Chemical and Biological Contamination of Stormwater Detention Pond Sediments in Coastal South Carolina

Final Project Report

prepared by

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Executive Summary

Stormwater ponds are a common management practice to protect the water quantity and quality of runoff entering natural receiving waters. Ponds are popular among developers for a number of reasons including the fact that they provide open space and wildlife habitat, provide fill material, can be aesthetically pleasing, and require little maintenance. However, the sediment which accumulates in the ponds must periodically be removed in order to maintain pond efficiency. Currently, in South Carolina, there are no requirements that stormwater pond sediments be tested for chemical or biological contaminants prior to sediment removal. Recent reports from elsewhere have characterized stormwater pond sediments, and suggest that contaminant levels can vary widely from background levels to enrichment that is several orders of magnitude greater than background. The purpose of the current project was to characterize the chemical and biological contaminants in stormwater pond sediments here in coastal South Carolina.

Stormwater ponds (16 individual ponds) were selected from the suburban areas of Charleston, Beaufort, Georgetown and Myrtle Beach. These ponds represented a subset of stormwater ponds sampled in 2006 as part of a SCDHEC-OCRM water quality survey. Based upon information provided in the original SCDHEC permit, all stormwater ponds were classified into one of four land use classes: golf course, low density residential, high density residential, and commercial. In addition, two manmade ponds not known to be receiving any stormwater runoff were selected as reference ponds. Personnel from SCDHEC-OCRM provided us with information concerning permit age, drainage area, impervious cover, pond surface area, maintenance activities, and pesticide usage for 12 of the 16 stormwater ponds. Permit ages ranged from 4 to 12 years old with a median age of 10 years old. Drainage areas ranged widely from a size of 0.71 ha to 360.0 ha; however, the median value was 4.76 ha. Pond surface area also ranged widely from 0.04 ha to 2.27 ha, with a median value of 0.15 ha. Impervious cover ranged from 0% to 77.0% with a median value of 49.3%. Based upon survey results from homeowner's associations, property managers and pond maintenance firms, sediment removal has not occurred in any of these ponds. Several of the ponds are routinely treated with copper-based products to control aquatic vegetation.

During June 2007, sediment samples from one pond inlet and pond center were collected from these 16 stormwater and 2 reference ponds and analyzed for grain size, total organic carbon (TOC), polycyclic aromatic hydrocarbons (PAH), selected acid-soluble metals, selected pesticides and other organics, fecal coliforms, and toxicity. The physical characteristics of the stormwater pond sediments varied widely from muds (>50% clay) to sand (<15% clay); however, the majority of stormwater ponds were characterized by mixed (15-50% clay) sediment types. Distributional patterns of sediment particles were evident between the two sampling locations within ponds: clay content was significantly higher in pond centers, while sand content was significantly higher near pond inlets. Total organic carbon content varied widely from 0.1 to 15.0%; however, unlike particle sizes, distributional patterns for TOC were not evident.

In this project, the 16 individual PAH analytes that the U.S. Environmental Protection Agency (US EPA) considers to be "priority pollutants" were analyzed. Levels of the sum of these PAH analytes (ΣPAH_{16}) were significantly higher in the sediments of commercial ponds (24,371.6±6,415.7 ng/g) (mean±standard error) compared to that of reference (1,276.7±695.7 ng/g), low density residential (508.6±129.5 ng/g) and high density residential ponds (956.2±432.1 ng/g). These levels of PAHs are generally on the same order of magnitude as those of other suburban stormwater ponds reported in the literature; however, one commercial pond (Pond 87) had sediment levels of ΣPAH_{16} (159,040.8 ng/g) that were seven times higher than the mean value for other commercial ponds in this study. In general, there was no consistent pattern in PAH levels with regard to sampling location (inlet or pond center) among the various land use classifications. Isomer ratio analysis suggested that the predominant source of PAHs in these stormwater ponds were pyrogenic (e.g., derived from combustion); however, many ponds had a PAH signature consistent with mixed petrogenic (e.g., uncombusted fuel), and pyrogenic sources.

For metals, land use was a significant factor for both lead and zinc, with commercial ponds again having significantly higher levels than various other land use classes. Although not statistically significant, levels of copper also tended to be elevated in commercial, golf course and low density residential ponds compared to reference ponds. Levels of metals reported here are similar to that reported in the literature for other suburban stormwater ponds. Distinct distributional patterns were evident with respect to sampling location within a pond. Sediment levels of aluminum and cadmium were significantly higher at the pond inlets, whereas levels of copper and zinc were significantly higher in the pond center. Levels of lead also tended to be elevated in the pond centers compared to the pond inlets, but this relationship was not statistically significant. These distributional patterns can be explained in terms of the affinity of certain metals to adsorb to clay particles, and the fact that the sediments in the pond centers had significantly higher clay content compared to the pond inlets. The fact that aluminum and cadmium were elevated at the pond inlets suggest that these metals are less mobile in stormwater pond sediments compared to other metals.

Only a few pesticides were frequently detected in stormwater pond sediments. The organophosphate chlorpyrifos was the most frequently detected pesticide, found in 9 of the 16 ponds sampled. Less frequently detected were the organochlorides chlordane, DDD and DDE, found in only 2 of the 16 ponds. Pesticide levels in pond sediments were independent of land use and sampling site within a pond. In general, the polybrominated diphenyl ether (PBDE) congeners, used widely as flame retardants, were not detected in stormwater pond sediments. Only two congeners, both penta-PBDEs, were detected in the sediments of one commercial pond (Pond 68).

In general, sediment fecal coliform levels ranged from low to moderately high when compared to other environmental media. Fecal coliform levels in stormwater ponds were independent of land use and sampling location, and were generally similar to those levels found in reference ponds. It is noteworthy that high levels of fecal coliforms were found at the inlet of one commercial pond (Pond 68; 5,620 MPN/g) and at the center of another commercial pond (Pond 70; 19,400 MPN/g). These values, which were nine and eleven times higher than the mean values for other commercial ponds, are on the same order of magnitude as highly contaminated river sediments.

Multiple regression analyses indicated that sediment levels of PAH, Cu, Pb, Zn and chlorpyrifos could be modeled using several independent variables. In most models, the only two significant factors were drainage area and pond surface area. Of these two variables, drainage area was generally the most important variable, explaining 72.5%, 61.1% and 59.0% of the variability in the PAH, Pb and Zn data. These results suggest that in addition to land use, drainage area may be an important determinant of stormwater pond contamination.

As part of an initial risk assessment of these sediments, toxicity tests and screening level assessments were performed. Toxicity tests demonstrated that there was little evidence of acute toxicity of these sediments to the amphipod *Hyalella azteca* in short-term (10 day) exposures. By contrast, screening level assessments, based largely upon chronic endpoints, demonstrated that the sediments of several commercial ponds, and one residential pond with a large drainage area, exceeded both ecological and human health benchmarks for PAHs. In general, metal levels in stormwater pond sediments were below that of ecological and human health concern. The only exception was Zn, which exceeded ecological benchmarks in most commercial ponds. Sediments from nine stormwater ponds had levels of the pesticide chlorpyrifos which exceeded ecological benchmarks, but none of these ponds had levels that were above the benchmarks for human health concern. By contrast, two commercial ponds had levels of DDD and DDE which exceeded benchmarks for both ecological and human health concern.

These results demonstrate that the sediments from commercial ponds, and those residential ponds with large drainage areas, have the potential to be contaminated. Furthermore, these sediments may pose ecological risks to wildlife *in situ*, and may pose human health risks once removed from the ponds and used as fill material. Finally, these findings suggest that land use, in combination with drainage area, could be used to provide guidance to resource managers concerning the need for analytical testing in order to evaluate appropriate sediment disposal options.

Introduction

Stormwater management ponds, including detention and retention ponds, are an increasingly common feature of the landscape of coastal South Carolina. According to Siewicki *et al.* (2007), there are approximately 8,114 stormwater ponds in coastal South Carolina, and this number appears to be increasing at a rate of 13% per year. Stormwater ponds are primarily used by developers to receive stormwater runoff and address water quantity and water quality concerns of natural receiving waters. Primary benefits of stormwater ponds include reducing runoff peaks, removing sediment and chemical contaminants, and enhancing overall stormwater quality through a variety of physical, chemical and biological processes (Urbanos & Stahre, 1993). Stormwater ponds are also popular with developers because they can be used to irrigate golf courses and residential landscaping, provide open space and recreation, enhance aesthetics, provide a natural environment for wildlife and vegetation, require little maintenance, and their presence increases property values. In fact, US EPA (1995) has identified several case studies nationwide where properties located within sight of a stormwater pond sold at premiums of between 10 to 150% compared to those properties with no water view.

Because stormwater ponds are designed to remove suspended sediment from incoming stormwater, there are two direct consequences for the pond bottom. First, many chemical contaminants are bound to suspended solids in stormwater, and as these solids settle out in the pond, there exists the potential that certain contaminants in bottom sediments would increase over time. If these contaminants are bioavailable to epibenthic and pelagic organisms in the pond, they then could pose a risk to wildlife using the pond, such as wetland bird species and overwintering ducks. Second, the settled material in stormwater is expected to slowly fill in the pond over time, thereby reducing the detention time of water in the pond. This, in turn, could reduce the designed efficiency of the pond and jeopardize the quality of nearby natural receiving waters.

The solution to maintaining the efficiency of stormwater ponds is periodic sediment removal. Current recommendations in South Carolina are that sediment be removed from stormwater ponds roughly every 10 years (Reynolds *et al.*, 2005). Other states have similar guidelines. For example, in Georgia, it is recommended that sediment removal around the pond inlets occurs every 5 to 7 years (Atlanta Regional Commission, 2001). Although the timing of sediment removal is important from a pond performance perspective, there are also important sediment handling and disposal perspectives which must be considered. Foremost among these issues is (1) to what extent are these stormwater pond sediments contaminated and (2) what risks, if any, do they pose to wildlife and human health both *in situ* and after sediment removal?

Stormwater management regulations in South Carolina

In the coastal counties (=coastal zone) of South Carolina, land disturbing activities are regulated by the SC Department of Health and Environmental Control – Office of Ocean and Coastal Resource Management (SC DHEC-OCRM). For land disturbing activities that drain into a single outlet, such as a river, stream, tidal creek, or salt marsh, the stormwater management plan must address both the quantity and quality of the runoff (SC DHEC, 2003; 2005; 2006). One recommended practice to treat the runoff is through the construction of a stormwater pond. There are three basic types of stormwater ponds: (1) wet detention ponds are a permanent pool of water designed to temporarily store stormwater before being discharged downstream, (2) dry detention ponds are a temporary pool of water designed to temporarily store stormwater before being discharged downstream, and (3) retention ponds are a permanent pool of water that uses its entire volume for stormwater storage with no discharge.

In South Carolina, regulations specify that an approved stormwater management permit is required if the land disturbing activity involves 1 or more acres and is not within ½ mile of a receiving water or if the activity is more than ½ acre and is within ½ mile of a receiving water. Stormwater ponds must be designed as both quantity and quality structures. Stormwater ponds, whether detention or retention, must be able to store and release the first ½ inch of runoff from the site over a 24-hour period. The storage volume must be designed to accommodate, at least, ½ inch of runoff from the entire disturbed site. Stormwater ponds must also retain 80% of the suspended sediment load during the construction phase. Many contaminants, such as polycyclic aromatic hydrocarbons (PAH), pesticides and metals, bind to sediments and particulate matter, and are retained in the ponds.

According to current regulations, permitted stormwater ponds, once constructed, must be maintained in order to ensure proper functioning. Maintenance responsibilities are often assumed by homeowner's associations, property management firms, and local governments. Routine preventative maintenance of stormwater ponds include: (1) ensuring inlet and outlet channels are clear of debris, (2) ensuring that embankments have remained stable, and (3) removing sediments which have built-up on the pond bottom (SC DHEC, 2003; 2005; 2006).

Chemical contamination

Several studies outside the state of South Carolina have characterized the chemical contamination in bottom sediments of stormwater ponds (Table 1). These studies collectively suggest that contamination is related to current land use, historical land use, and age of the pond. Based on a survey of the literature, stormwater ponds draining commercial areas and highways generally have higher levels of contaminants than those draining residential areas. In one study conducted in Ontario, Canada, the bottom sediment in a stormwater pond draining a 13-ha commercial plaza was contaminated with both polycyclic aromatic hydrocarbons (PAHs) and metals (Pb, Cu, and Zn) (Marsalek *et al.*, 2002). In fact, levels of these metals in pond sediments were so elevated the removed sediment would not meet the criteria for reuse as residential fill material nor could they be disposed of at municipal landfills (Marsalek & Marsalek, 1997). In a similar study of the bottom sediments of a highway stormwater pond in Oxted, Surrey, UK, even higher levels of total PAH and metals (V, Cr, Mn, Fe, Co, Ni, Cu, Pb, and Zn) were found (Kamalakkannan *et al.*, 2004).

Age of the pond is also an important determinant in the extent of chemical contamination in stormwater pond sediments. In a study involving three stormwater ponds in Pinellas County, FL, the oldest pond (a 30-year old pond in Largo, FL) was highly contaminated by PAH (total PAH >7,000,000 ng/g), metals (including Cr, Cu, Pb, Fe, and Zn) and pesticides (dieldrin, hepatoclor, and DDT) (Fernandez & Hutchinson, 1993). The second-oldest pond studied (a 20 year old pond in Clearwater, FL) also was contaminated with PAH, metals, and pesticides, but not to the same extent as the oldest pond. A newly constructed pond (a 1 year old in Seminole, FL) had levels of PAH, metals and pesticides that were below their respective limits of detection. Interestingly, the authors suggest that the presence of the high levels of organochloride pesticides in the sediments of the oldest pond pose a risk to higher trophic levels, including native wildlife, from bioaccumulation of these compounds through the food chain (Fernandez & Hutchinson, 1993).

Historical use of the land prior to stormwater pond construction can also influence bottom sediment contaminantion. In a study of 24 stormwater ponds in Phoenix, AZ elevated levels of organochloride pesticides (DDT, DDE, chlordane, dieldrin, and toxaphene) were found in ponds from a variety of land uses, including residential, industrial, and commercial (Parker *et al.*, 2000). The presence of organochloride pesticides in these pond sediments is somewhat surprising given the fact that they were banned by the United States Environmental Protection Agency (US EPA) in the 1970s and 1980s. Parker *et al.* (2000) attributed their presence to the historical use of these pesticides on these former agricultural lands and the comparatively long half-lives of these compounds in the environment. It is interesting to note that a number of stormwater ponds built in coastal South Carolina drain historic agricultural fields, and for the same reasons listed above, may also be retaining organochloride pesticides in their bottom sediments.

Other factors, including percent impervious cover in the pond drainage area, may also be contributors in the extent of chemical contamination in stormwater pond sediments. In a study of commercial stormwater ponds in Minneapolis, MN, metal concentrations increased with increasing paved area in the drainage area of the pond (Polta *et al.,* 2006). In a recent study on Kiawah Island in South Carolina, low levels of PAH (total PAH = 174 ng/g dry weight) were found in the sediments of a stormwater pond associated with low density residential housing and a golf course (Flemming *et al.*, in press). These low levels of PAHs were attributed to the fact that these pond drainage areas have relatively low percentages of impervious cover. It is important to note that Kiawah Island is unique in that residential and resort development has progressed in an ecologically-sensitive manner, with a high value placed on the protection of natural resources. And, although PAH levels in these pond sediments were relatively low, this study did not address chemicals associated with golf course maintenance, including insecticides and herbicides.

Toxicity

Only a few studies have investigated the toxicity of bottom sediments in stormwater ponds. However, based upon published ecological screening benchmarks values for effects range-low (ERL) and effects range-median (ERM) (Long *et al.*, 1995), some adverse effects on biota would be expected at most of the stormwater ponds described above. For example, the ERL and ERM values for total PAH are 4,022 ng/g and 44,792 ng/g, respectively (Long *et al*., 1995). Based on these values, sediment toxicity would be expected to occur at stormwater ponds studied in Largo and Clearwater, FL, and to a lesser extent, ponds studied in Surrey, UK and Ontario, Canada (Table 1). In fact, laboratory toxicity tests for the Ontario, Canada stormwater pond (contaminant data previously described) indicated that pond sediments collected from several areas, including the pond inlet and outlet, were toxic (Marsalek *et al*., 2002), although the chemical source of the toxicity was not investigated. Sediment Microtox® tests exhibited a range of percent inhibition from 25 to 50%, indicating moderate toxicity of the sediments to bacterium. Sediment toxicity tests using the nematode (*Panagrellus redivivus*) exhibited mortality ranging from 92% to 100%, indicating these bottom sediments were highly toxic to this species. Sediments collected both upstream and downstream of this stormwater pond in Ontario were significantly less toxic using both of these tests.

