Locations

In addition to identifying effective tree species for PM capture, it is equally important to determine how trees are to be planted in an urban environment (arrangement and density) and where they should be planted to maximize effectiveness. In addition to mitigating air pollution, tree planting should improve the appearance of towns, provide shaded regions to reduce urban heat island effects, and buffet noise and wind. Common practices are to create screens of shrubs, followed by a row of deciduous broadleaves, and finally supplemented with coniferous trees at the back of the vegetation buffer. This model is effective at filtering PM and provides the coniferous trees with a protective barrier from harsh pollutants such as road salt (Freer-Smith, 2004).

In regards to PM concentrations in a large urban conurbation, an urban pulse has been noted to be greatest in central city locations with suburban and rural areas downwind also showing a greater concentration of PM in air and soil samples (Harrad and Hunter, 2006). It can be hypothesized that the best areas to plant trees for air quality control are where air pollution and human population densities are high and tree land cover is low (Morani et al., 2011). Based on this idea, it may seem that the best way to combat this problem would be to concentrate planting projects in inner city locales. Wania et al. (2012), however, stress that dense planting of vegetation in urban canyon settings could actually decrease air quality. Dense tree cover in urban canyons has the potential to decrease wind speeds and inflow resulting in poor ventilation and particle dispersion.

A better way to determine where planting will effectively reduce urban PM loading is to look at areas with high future planting potential (FPP). FPP is highest away from city centers where brownfield and green sites such as parks, gardens, and arable land are already present (Bealey et al., 2007; McDonald et al. 2007). The two aforementioned studies examined areas in the highly industrial areas of the West Midlands and Glasgow, UK. The results of FPP computer modeling suggest that suburban planting will result in a greater reduction in air pollution than highly concentrated planting in city centers based upon greater FPP values. In terms of what this means for reducing urban PM concentrations, a wider scale approach focused on heavy planting near suburban and rural areas should be the focus of management programs. ■

Drawbacks of Trees for PM Capturing

Though the effectiveness of trees for capturing atmospheric PM is well documented, trees are by no means a perfect answer to improving air quality in urban areas. As Donovan et al. (2005) accounted for in assessing UTAQS, differing tree species can either act as sinks or sources of O_3 and may in fact lesson air quality in a particular area. Biogenic volatile organic compounds (BVOCs) are produced by numerous species and can lead to the formation of large quantities of secondary PM. BVOCs, such as isoprene and monoterpenes, photochemically react with NO_x and create O_3 as a byproduct. Additional tree-based pollution occurs in the form of pollen, which is greatly produced by preferred coniferous species and can lead to seasonal respiratory complications (Beckett et al., 1998).

scenario description: existing urban forest + 20% of the following:	change in concentration (%)					UTAQS option				
	O_3[b]	NO_2[c]	HNO_3[d]	NO	PAN	O_3	NO_2	HNO_3	$O_3 + NO_2$	$O_3 + NO_2 + HNO_3$
U1: English oak (*Quercus robur* L.)	2.9	2.7	−5.0	−8.0	20.6	−3.85	-1.58×10^{-02}	1.22	−3.87	−2.65
U2: White willow (*Salix alba*)	2.4	2.5	−3.3	−6.6	16.6	−3.17	-1.47×10^{-02}	0.81	−3.19	−2.38
U3: Crack willow (*Salix fragilis*)	2.3	2.5	−3.1	−6.4	16.3	−3.06	-1.49×10^{-02}	0.76	−3.07	−2.31
U4: aspen (*Populus tremula*)	1.9	1.4	−4.5	−5.3	14.8	−2.53	-8.03×10^{-03}	1.10	−2.54	−1.44
U5: Sessile oak (*Quercus petraea*)	1.2	1.5	−1.6	−3.4	9.8	−1.53	-9.02×10^{-03}	0.39	−1.54	−1.15
U6: Red oak (*Quercus rubra*)	1.0	1.2	−1.4	−2.9	8.3	−1.35	-6.71×10^{-03}	0.36	−1.36	−1.00
U7: Goat willow (*Salix caprea*)	0.8	0.9	−1.4	−2.2	7.1	−0.99	-4.97×10^{-03}	0.34	−1.00	−0.65
U8: lilac (*Syringa vulgaris*)	−0.1	0.0	−0.3	0.1	2.0	0.09	9.33×10^{-05}	0.07	0.09	0.16
U9: sycamore (*Acer pseudoplatanus*)	0.0	0.3	−0.4	−0.2	3.3	0.06	-1.93×10^{-03}	0.11	0.06	0.17
U10: Mountain ash (*Sorbus aucuparia*)	−0.1	0.1	−0.2	0.1	2.2	0.18	-5.18×10^{-04}	0.06	0.18	0.24
U11: apple (*Malus spp.*)	−0.1	−0.1	−0.5	0.4	2.1	0.12	4.28×10^{-04}	0.12	0.12	0.24
U12: elder (*Sambucus nigra*)	−0.2	0.0	−0.2	0.5	1.9	0.20	-1.67×10^{-04}	0.05	0.20	0.24
U13: Common lime (*Tilia x europea*)	−0.1	0.2	−0.4	0.0	2.8	0.17	-1.32×10^{-03}	0.09	0.17	0.26
U14: holly (*Ilex aquifolium*)	−0.2	0.0	−0.4	0.5	2.0	0.21	1.85×10^{-04}	0.09	0.21	0.29
U15: Italian alder (*Alnus cordata*)	−0.2	0.0	−0.2	0.5	1.9	0.25	-2.12×10^{-04}	0.05	0.25	0.30
U16: hazel (*Corylus avellana*)	−0.2	0.1	−0.3	0.5	2.1	0.27	-3.33×10^{-04}	0.06	0.27	0.33
U17: Leyland cypress (*x Cupressocyparis leylandii*)	−0.2	0.0	−0.4	0.5	2.0	0.23	2.38×10^{-04}	0.11	0.23	0.34
U18: Grey alder (*Alnus incana*)	−0.2	0.0	−0.2	0.6	1.9	0.28	-2.32×10^{-04}	0.06	0.28	0.34
U19: cherry (*Prunus avium*)	−0.2	0.0	−0.3	0.6	1.9	0.27	8.17×10^{-05}	0.07	0.27	0.34
U20: English elm (*Ulmus procera*)	−0.2	0.0	−0.3	0.6	1.9	0.29	-1.32×10^{-04}	0.07	0.29	0.36
U21: ash (*Fraxinus excelsior*)	−0.2	0.0	−0.2	0.6	1.9	0.32	-2.22×10^{-04}	0.05	0.32	0.38
U22: Field maple (*Acer campestre*)	−0.3	0.0	−0.2	0.7	1.9	0.34	-2.27×10^{-04}	0.05	0.34	0.39
CU: existing composition	−0.3	0.0	−0.2	0.7	1.9	0.34	-2.27×10^{-04}	0.05	0.34	0.39
U23: alder (*Alnus glutinosa*)	−0.3	0.0	−0.2	0.5	1.9	0.40	-2.65×10^{-04}	0.06	0.40	0.46
U24: laurel (*Prunus laurocerasus*)	−0.2	−0.2	−0.8	0.6	2.1	0.29	1.26×10^{-03}	0.19	0.29	0.48
U25: Lawson cypress (*Chamaecyparis lawsoniana*)	−0.2	−0.2	−0.9	0.5	2.2	0.26	1.44×10^{-03}	0.22	0.26	0.48
U26: hawthorn (*Crataegus monogyna*)	−0.2	−0.4	−1.2	0.5	2.3	0.24	2.24×10^{-03}	0.28	0.24	0.53
U27: Norway maple (*Acer platanoides*)	−0.3	−0.2	−0.8	0.5	2.1	0.37	1.24×10^{-03}	0.19	0.37	0.57
U28: Silver birch (*Betula pendula*)	−0.3	−0.5	−1.4	0.5	2.3	0.33	2.74×10^{-03}	0.34	0.33	0.67
U29: larch (*Larix decidua*)	−0.4	−2.2	−5.2	0.5	3.6	0.50	1.28×10^{-02}	1.28	0.51	1.79
U30: Austrian, Corsican, Maritime pine (*Pinus nigra*)	−0.8	−4.9	−10.8	1.9	5.1	1.11	2.84×10^{-02}	2.65	1.14	3.79

[a] Runs are listed in rank order of multi-UTAQS option ($O_3 + NO_2 + HNO_3$). CU output values: $[O_3]$ = 66 ppb; $[NO_2]$ = 0.9 ppb; $[HNO_3]$ = 12 μg m⁻³; [NO] = 0.1 ppb; and [PAN] = 1.3 ppb. [b] 8 h running mean on fifth day. [c] Maximum 1 h mean on fifth day. [d] 24 h running mean on fifth day.

Table 1 – Urban tree air quality scores of different species of trees. From Donovan et al. (2005).

Another concern is that trees do not act as permanent sinks for particles captured through atmospheric deposition. Trees will only hold onto deposited particles until equilibrium is met; at saturated levels, particles will be lost to allow for the capture of new particles. Birch trees, which had the highest UTAQ amongst broadleaf, deciduous trees, reach equilibrium in only six days (Mitchell et al., 2012). In addition, the senescence of deciduous leaves leads to concentrated litter fall that is heavily concentrated with particulates. Leaf fall with high levels of heavy metals can lead to tree mortality owing to reduction in soil microbial activity around roots (Peachey et al., 2009). Finally, waxy cuticles on leaves can also be degraded, and as stated earlier, in regards to conifer cuticle formation, large amounts of PM can be stored in waxes Saebo et al., 2012).