PAHs are not the only toxic chemical found in stormwater pond sediment; pesticides and metals would also be expected to contribute to their overall toxicity. As previously described, a study of 24 stormwater ponds in Phoenix, AZ found elevated levels of organochloride pesticides (DDT, DDE, chlordane, dieldrin, and toxaphene) and metals (As, Cd, Cu, Pb, and Zn) in ponds from a variety of land uses, including residential, industrial, and commercial (Parker *et al.*, 2000). Toxicity tests using amphipods (*Hyalella azteca*) demonstrated that sediments from stormwater ponds from all three types of land uses had a wide range of toxicity with mortality ranging from 6–100%; however, sediments from 12 of the 24 ponds studied exhibited 100% mortality (Parker *et al.*, 2000).

Age of the pond appears to be an important determinant in the extent of toxicity of the stormwater pond sediments. Karouna-Renier & Sparling (1997) investigated the toxicity of the sediments from nine stormwater ponds, between the ages of 3 and 5 years old, in Prince George's, Anne Arundel, and Howard Counties, Maryland. Sediments from these ponds, which drained a variety of commercial, highway, and residential land uses, exhibited no acute or chronic toxicity to amphipods (*H. azteca*). Apparently, the contaminants in these sediments (1) had not yet accumulated to high enough levels to be toxic, and/or (2) were not bioavailable.

Impacts on native wildlife

One recent study suggests that heavy metals associated with stormwater pond sediments are contributing to adverse impacts on native wildlife. In a study of 12 stormwater ponds in Washington DC, elevated levels of Zn and Cu were found in bottom sediments of ponds and the carcasses of 8-day old red-winged blackbirds (*Agelaius phoeniceus*) inhabiting these ponds (Sparling *et al.*, 2004). Although nesting success in stormwater ponds was comparable to national averages, it is interesting to note that sediment Zn concentrations correlated with decreased clutch size, hatching success, and fledgling success. Sparling *et al.* (2004) attribute these decreases to the physiological stress associated with the Zn exposure.

Human health perspectives

Sediment screening benchmarks, called Preliminary Remediation Goals (PRGs), are available from the US EPA that incorporate maximum sediment chemical constituents, in combination with other factors, that can help determine associated human health risks (Smucker, 1998). Humans may be exposed to stormwater pond contaminants through contact with fill material following removal, groundwater infiltration, airborne dust during dry periods, and recreational contact (such as fishing or children playing near ponds). Previous studies have identified stormwater pond sediments having levels of chemical contaminants that exceed the published PRG values. For example, in Phoenix, Parker *et al.* (2000) found that <u>all</u> sampled stormwater ponds exceeded the PRG value for increased cancer risk for arsenic. Additionally, a few of the stormwater ponds sampled, including one in a residential area, exceeded the PRG values for organochloride pesticides (dieldrin, DDT, and toxaphene).

Project objectives

As stormwater ponds in coastal South Carolina age, responsible parties will need to perform sediment removal in order to maintain their designed efficiency as water quantity and quality structures. Currently, the regulatory authority for managing stormwater at the state level, SC DHEC, has no requirements that stormwater pond sediments be tested for chemical or biological contaminants prior to sediment removal. However, based upon a survey of the literature involving stormwater ponds outside of South Carolina, these sediments have the potential to be contaminated. And, under certain conditions, previous studies have demonstrated that these contaminants do pose risks to native wildlife. Given the extensive use of stormwater ponds as best management practices (BMPs) to control stormwater runoff from residential and commercial development in South Carolina, there is a critical need for further research into these important issues before widespread sediment removal occurs.

Therefore, the specific objectives of this project were to:

(1) Collect and analyze sediment from several stormwater ponds in coastal South Carolina for chemical (PAH, metals and pesticides) and biological (fecal coliform) contaminants.

(2) Quantitatively model the relationship between sediment contamination and land use, drainage area, age of pond, maintenance practices, and impervious surface in watershed.

(3) Identify contaminants of ecological and human health concern for each sampled stormwater pond using published screening benchmarks.

Materials and Methods

Pond selection, physical description, and maintenance history

During June 2007, a survey of sediment contamination was conducted in 16 stormwater detention ponds and two reference ponds in the coastal counties of South Carolina (Fig. 1). These ponds represented a subset of the 112 ponds surveyed in 2006 by SC DHEC–OCRM as part of a baseline water quality study. Specific ponds were chosen following consultation with SC DHEC–OCRM staff to ensure that ponds were sampled from a wide range of land uses (reference, golf course, low density residential, high density residential, and commercial) (Table 2). These stormwater ponds were generally freshwater, the only exception was Pond 89 which had a salinity of 11‰. Land uses and permit ages were identified directly from information provided in the original SC DHEC permit applications. Pond edges and drainage basins were delineated on a subsample of the surveyed stormwater ponds (12 of 16 ponds) and the 2 reference ponds using Environmental Systems Research Institute, Inc.'s ArcGIS® version 9.2. A combination of georeferenced engineering site plans, 1:24,000 United States Geological Survey topographic maps, and 2006 National Aerial Photography Program (NAPP) imagery were used to define the drainage basins. Areas were calculated using the "Calculate Area" tool located in the "Spatial Statistics Toolbox". The proportion of each watershed that represented impervious cover was estimated by overlaying a 10-meter grid to the NAPP imagery. The percent of impervious cover was calculated as the number of points that fell on roads, sidewalks, driveways, and structures divided by the total number of non-water points that fell within the drainage basin multiplied by 100. Drainage basin analysis was not performed on 4 stormwater ponds (Ponds 26, 57, 62, and 83) because engineering plans were unavailable. Delineation of these drainage basins was not attempted due to the complexity and intensity of stormwater management in an urban watershed.

A survey of maintenance activities in these stormwater ponds was conducted by SC DHEC-OCRM staff. Homeowner's associations, pond maintenance firms, and businesses were contacted and asked the following questions: (1) Have the ponds been dredged?, (2) Has vegetation been cleaned out of the ponds?, (3) Have pesticides and/or algaecides been applied?, (4) Have fish kills, discolored water and/or algae been present in the pond?, and (5) Are aerators present, and if so, are they to improve water quality or for aesthetics? Maintenance information was available for all 16 stormwater ponds sampled in this study.

Pond sampling

Within each pond, sediment was collected for physical, chemical and biological analyses from two sites. One site was located ~3 m from one of the major inlets of the pond (=Site A). Pond inlets were identified from permit applications and confirmed on-site. The second site was located in the center of the pond (=Site B). Sediment samples (~4 L) in 13 stormwater ponds and 1 reference pond were collected from a canoe using a stainless steel standard Ekman bottom grab sampler (15.24 cm x 15.24 cm x 15.24 cm) (Wildlife Supply Company, Buffalo, NY). These sediment samples were comprised of two composite sediment grabs. In three shallow stormwater ponds (Pond 26, 31, 83) and one shallow reference pond (Pond R1), sediment samples were collected using a 2.0 L stainless steel pot by wading to the respective sites (Site A and B). These sediment samples were also comprised of two composite grabs. All sediments, regardless of how they were collected, were homogenized on-site using a solvent-cleansed stainless steel bucket and spoon, then divided into six aliquots (see below). All aliquots were transported to the laboratory on ice. Between samples, all equipment was rinsed with pond water.

Physical characteristics of sediments

One sediment aliquot (~500 mL) was placed in a plastic bag for percent moisture, grain size and total organic carbon (TOC) analyses. Samples were stored frozen (-20°C) and processed in <30 days. Percent moisture was determined by drying 2 g of sediment at 105°C for 24 h, then calculating the difference between wet and dry weights. Grain size was determined using protocols modified from Plumb (1981). TOC analysis was performed by personnel at the SC Department of Natural Resources (SC DNR) using a Perkin-Elmer 2400 CHNS Elemental Analyzer at a 950°C combustion temperature.

PAH analysis

A second aliquot (250 mL) was placed in solvent-washed amber glass jars for PAH analyses. Samples were stored at 4°C and extracted in <7 days. PAHs from sediments were extracted using an ultrasonication method (Sun *et al.,* 1998). Sediment samples were prepared by first homogenizing them within amber jars with the use of a stainless steel spatula. Replicate samples of ~2 g were removed from the jar, placed on aluminum foil, covered loosely, and allowed to dry overnight at room temperature. Once dry, the samples were weighed and homogenized with a mortar and pestle. The homogenized sediment was ultrasonicated for 5 min in 10 ml of acetonitrile, and centrifuged at 3000 rpm for 20 min in glass centrifuge tubes. The supernatant was transferred to a clean bottle. The remaining sediment was ultrasonicated and centrifuged with an additional 10 ml of acetonitrile. Following the second centrifugation, any remaining sediment, was allowed to shake overnight on an orbital shaker with an additional 5 ml of acetonitrile. Processed samples were stored at -80°C, and filtered (0.2 µm, PTFE filter) immediately prior to HPLC analysis.

PAHs were quantified using reverse phase HPLC on a Perkin Elmer Series 200 HPLC system connected to a UV/Visible Absorbance Detector and a Fluorescence Detector in tandem. All detectors are part of the Perkin Elmer Series 200 system. Fifty microliters of sample or standard solution were injected directly onto a Hypersil Green PAH column (5µm, 100 x 4.6 mm), using a gradient elution of (time, % acetonitrile) 0,50; 5,50; 25,100 at 2 ml/min. The Fluorescence Detector was used for detection of all PAHs except acenapthylene, which fluoresces only weakly. Instead, it was quantified using the UV/Visible Detector. Peaks were recorded and quantified using TotalChrom software (version 6.2).

The low molecular weight (LMW) PAHs quantified in this study were naphthalene (NAPT), acenapthylene (ACNY), acenapthene (ACEN), fluorene (FLUR), phenanthrene (PHEN), and anthracene (ANTH). The high molecular weight (HMW) PAHs studied were fluoranthene (FLTH), pyrene (PYR), benzo[a]anthracene (B(a)A), chrysene (CHRY), benzo[b]fluoranthene (B(b)F), benzo[k]fluoranthene (B(k)F), benzo[a]pyrene (B(a)P), dibenz[a,h]anthracene (D(ah)A), benzo[g,h,i]perylene (B(ghi)P), and indeno[1,2,3,c,d]pyrene (INPY). Total PAH concentrations are reported as the sum of the 16 analytes ($\sum PAH_{16}$). All PAH concentrations are reported on a dry weight basis.

To determine the reliability of the PAH data, including efficiency of extraction methods and interferences that may be introduced during sample preparation, standard reference material and procedural blanks were analyzed. PAH extraction efficiency for sediments was determined using National Institute of Standards and Technology Standard Reference Material (SRM) 1944. Four samples of SRM 1944 were processed by using the same methods as the stormwater pond sediment samples to determine extraction efficiencies. The average extraction efficiency of $\sum PAH_{16}$ was 100.4 ± 8.2%, the average extraction efficiency of the 13 certified analytes was 94.9 ± 11.2%, and efficiency was greater than 85% for all analytes except NAPT and ANTH at 46.88 and 48.02%, respectively (Table 3). Using similar methods to those outlined here, Sun *et al.* (1998), Kayali-Sayadi *et al.* (2000), and De Luca *et al.* (2005) obtained comparable extraction efficiencies. For determination of method detection limits, eight samples of SRM

1944 were processed and analyzed using the same techniques as the other stormwater pond sediment samples.

Pesticide and PBDE analysis

A third aliquot was also placed in a solvent-washed amber glass jar (250 mL) for pesticide and PBDE analyses. Samples were stored at 4ºC until analysis. Samples were prepared for pesticide analysis using EPA Method 3540C followed by removal of sulfur with freshly cleaned copper (EPA Method 3660) and a florisil SPE cleanup (EPA Method 3620). Briefly, a weighed 10 g sample of wet sediment (excess water was decanted and foreign matter was removed) was mixed with 10 g of anhydrous sodium sulfate and 1.0 ml of a 10 ppm two-component pesticide surrogate solution containing dibutyl chlorendate and 2,4,5,6-tetrachloro-m-xylene (Supelco, Bellafonte, PA). In a few cases, additional sodium sulfate was required to ensure sample dryness, but no more than 15 g total was used. Samples were then packed into a soxhlet thimble and extracted for at least 16 hours with 300 ml of 1:1 methylene chloride:acetone. All glassware was pre-cleaned with three rinses of pesticide-grade acetone followed by three rinses with pesticide-grade methylene chloride. After cooling, the extract was passed through a column of sodium sulfate to ensure dryness then rotary-evaporated to 1 ml volume. The sample was transferred to a test tube with three additional washes of methylene chloride, then blown down with helium to a volume of 1 ml. Freshly acid-washed and thoroughly rinsed copper granules were added to remove sulfur and the solution transferred to a Florisil SPE tube. After rinsing with methylene chloride, a 1:1 mixture of ethyl acetate:methylene chloride was used to extract the pesticides. Samples were blown down to a 1 ml volume, capped, and stored in a freezer until analysis by GC-MS. In addition to samples, four 1.0 g portions of NIST SRM 1944, New York/New Jersey Waterway Sediment (dried) and four blanks (drying agent plus surrogate) were extracted and treated in the same manner.

Analysis was conducted by GC-MS using a Thermo Trace GC – Polaris Q MS (Thermo Fisher Scientific, Austin, TX) in both EI and NCI ionization modes using two separate injections. Calibration was conducted using external standards consisting of dilutions of single component solutions and multi-component mixtures. The GC method consisted of a 1 μL splitless injection of the methylene chloride solution into the injection port held at 200ºC. The initial oven temperature of 40ºC was held for 5 minutes followed by a ramp to 150ºC at 10 º/min then to 270ºC at 5 º/min and finally to 300ºC at 10º/min with a final hold time of 20 minutes. The helium carrier gas was held at a constant flow of 1 ml/min through the entire run. Extracted ion chromatograms were used for both EI and NCI analysis. Table 4 shows the ions used for quantitation, the estimated method limit of detection, as well as the mode (EI/NCI) used for each analyte. Surrogate recovery averaged 70% with a range of 21-178% (Table 5). If the highest recovery is removed (Sample 48A), the average is 67% with a range of 21-116%. In addition to surrogate recovery, results obtained for compounds in NIST SRM 1944 were compared with certified levels as shown in Table 6.

Metal analysis

A fourth aliquot was placed in an acid-washed plastic jar (500 mL) for metal analyses. Preparation of sediment samples for acid-extractable metals analysis was performed using a procedure similar to that described by others (Fan & Wang, 2003). Sediment samples were first dried in a 60ºC oven for several days to dryness. The samples were crushed and mechanically homogenized before weighing out two individual 0.25 g subsamples. The samples were placed in 50-ml plastic digestion tubes (Environmental Express, Mt. Pleasant, SC) and treated with 4 ml concentrated ultrapure HNO_3 (70%) plus 1 ml concentrated $HClO_4$ (60%). Samples were covered with plastic watch glasses and held at 90ºC for four hours. After cooling, the samples were diluted to 50 ml with ultrapure water and filtered with an in-tube Filtermate (Environmental Express, Mt. Pleasant, SC) apparatus. Samples were then capped and stored at room temperature until analysis. Four replicate extractions of NIST SRM 1944 were carried through the same procedure. Mean recovery of all metals analyzed (6 certified values, 1 reference value) was 99%, with a low of 73% (Fe) and a high of 151% (Al).

Analysis of Cu, Cd, Al, Fe, and Zn were conducted by flame atomic absorption on a Varian SpectrAA 200 (Varian Inc., Walnut Creek, CA) using an external standard calibration curve consisting of a minimum of five standards for each element prepared by dilution of single-element standard solutions (High Purity Standards, North Charleston, SC). Samples that were above the calibration range were diluted to fall within the range for analysis. Analysis of Pb and Cr were conducted by graphite furnace atomic absorption using a Varian SpectrAA Zeeeman 220 (Varian Inc., Walnut Creek, CA) using an external standard calibration curve consisting of seven standards for each element analyzed in triplicate. Standard solutions were prepared by dilution of single-element standard solutions. Samples that were above the calibration range were diluted to fall within the range for analysis. Table 7 contains the estimated limit of detection for each element analyzed.

Sediment fecal coliform analysis

A fifth aliquot of sediments was placed in a two, replicate plastic specimen cup (100 mL) and shipped overnight on ice to Commonwealth Labs in Greenville, SC. Sediment fecal coliform analysis was determined using a multiple tube technique (US EPA, 1978; APHA, 1989). Briefly, 30 g of well-mixed sample were transferred to a sterile blender containing 270 ml of sterile buffered water, then blended for 2 minutes. Diluted samples (1, 0.1, 0.01 and 0.001) were then inoculated into sterile fermentation tubes containing A-1 Medium and incubated. Most probably number (MPN) determinations were based upon probability tables provided in APHA (1989).