Trees themselves can become susceptible from the exposure to pollutants. Occlusion of stomata leads to a decrease in gas exchange. Particulates can also form a crust-like layer on leaves, which has negative impacts on bud break, pollination, light absorption, and reflection. With their success in both wet and dry deposition of particulates and lack of seasonal senescence, conifers have a propensity to hold onto PM for long periods of time, and therefore are subject to suffer physiological effects from high toxin load (Beckett et al., 1998). ■

Urban Planting in Durham, NC

Ultimately, the effectiveness of urban green space management resides at the municipal level. Urban forestry departments must evaluate environmental processes and patterns at a local scale in order to successfully implement ecosystem services such as addressing stormwater management issues (i.e. increased impermeable service and atmospheric PM deposition leading to pollution of stormwater systems and urban watersheds). Traditionally, the production of environmental services has been seen more often as an objective rather than a traditional service by municipal foresters. In the past five years, however, urban planting projects have become more specifically designed to mitigate urban environmental issues. Tree planting projects have focused on using both long lived species to function as carbon sinks and early-successional, fast-growing species to quickly sequester carbon. In regards to stormwater mitigation, the importance of tree plantings to reduce PM deposition and runoff in areas with high levels of impermeable surface has gained significant momentum in the new approach of mitigating stormwater related issues in urban

Figure 4 – Various green spaces within Durham, NC. *Upper Left:* Urban tree planting on a parking garage for the Duke Clinical Research building. *Upper Right:* A loblolly pine (*Pinus taeda*) stand within the Duke Forest containing both mature trees and reprod. *Lower Left:* A large green space between the Durham Performing Arts Center and American Tobacco Campus. This area represents a site with planting potential within the downtown area. *Lower Right:* Cedar trees (*Cedrus spp.*) planted next to the Durham Performing Arts Center. Tree plantings can be used for a wide variety of reasons. Here they function as a sound buffer against the building's large air-conditioning units.

areas (Young, 2010).

With over 1,800 acres of public parkland, the 7,060 acre Duke forest, 15 golf courses, and agricultural landscapes in the northern portion of the county, Durham, North Carolina is an urban area with numerous possibilities for urban planting. The Urban Forestry Division (UFD) within the city of Durham, North Carolina (2013) utilizes urban tree planting projects for windbreaks, screening, and dust reduction. The UFD recommends using a step like arrangement of deciduous trees and conifers with preferred species including red maple (*Acer rubrum*), crepe myrtle (*Lagerstroemia indica*), Southern magnolia (*Magnolia grandiflora*), redbud (*Cercis canadensis*), holly tree (*Ilex attenuate*), honeylocust (*Gleditsia triacanthos*), Eastern red cedar (*Juniperus virginiana*), Japanese cedar (*Cryptomeria japonica*), sweetgum (*Liquidambar styraciflua*), blackgum (*Nyssa sylvatica*), and several species of oak (*Quercus spp.*). Businesses and public areas in downtown Durham have increased green space in recent years, and by coupling continued tree plantings within available urban space with larger scale plantings in surrounding suburban and rural areas that have high FPP, the Durham municipality can continue to make positive strives to minimize amounts of PM that affect both air and water quality (Figure 4). ■

Conclusions

Urban tree planting is an effective stormwater management strategy that demonstrates positive results in regards to reducing primary and secondary atmospheric particulate levels. In regards to tree species selection for effective atmospheric deposition, coniferous trees with needle-like leaves have been widely supported as the most beneficial. For broadleaf tree selection, species with large, rough-surfaced leaves are effective. Areas within an urban conurbation must be examined for FPP levels, which oftentimes results in outlying suburban and rural areas as planting sites with the greatest PM mitigation capabilities. As the management of stormwater continues to gain importance in both state and local policy, municipal forestry departments must continue best management practices in regards to urban forestry projects aimed at reducing atmospheric particulate matter. ■

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Land Use and Water Quality Analysis of Falls Lake and Jordan Lake Watersheds

Fatima Hashmi

Rapid urbanization as a consequence of population surges in urban areas and economic growth significantly alter natural landscapes. Natural forests and wetlands are cleared to make room for industrial and residential areas, roads, parking lots, and agricultural/cultivated land. Such deforestation leads to overall increase in impervious cover across a landscape and fragmentation of forests. These impacts are commonly known to degrade water quality and habitat in rivers and streams (Doll et al 2002). Research has shown that as much as 10% of impervious cover can have significantly degrading impacts on stream quality by causing channelization, and increasing sedimentation. Sedimentation occurs when deforestation exposes soil to water erosion (Doll et al 2002). Deforestation also results in the decline of organic inputs such as leaf litter and woody debris into streams which are important for maintaining aquatic habitats (Sweeny et al 2004). Runoff from agricultural land affects water quality as nutrients like nitrates, phosphorous, and sulfates are leached from the soil into runoff flowing to streams leading to potential eutrophication (Lai 1997). Effluent discharge from industrial plants also contaminates watercourses. This results in degraded water quality and aquatic habitats.