Toxicology

A sixth aliquot of sediment was placed in a 1 L amber glass bottle for toxicity testing. Sediments were stored at 4°C and held <30 days prior to testing (US EPA, 2001a). Toxicity tests were conducted using the standard amphipod (*Hyalella azteca)* 10-day survival test for sediments (ASTM, 2005). Amphipods kept in culture at The Citadel were originally obtained from Willow Swamp Road Pond (Pond R1), located in the Francis Marion National Forest. Amphipods were taxonomically identified by Mr. David Knott (Southeast Regional Taxonomic Center, SC Department of Natural Resources, Charleston, SC). Tests were conducted on only one site per pond (either Site A or Site B), usually whichever site was most contaminated (based upon preliminary PAH data). Test chambers were 250 mL beakers filled with 75 mL of sediment and 150 mL of overlying water. For each pond site tested, three replicate chambers were set up, each replicate containing ten amphipods (<14-day old). Water for the toxicity tests were collected from Pond R1. Overlying water in the test chambers was renewed every 24 hours, and *H. azteca* were fed once per day with Tetramin. Water quality was monitored throughout the 10-day exposure period. At the termination of the toxicity test, surviving *H. azteca* were recovered from the sediment by sieving (250 μm mesh size).

Screening Assessments

For each pond sampled, chemicals of ecological and human health concern were identified by calculating Hazard Quotients (the ratio of contaminant level in the sediment to published toxicological screening benchmarks). Screening benchmarks are contaminant-specific guidelines that identify levels that are potentially hazardous to wildlife and/or human health (Jones *et al.* 1997). In the current study, two sets of ecological screening benchmarks were used for sediment contaminants: (1) threshold and probable effect concentrations (TEC and PEC, respectively) derived from the Assessment and Remediation of Contaminated Sediment (ARCS) Project (US EPA, 1996; Jones *et al.*, 1997) and (2) sediment screening values (SSV) used by EPA Region IV (US EPA, 2001b).

TEC and PEC values were calculated by Jones *et al.* (1997) using a ranking method based upon freshwater sediment toxicity tests involving (1) reduction in survival, growth, or sexual maturation of the amphipod *Hyalella azteca* and (2) reduction in survival or growth of the midge *Chironomus riparius*. TEC values represent a concentration below which adverse effects to these organisms are not expected, in other words, these are possible-effects benchmarks. PEC values represent a concentration in which adverse effects to these organisms are likely to occur, in other words, these are probable-effects benchmarks. TEC and PECs were specifically chosen in the current study because they have been derived for a relatively large number of contaminants relative to other ecological screen benchmarks.

Sediment screening values (SSV) used by EPA Region IV are derived from statistical interpretation of effects databases obtained from the literature as reported by Long *et al*. (1995). These values are based upon observations of direct toxicity in marine environments. Current evidence suggests that toxicity in freshwater environments occurs at contaminant levels that are within an order of magnitude of that observed in marine systems (US EPA, 2001b). Regardless, these values are currently being used by EPA Region IV until separate freshwater screening values are developed. In the current study, the use of these ecological benchmarks (TEC, PEC and SSV) as screening tools assumes contact with the sediments by wildlife *in situ*, and not necessarily contact with the sediments following their removal from the pond.

In the current study, toxicological screening benchmarks for human health were Preliminary Remediation Goals (PRGs) for residential soil developed by US EPA Region IX (Smucker, 1998). EPA Region IV currently recommends their use as screening values for human health risks (US EPA, 2000). PRGs combine current human health toxicity values with standard exposure factors to estimate contaminant concentrations in soil that are considered to be health protective of human exposures (including sensitive groups), over a lifetime. Specifically, PRGs are chemical concentrations that correspond to fixed levels of risk (i.e., either a one-in-one million [10^{-6}] cancer risk or a noncarcinogenic hazard quotient of 1). In the current study, when a substance causes both cancer and noncancer (systemic) effects, the 10^{-6} cancer risk value was used as the PRG. Routes of exposure specifically considered in derivation of residential soil PRGs include ingestion, inhalation of particulates and volatiles, and dermal absorption (US EPA, 2004). In contrast to the use of ecological benchmarks described above, the use of PRGs as a screening tool in the current study assumes human contact with the sediment after its removal from the pond, not necessarily contact with the sediment *in situ*.

The screening benchmarks used in the current study, whether ecological or human health, are designed to determine whether contaminants warrant further assessment or are at a level that requires no further attention. Chemical concentrations above screening benchmark levels do not necessarily designate a site as "polluted" or trigger a federal regulatory response action. However, if a chemical concentration exceeds a benchmark value, the chemical is assumed to be of concern and that further evaluation of the potential risks that may be posed by the chemical is warranted.

Data analysis

Univariate and multivariate statistical analyses were performed using SAS Enterprise 4.0. Pond sediment data was analyzed using a two-way analysis of variance (ANOVA) to determine differences between the pond classes and sampling sites. Differences between sampling sites within a pond class were examined using least square mean contrasts, and differences among the pond classes were examined using Duncan's Multiple Range Test. For the PAH data, sampling site was not a significant factor, so the data between sampling sites within a land use class was pooled and analysis re-accomplished using a one way ANOVA. One way ANOVA was also performed on the mortality data from the toxicity tests. Linear regression analyses were used to determine the relationship between sediment contaminant concentrations and sediment TOC and clay content. In order to satisfy the homoscedasticity and normality assumptions, percentages of clay, sand, TOC and mortality were arcsine square root transformed, and levels of PAHs and metals were log transformed for all analyses. Data values greater than 1.5X outside the interquartile range were considered outliers and not included in the data sets for ANOVA or linear regression. All values are reported as mean ± standard error (SE), unless otherwise noted.

Spearman correlation analysis was used to determine the relationships among the various pond physical characteristics (e.g., drainage area, pond surface area, impervious cover in the watershed, permit age, TOC, percent sand and percent clay) and the various sediment contaminants. Multiple regression analyses (maximum r-squared improvement) was used to model the relationship between contaminant levels and potential determinants including drainage area, pond surface area, impervious cover in the watershed, and permit age.

Similarities in sediment contaminant composition among the 16 stormwater ponds and two reference ponds were analyzed by principle component analysis (PCA) using the PC-ORD statistical software, Ver. 5. PCA is a statistical procedure used to transform a large number of variables into a smaller number of uncorrelated factors, called principle components, for easier interpretation. The first principle component accounts for the highest variance, and each subsequent component explains as much of the remaining variability as possible. When displayed in a principle component cross-plot, ponds with a similar contaminant composition will be located near each other. Variables entered into the analysis included the sediment levels of the 16 individual PAH analytes, all seven metals, two pesticides (chlorpyrifos and DDE), and fecal coliform. To remove the effect of absolute concentration on the first principal component, all contaminant values were log transformed.

Results

Physical description and maintenance history

Permit age, drainage area, pond surface area and percent impervious cover were determined on a subsample of 12 of the 16 total stormwater ponds examined in this project. This subsample of stormwater ponds had permit ages ranging from 4 years old for Pond 7 to 12 years old for Ponds 26, 31 and 70 (Table 2). The median permit age was 10 years old. Drainage areas for these stormwater ponds ranged widely from a size of 0.71 ha in Pond 80 to 360 ha in Pond 62. However the median drainage area was 4.76 ha. Pond surface area also ranged widely from a surface area of 0.04 ha in Pond 24 to 2.27 ha in Pond 62, with the median pond surface area being 0.15 ha. Percent impervious cover in this subsample of stormwater ponds ranged from 0% in the drainage area of Pond 80 to 77.0% in the drainage area of Pond 48. The median percent impervious cover in the drainage areas of these stormwater ponds was 49.3%.

A survey of maintenance activities in all 16 stormwater ponds (conducted by personnel at SC DHEC-OCRM) found that, in general, maintenance is limited to removal of unwanted vegetation and debris, and the occasional use of algaecides (e.g. $CuSO_4$, chelated copper products, diquat and glyphosate) (Table 8). In some ponds, carp and other fish species have been introduced to help combat the vegetation problems. However, based on these survey results, it appears that none of the stormwater ponds in the current study have had sediment removed.

Physical characteristics of sediments

The distribution of sediment particle sizes varied widely among the 16 stormwater ponds surveyed (Fig. 2). Pond sediments ranged from muds (>50% clay) to sands (<15% clay); however, the majority of stormwater ponds were characterized by mixed (15-50% clays) sediment types. The clay content of pond sediments ranged from 0.3-79.5% with a mean of 23.9% (±3.9%) (± SE). There was no consistent pattern in clay content among the different land use classes ($p=0.7598$) (Fig. 3). The clay content of pond sediment from reference (27.3±18.3%), golf course (24.8±12.0%), low-density residential (15.6±5.8%), high-density residential (25.6±11.5%), and commercial (28.6±6.1%) were similar to each other. In general, the clay content of sediments from the pond center (32.7±3.9%) was significantly higher than that of the inlet (15.0±3.3%) ($p=0.0360$), but these differences within the various land use classes were not statistically significant. Sand content of the ponds ranged from 5.0 to 98.4% with a mean of 60.8% (±5.6%). There was no consistent pattern in sand content among the different land use classes ($p=0.7470$) (Fig. 4). Sand content of pond sediment from reference (56.7±21.1%), golf course (53.7±15.0%), low-density residential (71.6±10.4%), high-density residential (59.8±17.3%), and commercial (57.0±9.2%) were similar to each other. In contrast to the distributional pattern for clay, the sand content of sediments from the pond center (32.7±5.5%) were significantly lower than that of the inlet (75.8±7.4%) ($p=0.0113$), but these differences within the various land use classes were not statistically significant.

Total organic carbon of the pond sediment ranged from 0.1-15.0% with a mean of 3.7 (±1.0%) (Table 9). There were no consistent pattern in the distribution of TOC with regard to land use class or site ($p=0.3679$) (Fig. 5). The TOC content of pond sediment from reference (6.6±3.8%), golf course (5.0±1.7%), low-density residential (2.6±1.4%), high-density residential (2.9±1.5%), and commercial (3.5±1.1%) were similar to each other. Although not statistically significant, there was a trend toward higher TOC content in the center of the stormwater ponds compared to the inlet, which was opposite to that found for the reference ponds.

Polycyclic Aromatic Hydrocarbons

Both clay and TOC content of the pond sediments were positively correlated with concentrations of ΣPAH_{16}, ΣPAH_{LMW}, or ΣPAH_{HMW} (correlation coefficients ranged from 0.6609-0.8398) (Table 10). Regression models for TOC and clay content were generally significant, but were only weakly correlated (r^2=0.11-0.19) (Table 11). This suggested that TOC and clay content were not highly associated with the concentration of PAH in the pond sediment. Therefore, concentrations of PAH were not normalized for grain size or TOC content in the analyses described in this report.

Levels of ΣPAH_{16} were significantly higher in the sediments of ponds classified as commercial (24,371.6±6,415.7 ng/g) compared to that of reference (1,276.7±695.7 ng/g), low density residential (508.6±129.5 ng/g) and high density residential ponds (956.2±432.1 ng/g) ($p=0.0053$) (Fig. 6). Levels of ΣPAH_{LMW} were also significantly higher in commercial ponds (1,468.3±359.6) compared to low-density residential (224.8±23.4) ($p=0.0128$) (Fig. 7). In addition, commercial ponds had significantly higher levels of ΣPAH_{HMW} (22,003.0±5,782.3 ng/g) compared to reference ponds (567.3±317.9 ng/g), low-density residential ponds (342.4±125.5 ng/g), and high-density residential ponds (584.2±331.7 ng/g), respectively ($p=0.0054$) (Fig. 8). Levels of the individual PAH analytes were also generally higher in commercial ponds compared to reference ponds; the only exceptions were ACNY, FLUR, and ANTH (Table 12). It is important to note that at one commercial pond (Pond 87), the inlet had high sediment levels of ΣPAH_{16} (159,040.8 ng/g), ΣPAH_{LMW} (19,015.5 ng/g), and ΣPAH_{HMW} (140,025.4 ng/g). These values, which were 7-11X higher than the mean values for the other commercial ponds, were determined to be outliers and not included in the statistical analyses. In general, there was no consistent pattern in PAH levels with regard to sampling site (inlet or pond center) among the various land use classes.

Regression analysis indicated that PAH levels (ΣPAH_{16}, ΣPAH_{LMW}, and ΣPAH_{HMW}) in stormwater pond sediments could be modeled using several independent variables (Table 13). Possible entries into these models as independent variables included drainage area, pond surface area, permit age, and percent impervious cover in the drainage area. Parameter estimates for drainage area and pond surface area were the only variables that were significant, and therefore were the only ones included in the models. These models were able to explain 91.2%, 84.7%, and 81.8% of the variability in the sediment levels of

ΣPAH_{16}, ΣPAH_{LMW}, and ΣPAH_{HMW}, respectively. The single most important variable in these models was drainage area, which was able to explain 72.5%, 60.0%, and 71.6% of the variability in the data, respectively (Fig. 9).

The ratios of PAH concentrations for isomers with equal molecular mass but different thermodynamic stability can be used to identify probable PAH sources (Yunker *et al.*, 2002; Zheng *et al.*, 2002; Walker *et al.*, 2005; Brown & Peake, 2006). A single ratio can often be misleading so multiple ratios are often used. These ratios are useful indicators of PAH sources because PAHs of equal mass tend to behave similarly during transport and deposition and the ratios are preserved (Yunker *et al.*, 2002). Several PAH isomer ratios (FLTH/PYR and PHEN/ANTH) and the ratio of LMW to HMW (LMW/HMW) were calculated for the sediment samples to determine probable PAH sources (Table 14). In general, pyrogenic, or combustion sources of PAH are indicated by high FLTH/PYR ratios (>1), and low PHEN/ANTH and LMW/HMW PAH ratios (<10 and <1, respectively). By contrast, petrogenic, or uncombusted fuel sources of PAH are indicated by low FLTH/PYR ratios (<1), and high PHEN/ANTH and LMW/HMW ratios (>25 and >1, respectively). The FLTH to PYR ratios found in these stormwater pond sediments were >1, and ranged from 1.20 at Ponds 44 and 68 to 3.88 at Pond 70. The PHEN to ANTH ratios varied widely among stormwater ponds, ranging from 5.61 to 86.72. However, only two ponds (Ponds 68 and 83) had ratios <10. The majority of stormwater ponds (9 of 16 total) had PHEN/ANTH ratios >25. The LMW to HMW ratio in stormwater pond sediments were generally <1, but four stormwater ponds (Ponds 31, 7, 80, and 48) had ratios >1.

Metals

In general, both clay and TOC content of the pond sediments were positively correlated with metal concentrations (correlation coefficients ranged from 0.3905-0.8276) (Table 10); the only exception was TOC content and Cd, which were not correlated to each other. Regression analysis demonstrated that sediment levels of Al were moderately correlated to both clay and TOC content (r^2=0.45 and 0.49, respectively), and sediment levels of Zn were moderately correlated to clay (r^2=0.40). Although the majority of the other relationships between specific metals and either clay or TOC content were significant, these relationships were only weakly correlated (r^2 ranging from 0.14 to 0.25) (Table 15). Collectively, these results suggest that clay and TOC content was, at best, only moderately associated with metals in the sediments. Therefore, concentrations of metals in pond sediments were not normalized for grain size or TOC content in the analyses described in this report. Concentrations of metals were also not normalized to Al concentrations because the toxicological screening benchmarks (see below) are based upon absolute concentrations, not normalized values. In this regard, we felt that the data would be most valuable to regulatory agencies as absolute concentrations.

Levels of metals in the sediments at the inlet and center of reference and stormwater ponds are presented in Figures 10-15. The two-way ANOVA models were generally significant for all metals; the only exceptions were Cd and Fe (p=0.1117 and 0.8306, respectively) (Table 16). These two-way models revealed that sampling site (inlet or center) was a significant factor for Al, Cu, and Zn. And, although the two-way ANOVA for Cd was not significant, when the class variable was removed from the model and the data re-analyzed using a one-way ANOVA, sampling site also became a significant factor for this metal (p=0.0240). In general, levels of Al and Cd in the sediments from the pond inlet (site A) tended to be higher than that of the center (site B), but these differences within the various pond classifications were not statistically significant (Fig. 10 and 11). By contrast, levels of Cu and Zn tended to be higher at the pond center than at the inlet, but again, these differences were also not significant within the various pond classifications (Fig. 12 and 15, respectively). Land use class was a significant factor for Pb and Zn (p= 0.0090 and 0.0083, respectively) (Table 16). Levels of Pb were significantly higher in the sediments of commercial ponds (5.8±1.1 mg/kg) compared to that of golf course ponds (1.6±0.5 mg/kg) (Fig. 14). Levels of Zn were significantly higher in the sediments of commercial ponds (216.6±48.9 mg/kg) compared to reference (41.0±23.4 mg/kg), low-density residential (37.7±10.9 mg/kg) and golf course ponds (30.2±10.8 mg/kg) (Fig. 15). Although not statistically significant, levels of Cu also tended to be higher in commercial (115.3±66.2 mg/kg), golf course (90.8±28.6 mg/kg) and residential low density ponds (155.0±48.7 mg/kg) compared to reference ponds (12.2± 4.2 mg/kg) (p=0.0870) (Fig. 12).