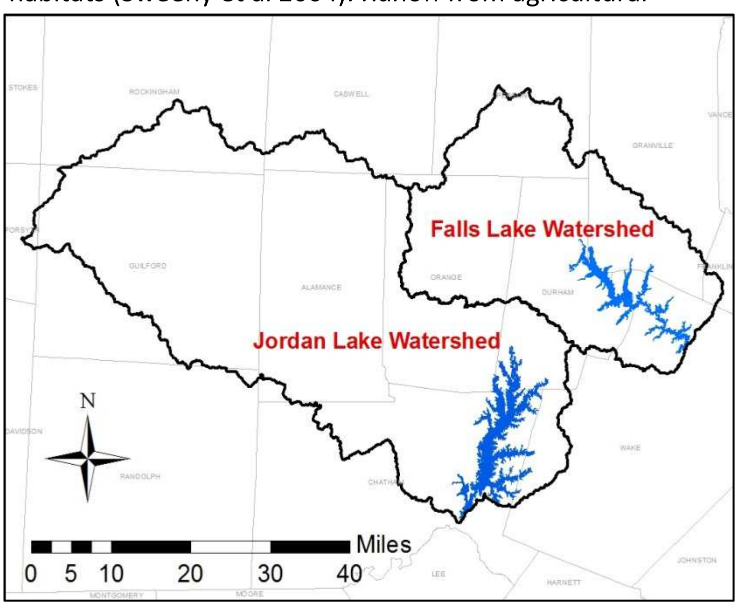

Figure 1 – Watersheds of Falls and Jordan Lakes.

In recent years, efforts have been made to reverse the effects of land use change of water quality and aquatic habitats by limiting the concentration of pollutants in industrial effluent, and establishing riparian buffers along streams and rivers. More recently, the ability of forests to provide clean water for human consumption and maintain biodiversity has been termed as an 'ecosystem service'. The Ecological Society of America (2000) defines ecosystem services as the benefits people receive from natural landscapes such as include flood reduction, better water quality, and recreation. Valuing forests as an ecosystem service translates into economic benefits as well. For example, the state of Virginia which saved about $5 billion per year in water quality services from conserving and protecting their forests (Paul, 2011). Even New York City's watershed protection program that diverted funds from water treatment facilities to preserving riparian forests lead to about $6 billion in savings (Whelan, 2010).

The discussion above highlights the significance of natural land use patterns in maintaining water quality in rivers and stream from not just an ecological perspective but from an economic one too. For the Triangle region of North Carolina, the Falls Lake and Jordan Lake reservoirs (Figure 1) are critically important sources of drinking water. However the watersheds of these reservoirs are vast with significant urban development and present many management issues and challenges for local county governments to address in terms of maintaining water quality and aquatic habitats. The North Carolina Division of Environment and Natural Resources (NCDENR) and its Division of Water Quality (NC DWQ) have played important roles in outlining management plans for nutrient and TMDL reductions in these lakes (NCDWQ, 2009).

The objective of this analysis is to contrast and compare land use patterns in the Falls Lake and Jordan Lake watershed and their water quality metrics as well. As discussed earlier, land use changes impacts the quality of natural waterways in a watershed, therefore observing land use patterns can indicate the level of degradation of streams, rivers and lakes.

Falls Lake Watershed

Falls Lake is also known as the Falls of the Neuse reservoir, and its watershed is known as Falls Lake Watershed, or the Upper Neuse Watershed. Falls Lake is a man-made reservoir on the Neuse River in the Triangle region of North Carolina and was constructed in 1983 by the U.S. Army Corp of Engineers. Within the Triangle Region, the Falls Lake Dam is located close to Raleigh in Wake County and the reservoir extends into Durham County. The reservoir is about 5,000 hectares in size. The watershed of Falls Lake extends into Wake, Durham, Orange, Pearson and Granville Counties. The watershed is about 199,809 hectares in size and home to 90,000 people (NCDENR, 2009). The headwaters of the watershed are made up of the Flat, Little and Eno Rivers, while other major tributaries include the Ellerbe, Knap of Reeds and Ledge, Kick, Little Lick and Beaverdam Creeks (NC DWQ, 2009). Besides the Falls Lake reservoir, the watershed includes nine other reservoirs for public water supply. These reservoirs together supply drinking water to 450,000 people (NCDENR, 2009).

Given the size of the watershed and its reservoir, as well as its role in providing drinking water to many residents, the watershed has been subjected to environmental management plans since the 1980s when Falls Lake was declared a Nutrient Sensitive Water (NSW). Since then efforts have been made to reduce phosphorous and nitrogen concentrations in the lake (NCDENR, 2009).

Jordan Lake Watershed

The Jordan lake watershed is about twice as large as the Falls Lake watershed with an area of about 436,000 hectares. The watershed is located in the piedmont region of North Carolina within the Cape Fear River Basin and extends across a total of ten counties in North Carolina including Durham. Like the dam on Falls Lake, the Jordan Lake Dam was constructed in 1983 and has since been used as drinking water resources, for flood control, conservation of wildlife and fish and recreation. The Jordan Lake reservoir was a result of a dam on the Haw River, just upstream of the confluence between Haw River and Deep River (Town of Apex, 2009). The Jordan lake watershed has been segmented into three sub watershed for management purposes and includes the Haw, Upper Hew Hope and Lower New Hope arm (NC DWQ, 2007).