Regression analysis indicated that levels of Cu, Pb and Zn in stormwater pond sediments could be modeled using several independent variables (Table 17). Possible entries into these models as independent variables included drainage area, pond surface area, permit age, and percent impervious cover in the drainage area. Parameter estimates for drainage area and pond surface area were the only variables that were significant for Cu and Zn, and therefore were the only ones included in the models. The single most important variable for Zn was drainage area, which was able to explain 59.0% of the variability in the data (Fig. 16). For Cu, regression models for drainage area and pond surface area alone were not significant. For Pb, two regression models were equally able to explain the data (Table 17). In model 1, Pb levels in sediments were dependent upon pond surface area alone, which was able to explain 71.5% of the data (Fig. 17). In model 2, Pb levels were dependent upon drainage area alone, which was able to explain 61.1% of the data (Fig. 18). Multiple regression analyses involving both pond surface area and drainage area together were not significant for Pb.

Pesticides/PBDE

Only one of the pesticides was frequently detected in stormwater pond sediments. The organophosphate chlorpyrifos (tradenames=Dursban® and Lorsban®) was found in the sediments of 9 of the 16 stormwater ponds sampled (Table 18). Chlorpyrifos levels in sediments were not correlated with either clay or TOC content (Table 10). Two-way ANOVA revealed that there were no differences in the levels of chlorpyrifos with regard to land use class or sampling site (p=0.7491) (Fig. 19). Chlorpyrifos levels in pond sediments from reference (<LOD), golf course (4.11±2.89 μg/g), low density residential (3.75±1.42 μg/g), high density residential (6.92±6.92 μg/g), and commercial (7.09±2.99 μg/g) were similar to each other. Regression analysis indicated that chlorpyrifos levels in stormwater pond sediments were dependent upon pond surface area (Table 19, Fig. 20). Other independent variables, such as drainage area, age, and percent impervious cover were not significant in the model.

Several other pesticides, and pesticide breakdown products, were infrequently detected in stormwater pond sediments. The oganochloride chlordane (either the alpha and gamma isomers) was detected in 2 of the 16 stormwater ponds (Table 18). Breakdown products of the organochloride DDT (DDD and DDE) were also detected in the sediments of 2 of the 16 stormwater ponds, and in both reference ponds. Other pesticide breakdown products detected in the sediments of only 1 stormwater pond include: (1) endosulfan sulfate, a breakdown product of the organochloride pesticide endosulfan, which was detected at the inlet of Pond 68, and (2) dichlorvos, a breakdown product of the organophosphate naled, which was detected in the center of Pond 7.

In general, the PBDE isomers, used widely as flame retardants, were not detected in stormwater pond sediments. Only 2,2',4,4',5-pentaBDE and 2,2',4,4',6-pentaBDE were detected at the inlet of Pond 68 at concentrations of 30.0 ng/g and 72.8 ng/g, respectively (Table 18). Although not specifically targeted in the present study, hexachlorobenzene, a historical fungicide and industrial solvent, was possibly detected in the sediments of the inlets at Pond 89 and 109, and oxadiazon (tradename=Ronstar®), a pre-emergent herbicide, was possibly detected in the sediments in the center of Pond 57 and at both the inlet and center of Pond 80. An unidentified chlorinated compound with a retention time of 14.7 minutes was also detected in Ponds 24, 26, 38 and 70.

Fecal Coliform

Fecal coliform levels in pond sediments ranged from 254-6,970 MPN/g with a mean of 1,079±242 MPN/g (Fig. 21). Two way ANOVA revealed that there were no differences in the levels of fecal coliform with regard to land use class or sampling site ($p=0.3475$). The fecal coliform levels of pond sediment from reference (1,044±441 MPN/g), golf course (773±147 MPN/g), low density residential (496±69 MPN/g), high density residential (490±78 MPN/g), and commercial (1,641±615 MPN/g) were similar to each other (Fig. 22). It is important to note that high levels of fecal coliforms were found at the inlet of one commercial pond (Pond 68; 5,620 MPN/g) and at the center of another commercial pond (Pond 70; 19,400 MPN/g). These values, which were 9X and 11X higher than the mean values for their respective classes, were determined to be outliers and not included in the statistical analyses. Correlation analysis revealed that levels of sediment fecal coliform were positively correlated with pond drainage area and pond surface area, as well as most chemical contaminants including PAHs and metals (correlation coefficients ranged from 0.4900 to 0.7449) (Table 10). However, regression analyses between sediment fecal coliform levels and these independent variables were not significant.

Toxicity Tests

Using the standard 10-day *Hyalella azteca* amphipod sediment test, mortality was generally low (≤10%) in sediments from the reference ponds and in the majority of stormwater ponds studied. The only exceptions were Pond 57 and 89, which each had an average mortality of 16.7% (Fig. 23). One way ANOVA revealed that mortality was significantly higher in sediments collected from golf course ponds (11.1±2.6%) compared to that of the reference ponds (1.7±1.6%) and commercial ponds (1.6±0.9%) ($p=0.0459$) (Fig. 24). Mortality in the golf course ponds was similar to that of the low density (4.7±2.9%) and high density (5.0±2.2%) residential ponds. Regression analysis revealed that acute toxicity in stormwater pond sediments was not dependent upon drainage area, pond surface area, pond age, or impervious cover in the watershed.

Principle Component Analysis

Principle component analysis (PCA) was used to investigate similarities in the correlation matrix of the chemical and biological contaminants among the 16 stormwater ponds and two reference ponds. The first principle component accounted for 50.1% of the variability in the data, while the second principle component accounted for 16.2% of the variability. Because the first two axes accounted for 66.4% of the total variability, they are the only two presented. PCA analysis delineated two distinct groups of stormwater ponds based upon their correlation matrices (Fig. 25). One group of stormwater ponds, consisting of Ponds 57, 24, 44, 48, 80, 31, 7, 89, 109, 70, and 62, had relatively low levels of chemical contaminants. A second group of ponds, consisting of Ponds 87, 83, 26, 38, and 68, had relatively high levels of contaminants. It is important to note that most of the stormwater ponds in the second group were classified as commercial ponds, with the exception of Pond 38. Pond 38 is a low density residential pond with a comparatively large drainage basin (6.83 ha). The two reference ponds (Ponds R1 and R2) had unique contaminant profiles and did not cluster with each other or the other two groups of stormwater ponds.

Screening Assessments

PAH

For the ΣPAH sediment levels (ΣPAH_{16}, ΣPAH_{HMW}, ΣPAH_{LMW}), only the ecological benchmark values for TEC and PEC are available. Several stormwater ponds had PAH sediment levels in at least one site (either pond inlet or center) which exceeded both the TEC and PEC values. For ΣPAH_{16}, four commercial ponds (Ponds 26, 68, 83 and 87) and one low density residential pond (Pond 38) had sediment levels which exceeded the PEC (Fig. 26). Sediment levels in one additional residential low density pond (Pond 24) and one golf course pond (Pond 57) also exceeded the TEC value. Similar results were obtained for both ΣPAH_{HMW} and ΣPAH_{LMW}. For ΣPAH_{HMW}, all seven ponds mentioned above had at least one site with sediment levels which exceeded the PEC value (Fig. 26). One additional golf course pond (Pond 62) had sediment levels which also exceeded the TEC value. For ΣPAH_{LMW}, only three commercial ponds (Ponds 26, 83 and 87) had sediment levels which exceeded the PEC value, while Ponds 24, 38, 57 and 68 also exceeded the TEC value (Fig. 26). One reference site (Pond R2) also had ΣPAH_{LMW} levels in sediments which exceeded the TEC value.

For most individual PAH analytes, ecological benchmarks (TEC, PEC and Region IV SSV) and human health benchmarks (PRG values) are available (Figs. 27-32). In terms of the ecological benchmarks, several PAH analytes had an exceedance pattern similar to that described above for ΣPAH_{16}. Four commercial ponds (Ponds 26, 68, 83, and 87) and one low density residential pond (Pond 38) consistently had sediment levels of FLTH, PYR, B(a)P, D(a,h)A, and INPY in at least one sampling site which exceeded the PEC values, and sediment levels of ANTH, B(a)A, CHRY, and B(ghi)P which exceeded the TEC values. These same five ponds, plus one golf course pond (Pond 57) consistently had sediment levels which exceeded the Region IV SSV for PHEN, FLTH, PYR, B(a)A and B(a)P. Ponds 68 and 87 also had sediment levels which exceeded the Region IV SSV for ANTH and D(ah)A. For CHRY, the majority of ponds (10 out of 16) had sediment levels which exceeded the Region IV SSV value. For NAPT, the majority of stormwater ponds (11 out of the 16) had sediment levels which exceeded the TEC value in at least one site, but none of the ponds had sediment levels which exceeded the PEC value. Sediment levels in only one pond (Pond 48) exceeded the Region IV SSV for NAPT. For ACEN, only three ponds (Ponds 57, 48, and 87) had sediment levels which exceeded the Region IV SSV. No TEC or PEC values are available for ACNY. For FLUR, nearly all of the stormwater ponds surveyed (15 out of 16), including one reference ponds (Pond R2), had sediment levels that exceeded the TEC value, but none exceeded the PEC value. Sediment in only one pond (Pond 87) exceeded the Region IV SSV for FLUR.

In terms of human health benchmarks, only those individual PAH analytes that are known or suspected carcinogens (B(a)A, B(b)F, B(a)P, D(a,h)A, INPY) had sediment levels in stormwater ponds that exceeded the published PRG values (Figs. 27-32). Four commercial ponds (Ponds 26, 68, 83, and 87) and one low density residential pond (Pond 38) had sediment levels of B(a)A, B(b)F, D(a,h)A, and INPY which exceeded the PRG value. For B(a)P, the majority of stormwater ponds sampled (11 out of 16), including those ponds mentioned above and one reference pond (Pond R2), had sediment levels which exceeded the PRG value.

Metals

Ecological benchmarks (TEC, PEC and Region IV SSV) are available for Cr, Cd, Cu, Pb and Zn (Figs. 33-35). For Cr, only one residential pond (Pond 109) had sediment levels which exceeded the TEC and Region IV SSV values (Fig. 33). For Cd, all stormwater ponds and one reference pond (Pond R2) had sediment levels which exceeded both the TEC and Region IV SSV values (Fig. 33). None of the ponds had sediment levels which exceeded the PEC values for either Cr or Cd. For Cu, the majority of stormwater ponds had sediment levels which exceeded both the PEC (10 out of 16) and SSV values (12 out of 16) (Fig. 34). For Pb, none of the stormwater ponds had sediment levels which exceeded any of the ecological benchmarks (Fig. 34). For Zn, nearly all of the commercial ponds (Ponds 70, 48, 26, 68, and 87) had sediment levels which exceeded both the PEC and SSV values (Fig. 35). In terms of the human health benchmarks, none of the stormwater ponds had sediment levels which exceeded any of the PRG values.

Pesticides

For chloropyrifos, DDD, and DDE, Region IV SSV are available as ecological benchmarks and PRG values are available as human health benchmarks. Two golf course ponds (Ponds 80 and 57), two low density residential ponds (Pond 89 and 109), one residential high density pond (Pond 44), and three commercial ponds (Ponds 70, 48 and 68) had sediment levels of chloropyrifos that exceeded the Region IV SSV values (Fig. 36). None of the stormwater ponds had sediment levels of chloropyrifos that exceeded the PRG values. For both DDD and DDE, two commercial ponds (Ponds 68 and 83) and two reference ponds (Ponds R1 and R2) had sediment levels which exceeded both the Region IV SSV and PRG values (Fig. 36).

Discussion

The results of the current study suggest that stormwater detention ponds in coastal South Carolina are functioning as designed. These ponds are removing suspended sediments and contaminants from incoming stormwater, presumably enhancing the water quality of the runoff prior to reaching a natural receiving body. However, as a consequence, certain contaminants have become elevated in these pond sediments compared to reference ponds. Contaminant levels were dependent upon a variety of factors, chief among those were land use and drainage area. The highest overall contaminant levels were generally found in commercial ponds and those residential ponds with large drainage areas. Below we discuss each contaminant in more detail, emphasizing data trends, comparisons with other studies, and possible sources of contaminants.

PAHs

Commercial stormwater ponds had significantly higher levels of PAHs compared to ponds associated with other land uses. Sediment levels of ΣPAH_{16} in these commercial ponds ranged from 570 to 63,689 ng/g with an average level of 24,372 ng/g. These levels are generally similar to those reported in other studies involving commercial stormwater ponds. For example, in a study of a variety of stormwater detention ponds, wetlands and swales in Minneapolis, MN, sediment levels of total PAH (13 analytes) in commercial stormwater ponds ranged from 5,007 to 26,230 ng/g with an average level of 16,771 ng/g (Polta *et al.*, 2006). A commercial stormwater pond in Ontario, Canada and a highway stormwater pond in Surrey, UK also had total PAH levels (16,370 and 10,200 ng/g, respectively) similar to those reported in the current study. However, the levels reported here are generally an order of magnitude less than those reported for certain commercial ponds in Clearwater and Largo, FL, which had total PAH levels of 592,000 ng/g and 7,161,000 ng/g, respectively (Fernandez & Hutchinson, 1993). These high levels of PAHs are probably related to their age (20 and 30 years, respectively) and the fact that they had not had sediment removed as part as routine maintenance (Fernandez & Hutchinson, 1993).

Although interpretation of the PAH source ratios is somewhat confounded by the lack of alkyl-PAH data in the current study, the calculated ratios collectively suggest that the sources of PAH in the majority of stormwater ponds were predominantly pyrogenic (e.g., derived from combustion) in nature, although many ponds also had a distinct petrogenic (e.g., uncombusted fuel) signal. The ratio of FLTH/PYR has been used extensively to distinguish between petrogenic and pyrogenic PAH sources (reviewed by Neff *et al*., 2005). FLTH is less thermodynamically stable than PYR and a predominance of FLTH over PYR is characteristic of pyrogenic PAH sources. In petroleum-derived PAHs, PYR is more abundant compared to FLTH (Budzinski *et al*., 1997). During the combustion process, pyrogenic products are characterized by a FLTH/PYR ratios >1 (Baumard *et al*., 1999). In the current study, all 16 stormwater ponds had FLTH/PYR ratios >1, indicating predominantly pyrogenic sources for these two PAHs.

Another ratio used extensively to delineate petrogenic and pyrogenic sources of PAH contamination is PHEN/ANTH (Gschwend & Hites, 1981; Soclo *et al.*, 2000; Magi *et al.*, 2002). PHEN and ANTH are two structural isomers with different physico-chemical properties. PHEN is thermodynamically more stable than ANTH, and PHEN/ANTH ratios >25 have been attributed to the slow maturation of petroleum at low temperatures (Caredellicchio *et al.*, 2007). By contrast, pyrolysis of organic matter at high temperatures generates PAHs characterized by a low PHEN/ANTH ratio (<10). Ratios of PHEN/ANTH in the current study ranged widely from 5.61 to 86.72; however, more than half of the sampled stormwater ponds (9 of the 16 stormwater ponds) had PHEN/ANTH ratios >25, indicating petrogenic sources of PAH in these stormwater ponds. Most of these ponds with ratios >25 were either classified as residential or golf courses. These ratios suggest that uncombusted fuels, such as motor oil and transmission fluid, may be an important contributor to the overall PAH profile in these types of ponds. By contrast, only two ponds (Ponds 68 and 83), both classified as commercial, had a PHEN/ANTH ratio that clearly indicated pyrogenic sources for these two PAHs (ratio<10).

Petrogenic and pyrogenic sources of PAH have also been distinguished using a LMW/HMW ratio (Neff, 1979; Wise *et al.*, 1988). Ratios >1 usually indicate petrogenic sources, whereas ratios <1 indicate pyrogenic sources. In the current study, the majority of stormwater ponds (12 of 16 total) had LMW/HMW ratios that were <1, indicating predominantly pyrogenic sources of PAHs in these ponds. The only stormwater ponds that had ratios >1 were Ponds 31, 7, 80, and 48. When the ratios of LMW/HMW and FLTH/PYR were plotted against one another (Fig. 37), the majority of stormwater ponds had a clear pyrogenic fingerprint. Only Ponds 31, 7, 80, and 48 had signatures consistent with mixed pyrogenic and petrogenic PAH sources. The reference ponds, Pond R1 and R2, had clearly different PAH signatures compared to the stormwater ponds (Table 14 and Fig. 37). These unique signatures are a reflection of their overall low levels of PAH contamination. These PAHs are probably derived from long-range atmospheric deposition, rather than a specific pyrogenic or petrogenic source (c.f. De Luca *et al.*, 2005).