Methodology

Since the objective of this paper is to compare and contrast land use patterns within each of the watersheds, the methodology consisted of geospatial analysis using ArcGIS (ESRI 2009). Watershed boundaries for Falls and Jordan Lake were obtained from the NC DENR, DWQ website. The shapefiles for the watersheds were downloaded and projected in North American Datum 1983. The North Carolina stream network and county boundaries were obtained from the Clean Water Management Trust Fund (CWMTF) in Raleigh.

In order to determine land use patterns within the watersheds, the National Land Use and Land Cover (LULC) raster datasets (30m by 30m pixel size) for the state of North Carolina were obtained by downloading from the USGS National Map Viewer. This LULC was developed by the USGS in 2006 and is referred to as the National Land Cover Database (NLCD). The NLCD 2006 is the latest land cover database available by the USGS and is collected by satellites. Previous NLCDs were collected in 2001 and 1992 (NLCD 2012). The NLCD consists of the following 20 land cover classifications

(NLCD 2012). The watershed boundaries were overlaid on top of the NLCD raster dataset in ArcGIS and land use patterns as proportion of total watershed area were extracted.

In order to compare water quality metrics between the watersheds, water quality data for the lakes were obtained from the USGS National Water Information System database. The USGS has specific sites around and at water bodies across the U.S. which record stream flow and water quality data. For both lakes, a number of sites were present, but only three sites for Jordan Lake and 2 sites for Falls Lake recorded water quality data. These sites contained pdf documents (created by the USGS) of water quality data between October 2010 and June 2011. In order to streamline the analysis of the water quality data, the reports for each site were combined, and monthly averages were determined for six water quality parameters, pH, specific conductivity, nitrate, phosphate, sulfate, and organic carbon.

Results and Discussion

As can be seen from Figure 1, the Jordan Lake watershed is almost twice as large as the Falls Lake watershed. The geospatial analysis of land use classes within these watersheds indicates almost equal percent of the various land cover classes in both watersheds (Table 1) especially in the case of wetlands, cultivated crops, shrubs and water classes. Figure 2 and Figure 3 show the land use cover within the Falls Lake and Jordan Lake Watersheds. Overall, the Jordan Lake watershed had more development (includes all