Metals

Although metal levels in sediments are influenced by natural background levels, and these background levels can vary from one region to another, the levels of metals reported in the current study were generally similar to that reported elsewhere (Table 20). Commercial stormwater ponds in the current study had significantly elevated sediment levels of Pb and Zn compared to the other classes of stormwater ponds and reference ponds. And, although there were no significant differences in Cu levels among the land use classes (p=0.0870), there was a trend towards higher Cu levels in the sediments of commercial, low density residential and golf course ponds. All three of these metals (Cu, Pb and Zn) are commonly enriched in sediments due to anthropogenic sources, particularly vehicular activity (Sutherland & Toloso, 2001; Turer & Maynard 2003). Specific vehicular sources of these metals include the combustion of fuel (Monaci *et al.*, 2000), and the wearing of tires and breaks (Councell *et al.*, 2004). Providing support for the notion that vehicular activity may be the primary source of metals in these stormwater detention ponds is the fact that Cr, which was not found at elevated levels in the current study, is not generally associated with vehicular activity (Turer & Maynard, 2003). Additional sources of Zn and Pb in the sediments of these stormwater ponds may be related to the wearing of roofing materials (Chang *et al.*, 2004). Van Metre & Mahler (2003) found that particles washed from rooftops contributed 20 and 18 percent, respectively, of the total Zn and Pb in an urban watershed (Austin, TX). An additional source of Cu in these stormwater pond sediments may be related to the use of copper-based algaecides. In fact, based upon the results of the maintenance survey (Table 8), those stormwater ponds previously treated with copper-based algaecides (127.8±41.0 mg/kg) had significantly higher sediment levels of Cu than those ponds which have not been previously treated (47.6±16.7 mg/kg) (t-test, p=0.0387). This may also explain the reason that low density residential and golf course ponds, in addition to the commercial ponds, had marginally elevated sediment levels of Cu compared to that of the reference ponds. Although several previous studies have also found elevated sediment levels of either Cd and/or Cr in stormwater detention pond sediments (Fernandez & Hutchinson, 1993; Marsalek & Marselek 1997, Parker *et al*., 2000), their levels were not significantly elevated in the various land use classes in the current study. Not surprisingly, sediment levels of Al and Fe were found to be similar among all stormwater pond classes in the current study. These metals are primarily derived from natural sources (Bruland *et al*., 1974; Goldberg *et al.*, 1977), and have generally not been found at elevated levels in stormwater pond sediments.

The distribution of metals in stormwater pond sediments was complex and probably related to sediment particle size. Sediment levels of Cu, Pb and Zn were significantly higher in the pond center compared to the pond inlet. Metals, in general, associate with fine-grained sediments, such as clays. And, in the current study, clay content was significantly higher in the pond center compared to the pond inlet. By contrast, pond inlets had a significantly higher sand content compared to the pond center. This higher sand content is probably the result of the deposition of larger, suspended materials from incoming stormwater near the inlet, and perhaps to a lesser extent, by erosion of the sides of the pond. The high sand content in this depositional material is probably related to the naturally high sand content of soils along the South Carolina coast. Similar observations have been made in Ontario (Canada), where sediment levels of Cu, Pb, and Zn have been found to increase from the sand-rich sediments of a stormwater pond inlet to the clay-rich sediments of the pond center (Marselek & Marselek, 1997). In the current study, the fact that Al and Cd levels were significantly higher in sediments of the pond inlets compared to that of the pond centers suggests that these metals may be less mobile in these coastal stormwater detention ponds than Cu, Pb or Zn.

Pesticides/PBDEs

Results of the current study demonstrate that both current use and historical pesticides are present in the sediments of stormwater ponds in coastal South Carolina; however, their levels are not dependent upon land use. The most frequently detected pesticide was the organophosphate chlorpyrifos, which was found in 9 of the 16 stormwater ponds. Chlorpyrifos is widely used as an insecticide in agricultural and in golf course settings. Prior to 2001, it was also widely used in residential and commercial settings for the control of termites, fire ants, mosquitoes, and a variety of indoor insects. However, due to concern over exposure to children, the US EPA cancelled many of the indoor and outdoor residential uses for chlorpyrifos in 2001 (US EPA, 2002). The half-life of chlorpyrifos in water has been estimated to be between 1 and 6 days, but when sediment-bound, its half life is considerably longer. In fact, in an outdoor pond mesocosm study, no appreciable loss of sediment-bound chlorpyrifos occurred during the 84-day study (Giddings *et al.*, 1997). Another study investigating urban streams in San Diego has found that the persistence of sediment-bound chlorpyrifos changes dramatically under anaerobic conditions, with half lives increasing from 20.3 days under aerobic conditions to 223 days under anaerobic conditions (Bonderancko & Gan, 2004). The fact that residential and commercial stormwater detention ponds continue to act as repositories of chlorpyrifos suggests that either the degradation of this compound in coastal stormwater pond sediments is very slow and/or property owners are continuing to use their existing supplies of chlorpyrifos for lawn and garden uses.

Several coastal stormwater ponds are also serving as repositories for organochloride pesticides. Ponds 31, 68, and 83 had detectable levels of either chlordane or the breakdown products of DDT (DDD and DDE). Since DDT and chlordane were banned by the US EPA in 1972 and 1988, respectively, these pesticide residues in stormwater pond sediments probably represent the legacy of historical agricultural uses of these products. Their presence in these pond sediments can be attributed to (1) most suburban development along the South Carolina coast occurs on former cultivated agricultural land (Alan and Lu, 2000), (2) stormwater ponds are designed to collect and concentrate suspended sediments in runoff and

(3) these compounds have very long half-lives when sediment-bound (some estimates >15 years), especially under anaerobic conditions (reviewed by US Dept. of Health and Human Services, 2002). Similar observations have been made in metropolitan Phoenix, where development on former cultivated agricultural land has resulted in residential stormwater ponds with elevated sediment levels of DDT, DDE, chlordane, dieldrin, and toxaphene (Parker *et al*., 2000). The presence of DDD and DDE in the two reference ponds, Ponds R1 and R2, are probably also related to the historical use of DDT. Pond R1, located in the Francis Marion National Forest near Awendaw, SC, is a railroad borrow pit pond surrounded by long leaf pine forest. The presence of DDT breakdown products in these sediments may be the result of treatment of this forest for Southern Pine Beetles. By contrast, Pond R2, located on the Dill Plantation on James Island, SC, is a former plantation irrigation pond. In this pond, the presence of the breakdown products of DDT is probably the legacy of former agricultural uses of the pesticide on this plantation.

Despite the widespread contamination of PBDEs in a variety of aquatic and terrestrial environments (reviewed by Rahman *et al.*, 2001), the results of the current study suggest that the levels of PBDEs in sediments of coastal stormwater ponds are generally below detection limits. PBDEs have been widely used as flame retardants in a variety of common household and workplace products, and as a result, have become a worldwide pollution problem reaching even remote areas. PBDEs are of environmental concern because (1) they are highly lipophilic, and as such, are expected to accumulate in fatty tissues and sediments (Rahman *et al*., 2000), (2) they are highly resistant to degradation processes with half-lives in sediment estimated to be on the order of months to years (Gouin & Harner, 2003), and (3) concentrations of PBDEs in the environment and people double every 4-6 years in North America (Hites, 2004). The only PBDE congeners found in the present study were penta-PBDEs, which were found in the inlet and center of Pond 68. Penta-PBDEs are the predominant congeners found in several technical flame-retardant mixtures, including DE-71 and Bromkal 70-5 DE (LaGuardia *et al.*, 2006), and their presence can also be indicative of the breakdown of larger PBDE congeners, including the deca-PBDEs (Rahman *et al.*, 2001). It is interesting to note that Pond 68 was the only pond in the current study to receive runoff from both commercial and high density residential areas, and the penta-PBDE congeners found in the sediments of this pond may have come from either or both areas.

Fecal Coliform

Fecal coliforms, like other contaminants, have a preference for adsorbing to the fine particle size fraction of suspended material (< 2 μm in size) in stormwater runoff (Davies & Bavor, 2000). Since detention ponds are designed to provide conditions conducive to the settlement of this material, it might be expected that there would be an accumulation of these particle-associated contaminants in stormwater sediments through time. As was the case with PAH, certain metals, and some pesticides, the results of the current study suggest that fecal coliforms are accumulating in the sediments of some stormwater ponds. Underscoring the fact that sediment fecal coliform are physically behaving like many chemical contaminants in stormwater runoff are the positive correlations that were observed between sediment fecal coliform levels and percent clay, ΣPAHs, Al, Cd, Fe, Pb and Zn (Table 10). However, unlike ΣPAH, Cu, Pb and Zn, there were no differences in sediment fecal coliform levels among the different land uses. Mean levels of fecal coliforms in the stormwater detention ponds sampled in the current study (1,079±242 MPN/g) are on the same order of magnitude as contaminated river sediments (Grimes 1980, Crabill *et al.*, 1999), but are several orders of magnitude lower than that reported for marina sediments and sewage sludge (Edmonds, 1976; Bujoczek *et al.*, 2001; An *et al.*, 2002) (Table 21).

It is also important to note that levels of fecal coliforms in these pond sediments are unlikely to significantly decrease in the short term. In a pond mesocosm study, Davies *et al.* (2003) reported that fecal coliform levels in sediments collected from stormwater detention ponds decreased less than one order of magnitude in 28 days. In a wetland study, Stenström & Carlander (2001) reported T_{90} (time taken for the population to decline by 90% of initial concentration) values of 370 days for sediment fecal coliforms. Given the likely long residence times of fecal coliforms in aquatic sediments, these coastal stormwater ponds are probably serving as a reservoir for indicator and disease causing enteric bacteria. And, following sediment resuspension (e.g., during a heavy rainfall), these ponds have the potential to serve as a source of enteric bacteria to natural receiving bodies of water. Since many of these natural water bodies also serve a role in recreation (e.g., fishing, swimming, etc.), the enteric bacteria from these ponds may pose a risk to public health.

Determinants of pond sediment contamination

Although land use classification was qualitatively an important factor in determining the overall level of sediment contamination in these stormwater ponds, other variables were also useful in quantitatively explaining levels of sediment contamination. One such variable was pond drainage area, which was able to explain 72.5, 61.1, and 59.0% of the variability in the ΣPAH_{16}, Pb, and Zn data. In suburban areas, these three common contaminants were associated with vehicular usage, particularly vehicular miles traveled (Page & Ganje, 1970; Van Metre *et al.*, 2000; Councell et al., 2004). It stands to reason that for suburban residential and commercial ponds, the larger the drainage area for a stormwater pond, the more vehicular usage there would be. Golf course ponds are fundamentally different than residential and commercial ponds in that increases in drainage area may not necessarily equate to an increase in vehicular usage. In the current study, golf course ponds did not have significantly elevated levels of these vehicular-related contaminants. In addition to drainage area, we also investigated the relationship between contaminant levels and percent impervious cover in the watershed. In the current study, by itself, percent impervious cover was not an important determinant of overall contamination levels. This is in contrast to other studies that have found percent impervious cover to be important determinant in contaminant loadings in a variety of aquatic habitats, including streams and tidal creeks (Arnold & Gibbons, 1996; Paul & Meyer, 2001; Holland *et al.*, 2004). The lack of a relationship between sediment contamination and percent impervious cover in the residential and commercial ponds may be related to the relatively large range of drainage areas (i.e., 3.14-19.7 ha) in combination with the relatively narrow range of percent impervious cover (i.e., 27.3-77.0%) examined in the current study.

Pond surface area was a minor determinant in the explaining sediment levels of PAH, Cu and Zn, and the major determinant for sediment levels of chlorpyrifos In addition, levels of clay, PAH, Pb, Zn and fecal coliforms were all positively correlated to pond surface area. These correlations suggest that ponds with larger surface areas, and presumably larger volumes, are more effective in trapping fine particles and the chemical contaminants associated with them, compared to those stormwater ponds with smaller surface areas.

Previous studies have indicated that pond age, in the absence of maintenance sediment removal, is an important determinant of sediment levels for some contaminants. For example, PAH sediment levels in Florida vary from near background levels in stormwater ponds that were <1 year old to >7,000,000 ng/g in 30 year old ponds (Fernandez & Hutchinson, 1993). Evidence for increasing sediment levels of metals with age is less clear, and is probably related to several factors including land use and sediment particle size distributions. In suburban Maryland, Casey *et al.* (2006) found that Cu and Pb levels remain fairly constant over a 10 year period, but levels of Zn increased 10X over the same 10 years. In Pensacola, FL, Liebens (2002) conducted a survey of 24 residential and commercial ponds and found a significant relationship between heavy metal contamination and age. In the current study, pond age was not an important determinant of overall contaminant levels. The lack of a significant relationship between contaminant levels and age may be related to the relatively small range of pond ages (4 to 12 years old) used in the current study. Certainly, further investigation into the influence of pond age on sediment contaminant levels in stormwater ponds of coastal South Carolina is warranted.

Principle component analysis delineated two distinct groups of stormwater ponds based upon their correlation matrices. One group consisted of stormwater ponds with highly contaminated sediments. This group included four commercial ponds (Ponds 87, 83, 26 and 68) and one residential pond (Pond 38) with a large drainage area. The fact that these five stormwater ponds had similar contaminant profiles is probably the result of similar contaminant sources. Evidence presented in this study suggests that vehicular use is the predominant source of contamination in these ponds, especially with regard to sources of PAH, Cu, Pb, and Zn. A second group of stormwater ponds had relatively low levels of contamination. This group included most residential and golf course ponds. In general, levels of contaminants in these ponds were not significantly different than that of the reference ponds. The two reference ponds (Pond R1 and R2) had unique contaminant profiles and did not cluster with either each other or the other two groups of stormwater ponds. Contaminant profiles in these two ponds represent background profiles, and their most likely source is atmospheric deposition for the PAHs and historical land use for the pesticides. Differences in their respective profiles are probably related to their location, particularly their proximity to nearby contaminant sources. Pond R1 is located in a rural area of the Francis Marion National Forest, located ~15 km from Mount Pleasant and ~60 km from Georgetown. By contrast, Pond R2 is located on suburban James Island, and is only ~5 km from downtown Charleston, SC.

Toxicity

One of the major unresolved issues in the literature related to stormwater ponds is whether or not the sediments pose a hazard to native wildlife. The results of the present study indicate that there was little evidence of acute toxicity of the stormwater pond sediments to the amphipod *H. azteca* in short-term (10 day) exposures. This was despite the fact that numerous commercial ponds exceeded the TEC and PEC values for PAHs and Zn. In fact, the only land use class with significantly elevated mortality compared to that of the reference ponds was the golf course ponds, which had comparatively few TEC and PEC exceedances. Although the mortality rates observed in the sediments collected from golf course ponds would generally be considered low (replicate mortality ranged from 2.7 to 16.7%), the fact that it was significantly higher than that observed in sediments from the reference ponds suggests that the use of some unmeasured compound related to golf course maintenance (e.g. novel pesticides, fertilizers, turf dyes, etc.) may be contributing to the mortality observed in these laboratory tests.

The general lack of acute toxicity in the commercial pond sediments suggests that these contaminants were either not bioavailable for uptake, or they were bioavailable but to an extent less than that required to elicit acute toxicity in amphipods. In a survey of 20 stormwater ponds in suburban Maryland, Karouna-Renier & Sparling (2001) found that both Cu and Zn are bioavailable to macroinvertebrates, and they do accumulate higher levels of these two metals in stormwater detention ponds compared to that of their counterparts in ponds not receiving stormwater. However, similar to our findings, they found little mortality (ranged between 4.3 to 13.3%) of these stormwater pond sediments to *H. azteca* in the laboratory (Karouna-Renier & Sparling 1997). By contrast, other studies have documented acute toxicity resulting from exposure to stormwater pond sediments. In a survey of stormwater ponds in Phoenix, Parker *et al.* (2000) found mean mortalities of *H. azteca* between 29 to 82%, and this toxicity was apparently seasonally-dependent based upon behavioral interactions between *Hyalella* and the sediments.

These conflicting findings in the literature underscore the fact that toxicity is dependent upon a variety of factors, including sediment characteristics and complex biological and chemical interactions, which are not completely understood. As such, environmental management decisions need to be based upon multiple toxicity assays at a site using a variety of species, endpoints, and weather conditions. The current study did not specifically address (1) bioavailability, (2) chronic toxicity of the sediment (e.g. growth, reproduction, etc.), (3) toxicity of these sediments to other taxa, (4) seasonal variation in toxicity, or (5) toxicity of the sediments following rain events. All of these issues need to be addressed in order to make a scientifically sound management decision concerning the overall toxicological hazard of these pond sediments to native wildlife.

Screening Assessments

Screening assessments are typically the first step in an ecological risk assessment at a hazardous waste site, and they are designed to identify contaminants of potential toxicological concern (reviewed by Jones *et al*., 1997). Screening is typically performed using a set of several toxicological benchmarks. These benchmarks are useful in determining whether contaminants warrant further investigation or are at a level that requires no further attention. If a chemical concentration exceeds a lower benchmark, further investigation is needed to determine the hazards posed by the presence of that chemical. However, if the chemical concentration falls below the lower benchmark value, then the chemical is generally regarded as not posing a hazard. Chemical concentrations exceeding an upper screening benchmark indicate that the chemical is clearly of concern and that remedial actions are likely to be needed. The use of multiple benchmarks can indicate the likelihood and nature of effects. For example, exceedance of only one benchmark may provide weak evidence of real effects, whereas exceedance of multiple benchmarks of provides strong evidence of real effects.

Based upon the results of the screening assessment performed here, it is obvious that PAHs are the contaminant of greatest toxicological concern in the sediments of stormwater detention ponds in coastal South Carolina. Sediment from most commercial ponds (Ponds 26, 68, 83 and 87) and one low density residential pond (Pond 38) consistently exceeded multiple benchmarks for both ecological and human health. These five ponds had at least one site which exceeded 100% of the TEC values for individual analytes (Fig. 38). These same five ponds exceeded PEC values for between 36.4 and 90.9% of the PAH analytes (Fig. 39) and exceeded the EPA Region IV SSV values for between 50 and 75% of the PAH analytes (Fig. 40). This strongly suggests that toxicological effects due to PAH exposure are likely occurring to wildlife *in situ*. In light of the high survivorship in the sediment toxicity tests with *H. azteca*, any on-going toxicological effects may be more chronic in nature rather than acute. Results from the screening level assessment also suggest that the sediments from these five ponds would also likely pose a hazard to human health if they were to be removed and used as fill in residential areas. These sediments exceeded the published PRG values for human health risks from between 33.3 to 41.7% of the PAH analytes. Many of the exceeded PRG values, including B(b)F, B(a)P, CHYR, D(ah)A and INPY, are known or suspected carcinogens.

Despite evidence of the anthropogenic buildup of some metals in certain stormwater pond sediments, results of the screening assessment suggest they are generally less of a toxicological concern than the PAHs (Figs. 42-45). One metal of ecological concern was Cu, which exceeded the published PEC values in 11 of the 16 stormwater ponds and the EPA Region IV SSV values in 15 of the 16 ponds. Unlike the trends observed for the PAHs, all land use classes, including golf courses, contained ponds with sediments exceeding these benchmark values. Interestingly, 6 of the 11 ponds exceeding the PEC values have been identified as having used copper-based products to control algae. Of lesser ecological concern was Zn and Cd. Sediments from most of the commercial ponds exceeded the published TEC and SSV values for Zn; however, none of the ponds had sediments that exceeded the PEC value for Zn. Sediment from nearly all stormwater ponds sampled exceeded the TEC and SSV values for Cd, but again, none of the sediments had levels which exceeded the PEC value for Cd. The fact that the two reference ponds (Pond R1 and R2) also had levels of Cd similar to that of the stormwater ponds suggests that the sandy soils of coastal South Carolina may be naturally-enriched in Cd. None of the sediments exceeded any of the published PRG values for metals, suggesting that these levels of metals would not, by themselves, pose a hazard to human health when removed.

Results of the screening assessment also determined that the sediment levels of a few pesticides and their breakdown products, particularly chlorpyrifos, DDD and DDE, are of both ecological and human health concern in certain stormwater ponds. The SSV value for chlorpyrifos was exceeded in 9 of the 16 ponds; however, these exceedances were independent of land use. Given that (1) most residential and commercial uses for this compound were restricted by the US EPA in 2001 and (2) it has a relatively short half life, it is likely that sediment levels in these ponds will decrease in the future. For DDD and DDE, both the SSV and PRG values were exceeded by two of the commercial ponds and both reference ponds. The degree to which these organochloride compounds represent an environmental concern is difficult to discern. These compounds have considerably long half lives, but are generally tightly bound to sediments resulting in an inherently low mobility *in situ*. In fact, they are really only bioavailable when ingested by sediment-feeding organisms (Luoma, 1989). When ingested, bioaccumulation of these compounds to higher trophic levels may occur as these sediment-feeding organisms are preyed upon. However, when these sediments are disturbed (as is the case during sediment removal), they become increasingly mobile. As the mobility of these compounds increase, the likelihood of exposure of these compounds to wildlife and humans also increases. Consequently, these contaminated sediments may be hazardous following removal. For example, once these sediments have been dried following removal, they may be inhaled as airborne dust by both wildlife and humans.

Summary

The biological and chemical contamination of stormwater pond sediments along the South Carolina coast can be summarized as follows: (1) commercial pond sediments had overall levels of PAHs, Pb, and Zn that are elevated relative to that of various other land use classes, (2) residential and golf course pond sediments generally had levels of contaminants that were similar to those of reference ponds, (3) PAH levels were largely independent of sampling location within a pond; however, several metals, including Cu, Pb and Zn had significantly higher levels in pond centers reflecting their high affinity towards clay-rich sediments, (4) a few pesticides and pesticide breakdown products, including chlorpyrifos, DDD and DDE, are elevated in some stormwater ponds; however, their presence was independent of land use and sampling location, (5) fecal coliform levels in stormwater pond sediments were generally similar to reference pond sediments, although two ponds had significantly elevated levels, (6) contamination of pond sediments by PAHs, Cu, Pb and Zn was largely dependent upon drainage area, and to a lesser extent upon pond surface area, (7) the predominant source of contamination in these ponds appears to be related to vehicle usage, (8) based upon screening level assessments, PAHs in stormwater pond sediments are the contaminant of greatest concern, and (9) several commercial ponds, and one residential pond with a relatively large watershed, contained sediment levels of several contaminants which exceeded toxicological benchmarks for both ecological and human health, although laboratory-based toxicity tests demonstrated that these sediments were generally not acutely toxic to amphipods.

Recommendations/Future Needs

In coastal South Carolina, the majority of stormwater ponds have been constructed in the past 15 years likely due to regulations established in 1991. If homeowner's associations, property management firms and local governments are following recommendations for sediment removal every 10 years, then the overwhelming majority of existing stormwater ponds have not had accumulated sediment removed. Given their age and history, it is quite likely that many of the existing ponds will need to have sediment removed in the next 5-10 years to maintain efficiency. More likely, however, the need for sediment removal will only become apparent when the stormwater pond becomes very shallow and performance has already significantly degraded. Either way, the inventory of stormwater pond sediment to be removed in the future is likely very high. Based upon our findings, we make the following recommendations:

1. Sediment removed from commercial ponds, as well as residential ponds with large drainage basins, pose a potential human health hazard if used as fill material in residential and commercial settings. Therefore, these types of ponds should be tested for sediment contaminants prior to removal to determine the appropriate method for sediment disposal.

2. Sediments in commercial ponds, as well as residential ponds with large drainage basins, also pose a potential ecological hazard *in situ*. During sediment removal, stormwater ponds should be cut off from their receiving body in order to protect downstream biota from exposure to resuspended, contaminated materials.

3. Enforcement of maintaining pond efficiencies, including periodic sediment removal, may be necessary to prevent stormwater ponds from achieving those contaminants levels observed in older Florida ponds.

4. Further research is required in the following five areas:

    a. Further characterization of the toxicity of stormwater pond sediments. Toxicity tests should include both acute and chronic endpoints involving several different taxa.
    b. Influence of age on overall pond contaminant levels. Does a threshold age exist at which pond sediments are transformed from having contaminant levels not significantly different than background to highly contaminated?
    c. Influence of drainage area on overall pond contaminant levels in residential settings. Again, does a threshold drainage area exist at which pond sediments prematurely become highly contaminated?
    d. Monitoring contaminant levels in the sediments of one pond (or a few ponds) over time to determine the rates of decay of some contaminants (e.g., chlorpyrifos) and the rates of accumulation of other contaminants (e.g., PAHs).
    e. Characterization of native wildlife usage in these stormwater ponds. Over the past 10-15 years, thousands of hectares of freshwater habitat have been created in coastal South Carolina. This significant change in the ecological landscape of the region may have important consequences for bird and mammal populations.
    f. Characterization of harmful algal bloom cysts in stormwater pond sediments. These cysts may pose additional risks to both biota *in situ* and to human health if the sediments are used as fill.

References

Alan JS, Lu KS. 2000. Cultivated Agricultural Lands at Risk from Potential Urbanization in the Tri-County Area. South Carolina Water Resources Center and Strom Thurmond Institute, Clemson University.

Alm EW, Burke J, Spain A. 2002. Fecal indicator bacteria are abundant in wet sand at freshwater beaches. Water Research 37:3978-3982.

American Public Health Association (APHA). 1989. Standard Methods for the Examination of Water and Wastewater, 17th ed. American Public Health Association, Water Works Association and Water Environmental Federation. Washington, DC.

An Y-J, Kampbell DH, Breidenbach GP. 2002. *Escherichia coli* and total coliforms in water and sediments at lake marinas. *Environmental Pollution* 120:771-778.

Arnold CL, Gibbons CJ. 1996. Impervious surface coverage: The emergence of a key environmental indicator. *Journal of the American Planning Association* 62: 243-258.

ASTM. 2005. Standard Test Method for Measuring the Toxicity of Sediment-Associated Contaminants with Freshwater Invertebrates. E1706-05. ASTM Annual Book of Standards, Volume 11.05. ASTM, West Conshohocken, PA.

Atlanta Regional Commission. 2001. Georgia Stormwater Management Manual, Volume 2: Technical Handbook. www.georgiastormwater.com.

Baumard P, Budzinski H, Guarrigues P, Dizer H, Hansen PD. 1999. Polycyclic aromatic hydrocarbons in recent sediments and mussels (*Mytilus edulis*) from the Western Baltic Sea: Occurrence, bioavailability and seasonal variations. *Marine and Environmental Research* 47:17–47.

Barraud S, Dechesne M, Bardin J-P, Varnier J-C. 2005. Statistical analysis of pollution in stormwater infiltration ponds. *Water Science and Technology* 51:1-9.

Bonderenko S, Gan J. 2004. Degradation and sorption of selected organophosphate and carbamate insecticides in urban stream sediments. *Environmental Toxicology and Chemistry* 23:1809-1814.

Brown JN, Peake BM. 2006. Sources of heavy metals and polycyclic aromatic hydrocarbons in urban stormwater runoff. *Science of the Total Environment* 359:145-155.

Bruland KW, Bertine K, Koide M, Goldberg ED. 1974. History of heavy metal pollution in Southern California coastal zone. *Environmental Science and Technology* 8:425-432.

Budzinski H, Jones I, Bellocq J, Pierard C, Garrigues P. 1997. Evaluation of sediment contamination by polycyclic aromatic hydrocarbons in The Gironde Estuary. *Marine Chemistry* 58:85–97.

Bujoczek G, Reiners RS, & Olaszkiewicz JA. 2001. Abiotic factors affecting inactivation of pathogens in sludge. *Water Science and Technology* 44:79-84.

Cardellicchio N, Buccolieri A, Giandomenico S, Lopez L, Pizzulli F, Spada L. 2007. Organic pollutants (PAHs, PCBs) in sediments from the Mar Piccoloin Taranto (Ionian Sea, Southern Italy). *Marine Pollution Bulletin* 55:451-458.

Casey RE, Simon JA, Atueyi S, Snodgrass JW, Karouna-Renier N, Sparling DW. 2006. Temporal trends of trace metals in sediment and invertebrates from stormwater management ponds. *Water, Air, and Soil Pollution* 178:69-77.

Chang M, McBroom MW, Beasley RS. 2004. Roofing as a source of nonpoint water pollution. *Journal of Environmental Management* 73:307-315.

Councell TB, Duckenfield KU, Landa ER, Callender, E. 2004. Tire-wear particles as a source of zinc to the environment. *Environmental Science & Technology* 38:4206-4214.

Crabill C, Donald R, Snelling J, Foust R, Southam G. 1999. The impact of sediment fecal coliform reservoirs on seasonal water quality in Oak Creek, Arizona. *Water Research* 33:2163-2171.

Davies CM, Bavor HJ. 2000. The fate of stormwater-associated bacteria in constructed wetland and water pollution control pond systems. *Journal of Applied Microbiology* 89:349–360.

Davies CM, Yousefi Z, Bavor HJ. 2003. Occurrence of coliphages in urban stormwater and their fate in stormwater management systems. *Letters in Applied Microbiology* 37:299-303.

Dechcesne M, Barraud S, Bardin J-P. 2004. Spatial distribution of pollution in an urban stormwater infiltration basin. *Journal of Contaminant Hydrology* 72:189-205.

De Luca G, Furesi A., Leardi R, Micera G, Panzanelli A., Piu PC, Pilo MI, Spano N, Sanna G. 2005. Nature, distribution and origin of polycyclic aromatic hydrocarbons (PAHs) in the sediments of Olbia harbor (Northern Sardinia, Italy). *Marine Pollution Bulletin* 50:1223–1232.

Edmonds RL. 1976. Survival of coliform bacteria in sewage sludge applied to a forest clearcut and potential movement into groundwater. *Applied and Environmental Microbiology* 32: 537-546.

Fan W, Wang W. 2003. Extraction of spiked metals from contaminated coastal sediments: A comparison of different methods. *Environmental Toxicology and Chemistry* 22:2659-2666.

Fernandez M, Hutchinson CB. 1993. Hydrogeology and chemical quality of water and bottom sediment at three stormwater detention ponds, Pinellas County, FL. USGS Water Investigations Report 92-4139.

Flemming AT, Weinstein JE, Lewitus AJ. 2008. Survey of PAH in low density residential stormwater ponds in coastal South Carolina: False dark mussels (*Mytilopsis leukophaeta*) as potential biomonitors. *Marine Pollution Bulletin* in press.

Giddings JM, Biever RC, Racke KD. 1997. Fate of chlorpyrifos in outdoor pond microcosms and effects on growth and survival of bluegill sunfish. *Environmental Toxicology and Chemistry* 16:2353-2362.

Goldberg ED, Gamble E. Griffin JJ, Koide M. 1977. Pollution history of Narragansett Bay as recorded in its sediments. *Estuarine and Coastal Marine Science* 5:549-561.

Gouin T, Harner T. 2003. Modeling the environmental fate of the polybrominated diphenyl ethers. *Environmental International* 29:717-724.

Graney JR, Eriksen TM. 2004. Metals in pond sediments as archives of anthropogenic activities: a study in response to health concerns. *Applied Geochemistry* 19:1177-1188.

Grimes DJ. 1980. Bacteriological water quality effects of hydraulically dredging contaminated Upper Mississippi River bottom sediment. *Applied and Environmental Microbiology* 39:782-789.

Gschweng PM, Hites RA. 1981. Fluxes of polycyclic aromatic hydrocarbons to marine and lacustrine sediments in the Northeastern United States. *Geochimica et Cosmochimica Acta* 45:2359–2367.

Hites RA. 2004. Polybrominated diphenyl ethers in the environment and in people: A meta-analysis of concentrations. *Environmental Science & Technology* 38:945-956.

Holland AF, Sanger DM, Gawle CP, Lerberg SB, Santiago MS, Riekerk GHM, Zimmerman LE, Scott GI. 2004. Linkages between tidal creek ecosystems and the landscape and demographic attributes of their watersheds. *Journal of Experimental Marine Biology and Ecology* 298:151-178.

Jeng HAC, Englande AJ, Bakeer RM, Bradford HB. 2005. Impact of urban stormwater runoff on estuarine environmental quality. *Estuarine Coastal and Shelf Science* 63:513-526.

Jones DS, Suter GW, Hull RN. 1997. Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-associated Biota: 1997 revision. East Tennessee Technology Park, Technical Information Park, ES/ER/TM-95/R4.

Kamalakkannan R, Zettel V, Goubatchev A, Stead-Dexter K, Ward N. 2004. Chemical (polycyclic aromatic hydrocarbon and heavy metal) levels in contaminated stormwater and sediments from a motorway dry detention pond drainage system. *Journal of Environmental Monitoring* 6:175-181.

Karim MR, Manshadi FD, Karpiscak MM, Gerba CP. 2004. The persistence and removal of enteric pathogens in constructed wetlands. *Water Research* 38:1831-1837.

Karouna-Renier NK, Sparling DW. 1997. Toxicity of stormwater treatment pond sediments to *Hyalella azteca* (Amphipoda). *Bulletin of Environmental Contamination & Toxicology* 58:550-557.

Karouna-Renier NK, Sparling DW. 2001. Relationships between ambient geochemistry, watershed land-use and trace metal concentrations in aquatic invertebrates living in stormwater treatment ponds. *Environmental Pollution* 112:183-192.

Kayali-Sayadi MN, Rubio-Barroso S, García-Iranzo R, Polo-Díez LM, 2000. Determination of selected polycyclic aromatic hydrocarbons in toasted bread by supercritical fluid extraction and HPLC with fluorimetric detection. *Journal of Liquid Chromatography & Related Technologies* 23, 1913-1925.

LaGuardia MJ, Hale RC, Harvey E. 2006. Detailed polybrominated diphenyl ether (PBDE) congener composition of the widely used penta-, octa-, and deca-PBDE technical flame-retardant mixtures. *Environmental Science & Technology* 40:6247-6254.

Liebens J. 2002. Heavy metal contamination of sediments in stormwater management systems: The effect of land use, particle size and age. *Environmental Geology* 41:341-351.

Long ER, MacDonald DM, Smith SL, Calder FD. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environmental Management* 19:81-97.

Luoma SN. 1989. Can we determine the biological availability of sediment-bound trace elements? *Hydrobiologia* 176/177:379-396.

Magi E, Bianco R, Ianni C, Carro MD. 2002. Distribution of polycyclic aromatic hydrocarbons in the sediment of the Adriatic Sea. *Environmental Pollution* 119:91–98.

Marsalek J, Marsalek PM. 1997. Characteristics of sediments from a stormwater management pond. *Water Science & Technology* 36:117-122.