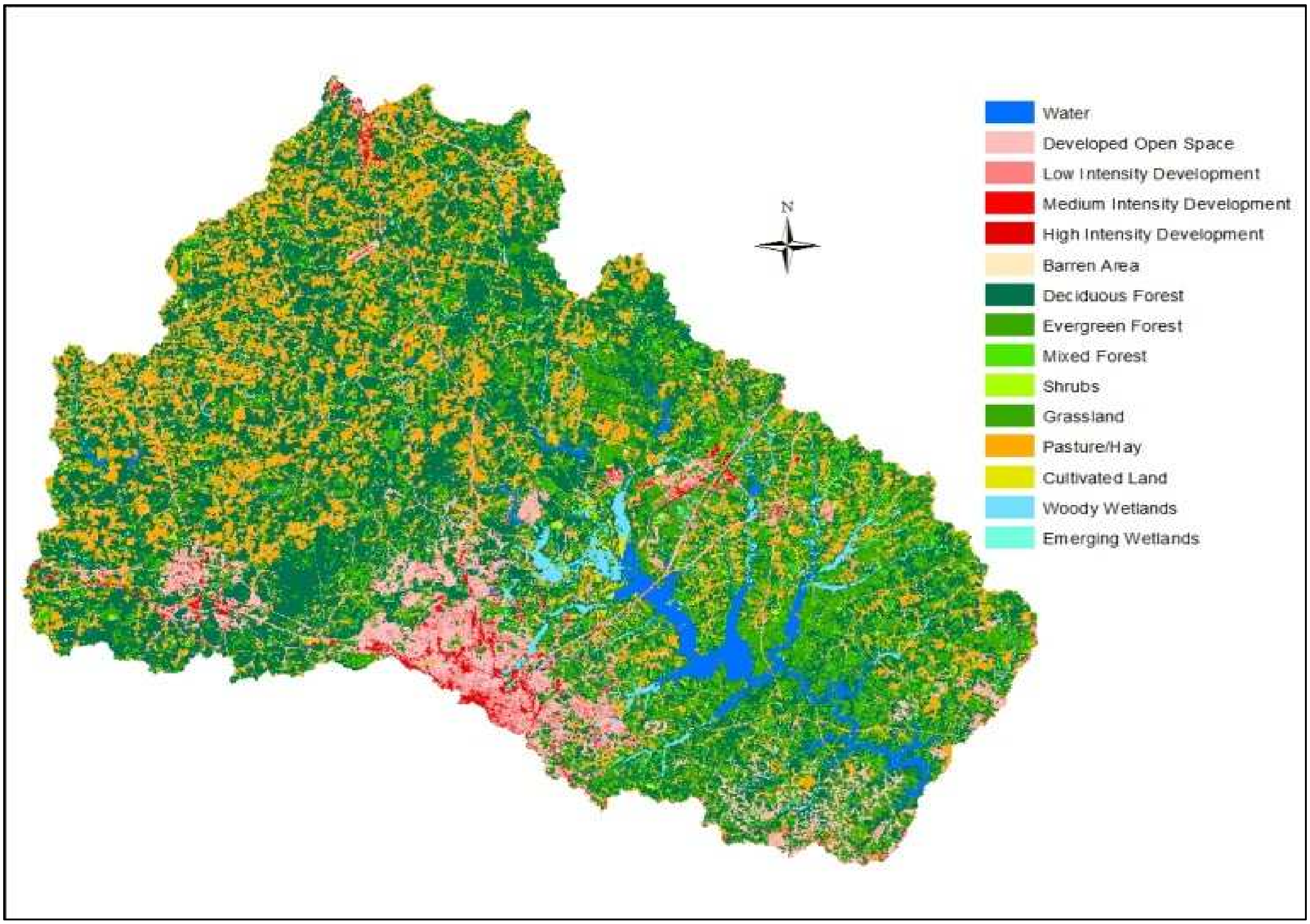

Figure 2 – Land Use and Land Cover Distribution in Falls Lake Watershed.

developed from open space to high intensity development) than Falls lake watershed. About 20% of the area within the Jordan Lake watershed is developed while only 14% of the Falls Lake watershed is developed. In terms of total forest cover (includes deciduous, mixed and evergreen), Falls Lake watershed has higher forest cover as percent of total watershed area than Jordan lake watershed, with 57% and 47% forest cover respectively. In terms of agricultural land cover, classified as pasture/hay and cultivated crops, Falls Lake watershed is lower than Jordan Lake, with 18% and 23% agricultural land cover respectively. Deciduous forest cover, pasture/hay and ever green forest cover are the dominant land use categories in both watersheds.

The averages of the water quality measurements (as determined by USGS at the specific gage sites) are shown in bar charts in Figure 4. The United States Environment Protection Agency does not have a set standard for specific conductivity, but states (on its website) that distilled water range for specific conductivity is generally between 0.5 to 3 microsiemens per cm (µs/cm) while rivers that support good fishery have a range between 150 to 500 µs/cm (U.S. EPA, 2012). According to the bar chart for specific conductivity, the range, in all months (from October to June) for both lakes is between 100 to 300 µs/cm, and hence is within the range for supporting fisheries. The pH values in both lakes is also within the U.S. EPA standards for Drinking water (between 6.5 and 8.5).