Marsalek J, Rochfort Q, Grapentine L, Brownlee B. 2002. Assessment of stormwater impacts on an urban stream with a detention pond. *Water Science & Technology* 45:255-263.

Monaci F, Moni F, Lanciotti E, Grechi D, Bargagli R. 2000. Biomonitoring of airborne metals in urban environments: New tracers of vehicle emission, in place of lead. *Environmental Pollution* 107: 321-327.

Neff JM. 1979. Polycyclic Aromatic Hydrocarbons in the Aquatic Environment. Sources, Fates and Biological effects. Applied Science, London, Great Britain, p. 262.

Neff JM, Scott SA, Gunster DG. 2005. Ecological risk assessment of polycyclic aromatic hydrocarbons in sediments: identifying sources and ecological hazard. *Integrated Environmental Assessment and Management* 1:22-33.

Page AL, Ganje TJ. 1970. Accumulations of lead in soils for regions of high and low motor vehicle traffic density. *Environmental Science & Technology* 4:140-142.

Parker JTC, Fossum KD, Ingersoll TL. 2000. Chemical characteristics of urban stormwater sediments and implications for environmental management, Maricopa County, Arizona. *Environmental Management* 26:99-115.

Paul MJ, Meyer JL. 2001. Streams in the urban landscape. *Annual Review of Ecology and Systematics* 32:333-365.

Polta R, Balogh S, Craft-Reardon A. 2006. Characterization of Stormwater Pond Sediments: Final Project Report. Metropolitan Council Environmental Services, Environmental Quality Assurance Department.

Plumb RH. 1981. Procedures for handling and chemical analysis of sediment and water samples. Environmental Laboratory, U.S. Army Waterways Experiment Station, Vicksburg, MS.

Rahman F, Langford KH, Scrimshaw MD, Lester JN. 2001. Polybrominated diphenyl ether (PBDE) flame retardants. *Science of the Total Environment* 275:1-17.

Reynolds W, McLeod A, Hajjar L, Rodgers T. 2005. A Citizen's Guide to Stormwater Pond Maintenance in South Carolina. South Carolina Department of Health and Environmental Control, Columbia, SC. CR-003069.

SC DHEC. 2003. South Carolina Stormwater Management and Sediment Control Handbook for Land Disturbing Activities. South Carolina Department of Health and Environmental Control, Columbia, SC.

SCDHEC. 2005. Stormwater management BMP handbook. SCDHEC, Bureau of Water.

SCDHEC. 2006. NPDES general permit for stormwater discharges from large and small construction activities. SCDHEC, Bureau of Water

Siewicki TC, Pullaro T, Pan W, McDaniel S, Glenn R, Stewart J. 2007. Models of total and presumed wildlife sources of fecal coliform bacteria in coastal ponds. *Journal of Environmental Management* 82: 120-132.

Smucker SJ. 1998. Region 9 Preliminary Remediation Goals (PRGs). Environmental Protection Agency, Region IX, online document "www.epa.gov/region09/waste/sfund/prg/intro.htm".

Soclo HH, Garrigues P, Ewald M. 2000. Origin of polycyclic aromatic hydrocarbons (PAHs) in coastal marine sediments: case studies in Cotonou (Benin) and Aquitaine (France) areas. *Marine Pollution Bulletin* 40:387-396.

Sparling DW, Eisemann JD, Kuenzel W. 2004. Contaminant exposure and effects in red-winged blackbirds inhabiting stormwater retention ponds. *Environmental Management* 33:719-729.

Stenström TA, Carlander A. 2001. Occurrence and die-off of indicator organisms in the sediment of two constructed wetlands. *Water Science & Technology* 44:223-230.

Sun FS, Littlejohn D., Gibson MD. 1998. Ultrasonication extraction and solid phase extraction clean-up for determination of US EPA 16 priority pollutant polycyclic aromatic hydrocarbons in soils by reversed-phase liquid chromatography with ultraviolet absorption detection. *Analytica Chimica Acta* 364: 1–11.

Sutherland RA, Tolosa CA. 2001. Variation in total and extractable elements with distance from roads in an urban watershed, Honolulu, Hawaii. *Water, Air, and Soil Pollution* 127:315-338.

Turer DG, Maynard JB. 2003. Heavy metal contamination in highway soils. Comparison of Corpus Christi, Texas and Cincinnati, Ohio shows organic matter is the key to mobility. *Clean Technologies and Environmental Policy* 4:235-255.

Urbonas B, Stahre P. 1993. Stormwater: Best Management Practices and Detention for Water Quality, Drainage, and CSP Management. Prentice Hall. pp 39-64.

US Department of Health and Human Services. 2002. Toxicological Profile for DDT, DDE, and DDD. Agency for Toxic Substances and Disease Registry, Atlanta, GA.

US EPA. 1978. Microbiological Methods for Monitoring the Environment: Water and Wastes. EPA-600/8-78-017.

US EPA. 1995. Economic Benefits of Runoff Controls. Office of Wetlands, Oceans and Watersheds, US EPA, Washington, DC. EPA- 841-S-95-002.

US EPA. 1996. Calculation and Evaluation of Sediment Effect Concentrations for the Amphipod *Hyalella azteca* and the Midge *Chironomus riparius*, EPA 905-R96-008, Great Lakes National Program Office, Chicago, IL.

US EPA. 2000. Supplemental Guidance to RAGS: Region 4 Bulletins, Human Health Risk Assessment Bulletins. EPA Region 4, originally published November 1995, Website version last updated May 2000: http://www.epa.gov/region4/waste/ots/healtbul.htm.

US EPA. 2001a. Methods for collection, storage and manipulation of sediments for chemical and toxicological analyses: Technical manual. Office of Water, US EPA, Washington, DC. EPA-823-B-01-002.

US EPA. 2001b. Supplemental Guidance to RAGS: Region 4 Bulletins, Ecological Risk Assessment. Originally published November 1995. Website version last updated November 30, 2001: http://www.epa.gov/region4/waste/ots/ecolbul.htm
US EPA, 2002. Interim reregistration eligibility decision for chlorpyrifos. Prevention, Pesticides and Toxic Substances. EPA 738-R-01-007. Washington, D.C.

US EPA. 2004. User's guide and background technical document for USEPA Region 9's Preliminary Remediation Goals table. Available at http://www.epa.gov/Region9/waste/sfund/prg/whatsnew.htm

Van Metre PC, Mahler BJ, Furlong ET. 2000. Urban sprawl leaves it's PAH signature. *Environmental Science & Technology* 34:4064-4070.

Van Metre PC, Mahler BJ. 2003. The contribution of particles washed from rooftops to contaminant loading to urban streams. *Chemosphere* 52:1727-1741.

Walker SE, Dickhut RM, Chisolm-Brause C, Sylva S, Reddy CM. 2005. Molecular and isotopic identification of PAH sources in a highly industrialized urban estuary. *Organic Geochemistry* 36:619-613.

Wise SA, Hilpert LR, Rebbert RE, Sander LC., Schantz MM., Chesler SN, May WE. 1988. Standard reference materials for the determination of polycyclic aromatic hydrocarbons. *Fresenius Journal of Analytical Chemistry* 332:573–582.

Yunker MB, Macdonald RW, Vingarzan R, Mitchell RH, Goyette D, Sylvestre S. 2002. PAHs in the Fraser River Basin: A critical appraisal of PAH Ratios as indicators of PAH source and composition. *Organic Geochemistry* 33:489-515.

Zeng GJ, Man BK, Lam JC, Lam MH, Lam PK. 2002. Distribution and sources of polycyclic aromatic hydrocarbons in the sediment of a sub-tropical coastal wetland. *Water Research* 36:1457-1468.

Table 1. Summary results of literature survey investigating chemical contamination of bottom sediments in a variety of stormwater management ponds (d.w. = dry weight, n.a. = not available, n.m. = not measured or reported in the study, l.d. = limit of detection).

Location	Pond type/age	Land Use	Organics	Metals	Reference
Surrey, UK	Dry Detention/ n.a.	Highway	ΣPAH = 10,200 ng/g d.w.	Elevated V, Cr, Mn, Fe, Co, Ni, Cu, Zn, Mo, Cd, Sb, Pb	Kamalakkannen *et al.*, 2004
Ontario, Canada	Wet Detention/n.a.	Commercial	ΣPAH = 16,370 ng/g d.w.	Elevated Cd, Cr, Cu, Fe, Pb, Zn	Marsalek & Marsalek, 1997 Marsalek *et al.*, 2002
Seminole, FL	Wet Detention/1 yr.	Residential	ΣPAH < l.d. Pesticides <l.d.	None elevated	Fernandez & Hutchinson, 1993
Clearwater, FL	Wet Detention/20 yr.	Residential	ΣPAH = 592,000 ng/g d.w. Chlordane = 35 μg/g d.w. DDT/DDE = 43 μg/g d.w. Dieldrin = 1.8 μg/g d.w. Hepatochlor = 0.7 μg/g d.w. Diazinon = 1.4 μg/g d.w.	Elevated As, Cd, Cr, Cu, Fe, Pb, Zn	Fernandez & Hutchinson, 1993
Largo, FL	Wet Detention/30 yr.	Commercial	ΣPAH = 7,161,000 ng/g d.w. Chlordane = 1500 μg/g d.w. DDT/DDE = 75 μg/g d.w. Dieldrin = 1.8 μg/g d.w. Diazinon = 4.4 μg/g d.w.	Elevated As, Cd, Cr, Cu, Fe, Pb, Zn	Fernandez & Hutchinson, 1993
Hillcrest, NY	Wet Detention/60 yr.	Industrial	n.m.	Elevated As, Cd, Cr, Cu, Ni, Pb, Zn	Graney & Eriksen, 2004
Minneapolis, MN	Variety	Variety	ΣPAH=227-26,230 ng/g d.w.	Elevated Fe, Cd, Cu, Cr, Pb, Mn, Ni, Zn	Polta *et al.*, 2006
Suburban Maryland	Wet Detention/variety	Variety	n.m.	None elevated	Karouna-Renier & Sparling 2001
Kiawah Island, SC	Wet Detention/n.a.	Low-density residential, golf course	ΣPAH = 84.3-184.7 ng/g d.w.	n.m.	Flemming *et al.*, in press.

Table 2. General characteristics of stormwater and reference ponds. Pond reference numbers are from a baseline water quality study conducted by SCDHEC-OCRM in 2006. Site A and B refer to sampling locations within a pond. Site A=pond inlet, Site B=pond center.

Reference Number	Description	Land-Use Category	Permit Age (yrs)	Pond Area (ha)	Drainage Area (ha)	Impervious Cover (%)	GPS Site A	GPS Site B
R1	Willow Swamp Road Pond	Reference	n/a	1.09	34.56	0	33.01108 N 79.70742 W	33.01101 N 79.70749 W
R2	Dill Plantation	Reference	n/a	0.22	11.69	0	32.72736 N 79.98742 W	32.72769 N 79.98658 W
80	Indigo Run	Golf Course	12	0.141	0.71	0	32.19015 N 80.75673 W	32.18980 N 80.75652 W
62	Arrow Head Country Club	Golf Course	10	2.271	360	n/a	33.70523 N 78.95442 W	33.69812 N 78.95330 W
57	Traditions Golf Course	Golf Course	10	0.675	n/a	n/a	33.48598 N 79.10952 W	33.48583 N 79.10708 W
89	Daniel Island	Low Density Residential	8	0.143	3.14	45.6376	32.85925 N 79.82430 W	32.85916 N 79.82374 W
31	Pawleys Place at Village Green	Low Density Residential	12	0.119	2.47	60.084	33.45006 N 79.12231 W	33.45064 N 79.12190 W
109	Whitehall Plantation	Low Density Residential	10	0.159	4.17	27.3418	32.91583 N 80.12320 W	32.91554 N 80.12350 W
24	Ashton Glenn	Low Density Residential	12	0.037	2.42	48.7288	33.62158 N 78.98856 W	33.62185 N 78.98789 W
38	Ricefields	Low Density Residential	11	0.109	6.83	28.356	33.45738 N 79.15079 W	33.45733 N 79.15103 W
7	Sable Palm Apartments	High Density Residential	4	0.059	2.22	52.2167	32.81444 N 80.07022 W	32.81404 N 80.07022 W
44	Cantebury at Carolina Forest	High Density Residential	8	0.437	8.51	52.5799	33.75906 N 78.95255 W	33.75862 N 78.95226 W
48	Tanger Outlet	Commercial	9	0.087	4.76	77.0241	32.24660 N 80.83227 W	32.24669 N 80.83217 W
70	Wal-Mart at Folly Road	Commercial	12	1.141	10.4	49.9459	32.71969 N 79.96882 W	32.71931 N 79.96924 W
26	Myrtle Beach Chevrolet	Commercial	12	0.113	n/a	n/a	33.70154 N 78.90609 W	33.70159 N 78.90597 W
68	Riverland Woods	Commercial	8	1.596	19.7	67.0549	32.71748 N 79.96966 W	32.71817 N 79.96975 W
83	Toys R Us	Commercial	8	0.634	n/a	n/a	33.71048 N 78.89679 W	33.71067 N 78.89656 W
87	Nascar Speed Track	Commercial	9	0.44	16.1	47.9551	33.71945 N 78.89247 W	33.72018 N 78.89153 W

Table 3. Extraction efficiencies and method detection limits for PAHs based upon NIST SRM 1944. Note that ACNY, ACEN, and FLUR are not certified (nc) components of SRM 1944. Mean±Relative Standard Deviation.

Analyte	Extraction Efficiency (%) (n=4)	Method Detection Limits (ng/g)
NAPT	46.88 ± 6.97	17.04
ACNY	nc	10.43
ACEN	nc	14.57
FLUR	nc	17.77
PHEN	98.41 ± 4.09	9.09
ANTH	48.02 ± 7.70	4.60
FLTH	115.59 ± 11.57	15.25
PYR	112.27 ± 6.72	5.58
B(a)A	114.28 ± 7.13	9.49
CHRY	90.28 ± 8.79	15.63
B(b)F	108.64 ± 14.37	9.19
B(k)F	92.59 ± 16.24	2.29
B(a)P	92.89 ± 20.7	3.00
D(ah)A	103.54 ± 28.21	9.54
B(ghi)P	100.61 ± 11.55	6.44
INPY	99.06 ± 8.31	15.57

Table 4. Summary of GC-MS method: pesticides and PBDE ionization mode, ions used for quantitation, and estimated limit of detection (in extract).

Compound	Ionization Mode for Quantitation	Ions used for Quantitation	Estimated Limit of Detection (ppb)
Dichlorvos (DDVP)	EI	109,185	500
Phosdrin (Mevinphos)	EI	127,192	1000
Disulfoton	EI	88,97	1000
Tetrachloroxylene	EI	207,209,242,244,246	Surrogate
Ethoprop	NCI	199	250
Thimet (Phorate)	EI	75,121,260	200
α-lindane	EI	217	50
Cygon (Dimethoate)	EI	87,125	2000
β+δ-lindane	EI	217	50
Fonofos	NCI	109	10
Dimpylate (diazinon)	NCI	169	20
γ-Lindane	EI	217	50
Chlorpyrifos methyl	NCI	141,212,214,216	10
Heptachlor	EI	270,272,274,335,337,339	40
Methyl parathion	NCI	154,263	100
Pirimiphos methyl	EI	180,290,305	75
Aldrin	EI	261,263,265	25
Malathion	NCI	157	75
Chlorpyrifos	NCI	212,214,313,315	10
Heptachlor epoxide	EI	351,353,355,357	20
γ-chlordane	NCI	35,37,264,266,268,298,300,302	5
α-chlordane	NCI	35,37,264,266,268,298,300,302	5
Endosulfan I	NCI	35,37,240,242,244	25
Dieldrin	NCI	35,37,235,237,239	25
DDE	EI	246,248,316,318,320	25
Endrin	EI	243,245,261,263,265	300
Endosulfan II	NCI	35,37,240,242,244,334,336,338	30
DDD	EI	165,235,237	50
2,4,4'-tribromodiphenyl ether	NCI	79,81,175,177,233,235	30
Ethion	NCI	185	10
Endrin aldehyde	NCI	270,272,274,306,308,310	10
Endosulfan sulfate	NCI	97	10
Methoxychlor	EI	227,228	200
2,2',4,4'-tetrabromodiphenyl ether	NCI	79,81,159,161,163	50

Zolone (Phosalone)	NCI	185	50
Butyl chlorendate	EI	371,373,388,390	Surrogate
Azinphos methyl	EI	132,160	2000
2,2',4,4',5-pentabromodiphenyl ether	NCI	79,81,163	45
Co-Ral (Coumaphos)	NCI	174,225,227	100
2,2',4,4',6-pentabromodiphenyl ether	NCI	79,81,159,161,163	50
2,2',4,4',5,5'-hexabromodiphenyl ether	NCI	79,81	40
2,2',4,4',5,6'-hexabromodiphenyl ether	NCI	79,81	50
2,2',3,4,4',5',6-heptabromodiphenyl ether	NCI	79,81,159,161,163	35

Table 5. Surrogate recoveries by GC-MS. Average of both surrogates (dibutyl chlorendate and 2,4,5,6-tetrachloro-m-xylene) is reported.