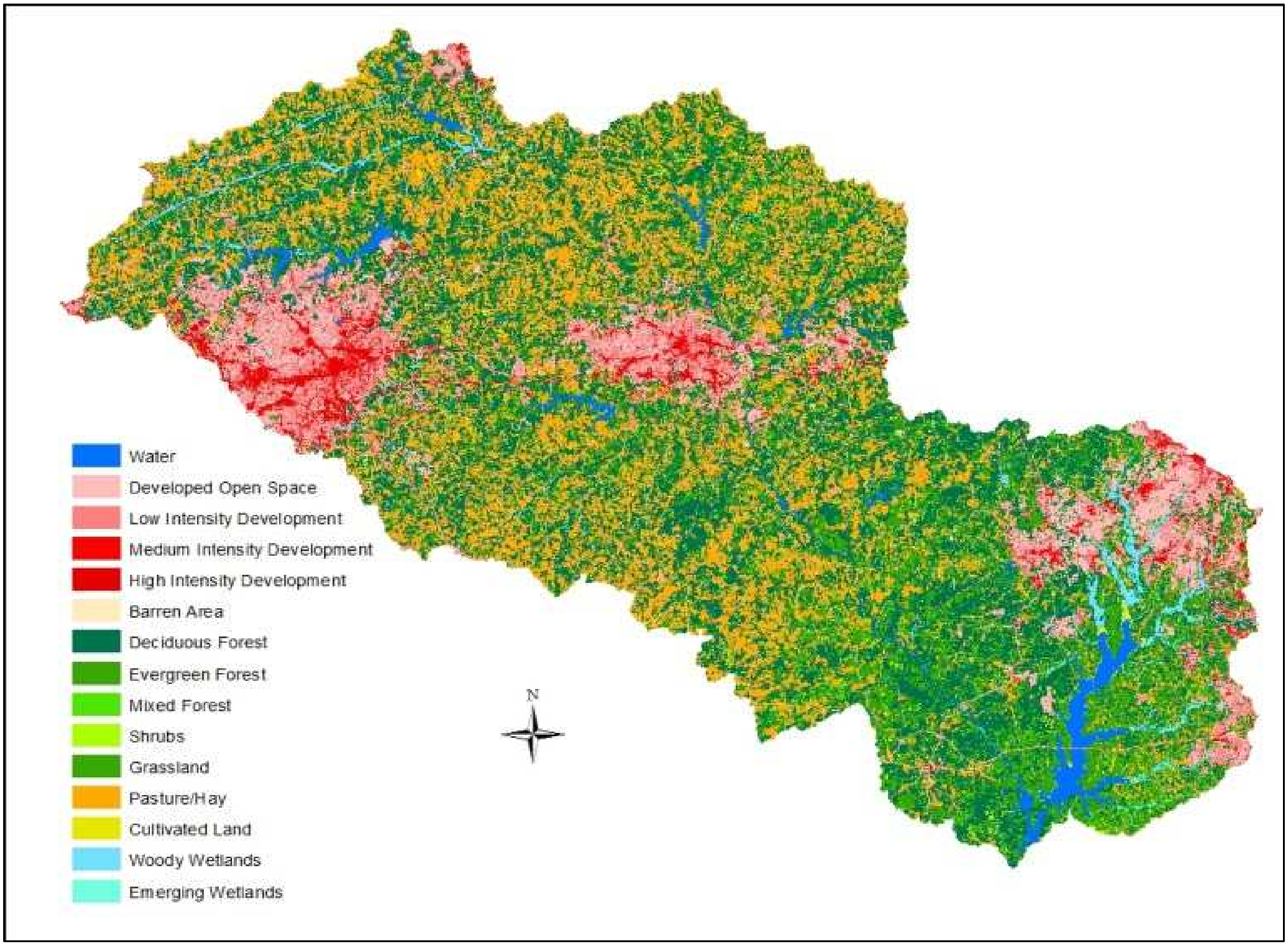

Figure 3 – Land Use and Land Cover Distribution in Jordan Lake Watershed.

Sulfate and nitrate concentrations in Falls Lake and Jordan Lake are also within the EPA's drinking water standards of 25 mg/L and 10 mg/L respectively.

The U.S. EPA does not have drinking water standards for organic carbon and phosphorous both of which are slightly higher in Falls Lake than in Jordan Lake across the months. In the remaining four water quality parameters, Jordan Lake concentrations are generally higher than Falls Lake concentrations.

It appears that the water quality of the lakes is within the set standards of the U.S. EPA despite that fact that the watersheds of these lakes are highly developed. This could be attributed to the past attention by the NC DENR and NC DWQ to protect these watersheds and the lakes from degradation and deteriorating water quality. In 2003, the Upper Neuse River Basin Association (UNRBA) developed an Upper Neuse (or Falls Lake) Watershed Management Plan aimed at monitoring nutrient loadings and water quality in the watershed. This plan was the result of funding provided by the NC General Assembly and had a few key objectives; to protect drinking water sources in the watershed, protect riparian and aquatic habitat, and maintain adequate water supply. These objectives were defined in light of the rapid development taking place in the river basin and the need to proactively take measures to ensure that water resources remained protected (UNRBA, 2003).

Before the 2003 watershed management plan, in 1983 Falls Lake had been identified as a "Nutrient Sensitive Waters" by the North Carolina Environmental Management Commission. Since then the watershed has been under constant management for phosphorous loads in the lake, which were significantly high in the lake. Even with phosphorus reductions, eutrophication continued to be a problem so in 1998 the Neuse River Basin NSW Management Plan was implemented to reduce nitrogen in the lake by regulating agricultural runoff (NCDENR, 2009).

Like Falls Lake, in 1983 the Jordan Lake reservoir had also been designated as "Nutrient Sensitive Waters". This classification was based on similar grounds as those for Falls Lake; effluent or discharge from wastewater and agricultural lands were increasing nutrient concentrations in the reservoir. Under this classification and the subsequent 1997 Clean Water Responsibility Act, total nitrogen and total phosphorous permits were issued to wastewater treatment facilities limiting their concentration in discharge. In 2003, nutrient modeling of the reservoir and its watershed indicated it to be highly impaired. Based on the outcome of this model, a nutrient management strategy plan was developed for the reservoir in 2007 (NCDENR, 2007).

This brief overview of the classification of both lakes as nutrient sensitive and subsequent management plans to limit nutrient loadings may have resulted in the relatively low nutrient concentration as shown in the water quality analysis of this paper.

LULC Classification	Falls Lake	Jordan Lake
	% Area of watershed	% Area of watershed
Water	2.90	2.31
Open Space Developed	9.58	10.50
Low Intensity Developed	2.75	5.96
Mild Intensity Developed	0.95	2.19
High Intensity Developed	0.28	0.88
Barren land	0.11	0.08
Deciduous Forest	39.01	34.34
Evergreen Forest	13.29	10.53
Mixed Forest	4.73	2.92
Shrub/Scrub	1.38	1.24
Grassland/Herbaceous	5.22	4.06
Pasture/Hay	16.76	22.64
Cultivated Crops	0.89	0.72
Woody Wetlands	2.09	1.53
Emergent Wetlands	0.06	0.07

Table 2 – Land Use and Land Cover Classification.

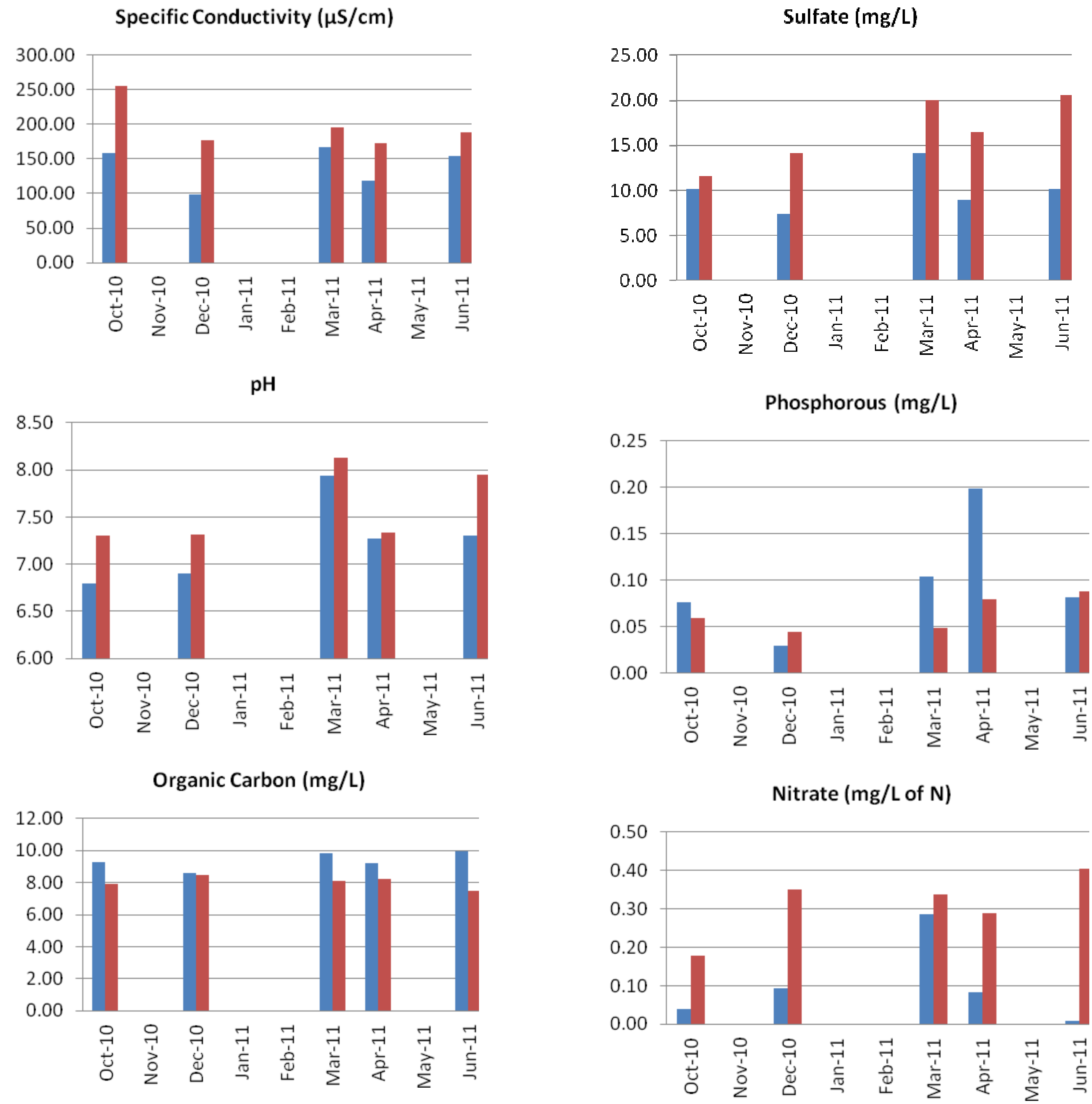

Figure 4 – Water Quality Comparison between Falls (blue bars) and Jordan Lake (red bars)

Conclusion

The land use and land cover analysis of both watersheds indicates similar patterns; with the Jordan Lake Watershed only slightly more impaired than the Falls Lake watershed in terms of development and forest cover. The water quality analysis shows that water quality was only slightly more degraded in the Jordan Lake watershed as well. Four of the six water quality parameters showed higher concentrations for Jordan Lake than Falls Lake. These findings perhaps indicate than on average, Jordan Lake's more degraded land use could be responsible for the slightly more degraded water quality as compared to Falls Lake. However this analysis is overly simplified and makes many assumptions. There could also be other reasons for water quality impairment, such as management issues. The Jordan Lake watershed is much larger than Falls Lake and therefore it may be more challenging to manage nutrient pollution sources and loadings.

However, given that both watersheds are currently under extensive watershed management plans it seems unlikely that conditions would worsen in either of the watersheds in the next few decades.■

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