Pond	Sample	Recovery (%)
7	B	20.90
83	B	36.50
68	B	39.06
57	A	41.14
24	B	42.32
44	B	43.63
83	A	45.27
38	B	45.73
70	A	46.35
38	A	46.99
26	A	51.38
44	A	53.01
24	A	58.18
89	B	58.71
80	B	61.48
31	B	68.10
68	A	73.24
7	A	75.39
70	B	79.11
87	B	81.02
57	B	84.15
48	B	87.00
87	A	90.14
62	B	100.55
31	A	103.10
26	B	106.92
89	A	109.98
80	A	116.45
48	A	178.50
Mean		70.49
Standard		32.79
Mean (w/o 48a)		66.64
Stdev (w/o 48a)		25.84

Table 6. Comparison of GC-MS results to certified levels in NIST SRM 1944. The average of three replicates with the standard deviation are shown.

Compound	Certified Level (ng/g)	Detected Level (ng/g)
α-chlordane	16.5	16.3±3.5
γ-chlordane	8	14.3±11.0
p,p'-DDD	108	107.5±22.7
p,p'-DDE	86	73.5±32.1

Table 7. Limit of detection for elements analyzed in this study (mg/kg dry weight).

Element	LOD (mg/kg)
Al	100
Cd	1.5
Cr	10
Cu	5
Fe	10
Pb	1
Zn	1

Table 8. Maintenance activities in the stormwater ponds surveyed in the current study.

Pond Reference Number	Description	Maintenance Activities
R1	Willow Swamp Road Pond	n/a
R2	Dill Plantation	n/a
80	Indigo Run	No sediment removal. Algaecides used include glyphosate, $CuSO_4$ and Cutrine-Plus. Vegetation cleared from pond as needed. Aerators present.
62	Arrow Head Country Club	No sediment removal. Algaecides commonly used include $CuSO_4$ and glyphosate (Rodeo). Vegetation cleared from pond.
57	Traditions Golf Course	No sediment removal. Algae problems, but management firm unable to disclose information concerning specific algaecides used. Pond stocked with carp. Aerators present.
89	Daniel Island	No sediment removal. Unwanted vegetation removed from pond. Pond stocked with carp.
31	Pawleys Place at Village Inn	No sediment removal. Algaecides used include $CuSO_4$, Cutrine and Round-up.
109	Whitehall Plantation	No sediment removal. Algaecides used include chelated copper products (e.g. Captain Liquid Copper Algaecide). Vegetation removed as needed. Ponds stocked with carp and talapia.
24	Ashton Glenn	No sediment removal. No recent use of pesticides or algaecides, however, they were used prior to 2006.
38	Ricefields	No sediment removal. Algaecides used include $CuSO_4$, chelated copper products, peroxides (e.g., Green Clean), and endothall (e.g. Hydrosol 191). Shade products commonly used include dyes. Vegetation treated chemically each month. Pond stocked with carp.
7	Sable Palm Apartments	Maintenance activities unknown.
44	Cantebury at Carolina Forest	No sediment removal.
48	Tanger Outlet	No sediment removal. Algaecides used including $CuSO_4$ and Reward (Diquat). Pond stocked with carp.
70	Wal-Mart at Folly Road	No sediment removal. Algaecides used as needed (no specifics provided). Vegetation cleared out as needed. Trash and debris removed on a regular basis.
26	Myrtle Beach Chevrolet	No sediment removal.
68	Riverland Woods	No sediment removal. Algaecides used include Cutrine-Plus, Diquat, and Glyphosate. Pond aerated.
83	Toys R Us	No sediment removal. Other maintenance activities unknown.
87	Nascar Speed Track	No known sediment removal or other maintenance activities.

Table 9. Physical characteristics of stormwater and reference pond sediments. Site A=pond inlet, Site B=pond center.

Pond #	Site	Moisture (%)	TOC (%)	Sand(%)	Silt(%)	Clay(%)
R1	A	26.82	0.20	92.72	4.04	3.24
	B	22.78	0.10	92.71	6.96	0.33
R2	A	87.06	14.95	28.30	45.47	26.23
	B	86.23	11.07	12.91	7.57	79.52
89	A	24.52	0.63	97.90	1.68	0.42
	B	66.49	2.02	29.60	31.82	38.57
31	A	20.83	0.36	97.93	1.40	0.67
	B	22.39	0.42	93.82	5.33	0.84
109	A	39.46	0.90	84.87	5.04	10.08
	B	67.21	2.61	20.42	36.40	43.18
24	A	29.54	0.32	97.37	1.17	1.46
	B	71.88	14.51	24.18	34.46	41.35
38	A	48.47	3.15	80.53	2.43	17.04
	B	59.90	0.59	89.19	8.84	1.96
7	A	61.57	1.10	74.37	10.72	14.91
	B	68.23	2.05	53.86	16.78	29.36
44	A	21.97	1.10	96.10	1.80	2.10
	B	63.45	7.23	14.75	29.38	55.87
68	A	74.39	2.85	47.87	8.69	43.44
	B	75.41	3.26	67.21	7.77	25.02
80	A	39.41	0.36	93.41	2.93	3.66
	B	80.47	7.37	60.82	33.15	6.03
62	A	80.19	10.10	7.41	32.73	59.86
	B	71.11	2.89	34.65	1.01	64.35
57	A	17.36	0.57	98.39	1.34	0.27
	B	77.21	8.81	27.97	57.17	14.86
70	A	59.97	4.35	83.47	4.86	11.67
	B	60.05	1.85	62.45	13.91	23.64
48	A	25.34	0.12	98.29	0.34	1.37
	B	77.92	13.79	35.94	21.35	42.71
26	A	18.40	1.12	97.94	0.34	1.72
	B	59.87	3.74	33.96	31.20	34.83
83	A	59.90	0.84	81.06	9.24	9.70
	B	57.29	1.54	64.58	10.55	24.87
87	A	70.99	4.35	6.42	31.52	62.06
	B	78.99	3.16	4.96	33.06	61.98

Table 10. Correlation matrix for the physical, chemical and biological characteristics of stormwater ponds. Significant correlations (α=0.1) are in bold.

	Drainage Area	Pond Surface Area	Percent Impervious Cover	Permit Age	TOC	Percent Clay	ΣPAH_{16}	ΣPAH_{LMW}	ΣPAH_{HMW}	Al	Cd	Cu	Fe	Pb	Zn	Chlorpyrifos	Toxicity	Sediment Fecal Coliform
Drainage Area	1	**0.74825** **<.0001**	0.32168 0.1253	-0.2370 0.2647	0.31132 0.1387	**0.44464** **0.0295**	**0.57019** **0.0036**	**0.48823** **0.0155**	**0.57367** **0.0034**	0.13252 0.537	0.31561 0.133	0.03503 0.8709	0.21273 0.3183	**0.53194** **0.0075**	**0.60519** **0.0017**	0.25277 0.2334	-0.4173 0.1772	**0.48998** **0.0151**
Pond Surface Area		1	0.02098 0.9225	-0.1437 0.503	0.26597 0.209	**0.35746** **0.0864**	**0.3714** **0.074**	0.34002 0.104	0.36443 0.08	0.11508 0.5923	0.32258 0.1242	-0.2119 0.3202	0.1517 0.4792	**0.63044** **0.001**	**0.52409** **0.0086**	**0.45423** **0.0258**	-0.2562 0.4215	**0.56147** **0.0043**
Percent Impervious Cover			1	-0.3125 0.1371	0.01395 0.9484	0.01918 0.9291	0.01395 0.9484	0.01395 0.9484	-0.01221 0.9549	-0.0506 0.8145	0.21273 0.3183	-0.3398 0.1043	0.15868 0.459	0.01791 0.9338	0.14214 0.5076	-0.0019 0.993	-0.0037 0.991	-0.0906 0.6735
Permit Age				1	-0.1218 0.5706	-0.3421 0.1017	-0.1308 0.5425	-0.12808 0.5509	-0.1487 0.4881	-0.3260 0.12	-0.1451 0.4987	-0.0693 0.7478	-0.0251 0.9074	-0.3064 0.1454	-0.2571 0.2252	-0.03222 0.8812	-0.2557 0.4225	-0.0081 0.9702
TOC					1	**0.74668** **<.0001**	**0.77973** **<.0001**	**0.83975** **<.0001**	**0.74494** **<.0001**	**0.82757** **<.0001**	0.32746 0.1183	**0.39048** **0.0592**	**0.69102** **0.0002**	**0.62447** **0.0011**	**0.78295** **<.0001**	0.30431 0.1482	0.3557 0.2566	**0.67362** **0.0003**
Percent Clay						1	**0.69391** **0.0002**	**0.76348** **<.0001**	**0.66087** **0.0004**	**0.78957** **<.0001**	**0.43391** **0.0341**	**0.45328** **0.0261**	**0.39217** **0.058**	**0.63237** **0.0009**	**0.79887** **<.0001**	0.20851 0.3282	0.0915 0.7773	**0.66522** **0.0004**
ΣPAH_{16}							1	**0.9513** **<.0001**	**0.99217** **<.0001**	**0.60435** **0.0018**	0.20435 0.3382	**0.41922** **0.0414**	**0.51565** **0.0099**	**0.60289** **0.0018**	**0.72407** **<.0001**	0.13743 0.5219	-0.2270 0.4782	**0.5913** **0.0023**
ΣPAH_{LMW}								1	**0.91478** **<.0001**	**0.65304** **0.0005**	0.19565 0.3595	**0.42446** **0.0387**	**0.55391** **0.005**	**0.61808** **0.0013**	**0.73364** **<.0001**	0.05687 0.7918	-0.0476 0.8833	**0.67478** **0.0003**
ΣPAH_{HMW}									1	**0.57826** **0.0031**	0.18957 0.375	**0.43669** **0.0329**	**0.49739** **0.0134**	**0.59664** **0.0021**	**0.71276** **<.0001**	0.15449 0.471	-0.3111 0.325	**0.54522** **0.0059**
Al										1	**0.53826** **0.0067**	**0.4821** **0.017**	**0.61739** **0.0013**	**0.57789** **0.0031**	**0.7232** **<.0001**	0.27676 0.1905	**0.5783** **0.0489**	**0.59391** **0.0022**
Cd											1	-0.0218 0.9193	**0.5287** **0.0079**	**0.47338** **0.0195**	**0.60491** **0.0017**	0.29192 0.1663	0.01098 0.973	**0.50783** **0.0113**
Cu												1	0.10568 0.6231	**0.31667** **0.1316**	**0.40227** **0.0513**	-0.23418 0.2707	0.033 0.9189	0.22795 0.284
Fe													1	**0.37067** **0.0746**	**0.59361** **0.0022**	0.31467 0.1342	0.4319 0.1609	**0.56957** **0.0037**
Pb														1	**0.85674** **<.0001**	**0.43711** **0.0327**	-0.1723 0.5923	**0.72169** **<.0001**
Zn															1	**0.3446** **0.0991**	0.01098 0.973	**0.74494** **<.0001**
Chlorpyrifos																1	0.20115 0.5307	0.34121 0.1027
Toxicity																	1	0.23791 0.4565
Sediment Fecal Coliform																		1

Table 11. Results of the regression analysis between PAH levels and the clay content or total organic carbon (TOC) content in stormwater pond sediments. The **bold values** indicate the regression was not significant at α=0.05.

Dependent variable	Clay r^2	Clay p value	TOC r^2	TOC p value
ΣPAH_{16}	0.15	0.0199	**0.05**	**0.1798**
ΣPAH_{lmw}	0.19	0.0081	0.11	0.0475
ΣPAH_{hmw}	0.14	0.0235	**0.04**	**0.1985**

Table 12. Results of the one-way ANOVA for individual and total PAH levels. Land use classes with different letters have significantly different levels of PAH.

		Duncan's Multiple Range Test				
Analyte	*p* value	REF	GC	RLD	RHD	COM
ΣPAH_{16}	0.0053	A	AB	A	A	B
ΣPAH_{lmw}	0.0128	AB	AB	A	AB	B
ΣPAH_{hmw}	0.0054	A	AB	A	A	B
NAPT	<0.0001	A	B	B	B	B
ACNY	0.6763					
ACEN	0.0344	A	AB	A	AB	B
FLUR	0.6563					
PHEN	0.0071	A	AB	A	A	B
ANTH	0.0068	AB	AB	A	A	B
FLTH	0.0005	A	A	A	A	B
PYR	0.0085	A	AB	A	A	B
B(a)A	0.0035	A	A	A	A	B
CHYR	0.0060	A	B	B	B	B
B(b)F	0.0007	A	BC	B	BA	C
B(k)F	0.0038	A	A	A	A	B
B(a)P	0.0115	A	A	A	A	B
D(ah)A	0.0002	A	A	A	A	B
B(ghi)P	0.0154	A	A	A	A	B
INPY	0.0085	A	A	A	A	B

Abbreviations: reference (REF), golf course (GC), low-density residential (RLD), high-density residential (RHD), commercial (COM), naphthalene (NAPT), acenapthylene (ACNY), acenapthene (ACEN), fluorene (FLUR), phenanthrene (PHEN), and anthracene (ANTH), fluoranthene (FLTH), pyrene (PYR), benzo[a]anthracene (B(a)A), chrysene (CHRY), benzo[b]fluoranthene (B(b)F), benzo[k]fluoranthene (B(k)F), benzo[a]pyrene (B(a)P), dibenz[a,h]anthracene (D(ah)A), benzo[g,h,i]perylene (B(ghi)P), and indeno[1,2,3,c,d]pyrene (INPY).

Table 13. Results of multiple regression analysis on the dependent variable PAH sediment levels.

Contaminant[a]	df[b]	SS[b]	MS[b]	F	p>F	Parameter	Value	SE[b]	Model r^2
ΣPAH_{16}[c]									
Regression	4	6.012	1.503	18.01	0.0009	Intercept	2.610	0.508	
Error	7	0.584	0.083			Drainage Area	0.192	0.027	
Total	11	6.596				Pond Surface Area	-1.010	0.039	0.912
ΣPAH_{LMW}[d]									
Regression	4	2.191	0.548	9.71	0.0055	Intercept	2.433	0.418	
Error	7	0.395	0.056			Drainage Area	0.120	0.022	
Total	11	2.586				Pond Surface Area	-0.708	0.263	0.847
ΣPAH_{HMW}[e]									
Regression	4	8.053	2.013	20.35	0.0006	Intercept	2.228	0.553	
Error	7	0.693	0.099			Drainage Area	0.221	0.029	
Total	11	8.746				Pond Surface Area	-1.134	0.348	0.818

[a]Only models showing the best model r^2 are shown. Possible entries into the model as independent variables included drainage area, pond surface area, permit age, percent impervious cover in the drainage area. A 0.1 level of significance was required for a parameter's entry into a model.
[b]df = degrees of freedom, SS = sum of squares, MS = mean square, SE = standard error
[c]$Log_{10}(\Sigma PAH_{16})$ = Drainage Area-Pond Surface Area; model $r^2 = 0.912$.
[d]$Log_{10}(\Sigma PAH_{LMW})$ = Drainage Area-Pond Surface Area; model $r^2 = 0.847$.
[e]$Log_{10}(\Sigma PAH_{HMW})$ = Drainage Area-Pond Surface Area; model $r^2 = 0.818$.

Table 14. PAH source diagnostic ratios from sediment analyses.

Pond	FLTH/PYR	PHEN/ANTH	LMW/HMW
R1	18.57	80.85	3.03
R2	0.34	4.74	1.30
80	2.93	77.87	1.74
62	2.68	29.72	0.48
57	2.35	29.56	0.65
89	3.01	38.47	0.60
31	1.97	71.18	1.27
109	1.58	86.72	0.68
24	3.23	39.37	0.77
38	1.44	21.73	0.10
7	2.38	26.15	1.48
44	1.20	17.39	0.49
80	2.93	77.87	1.74
62	2.68	29.72	0.48
57	2.35	29.56	0.65
48	2.27	56.49	1.88
70	3.88	13.86	0.69
26	1.32	12.94	0.11
68	1.20	5.61	0.06
83	1.34	8.44	0.11
87	1.35	23.02	0.12

Abbreviations: fluoranthene (FLTH), pyrene (PYR), phenanthrene (PHEN), anthracene (ANTH), low molecular weight (LMW), and high molecular weight (HMW).

Table 15. Regression analysis results between levels of metals and the clay content or total organic carbon (TOC) content in stormwater pond sediments. The **bold values** indicate the regression was not significant at α=0.05.

	Clay		TOC	
Dependent variable	r^2	p value	r^2	p value
Al	0.45	<0.0001	0.49	<0.001
Cd	0.16	0.0169	0.15	0.0188
Cu	0.25	0.0021	0.14	0.0240
Fe	**0.09**	**0.0689**	0.31	0.0004
Pb	0.17	0.0114	**0.03**	**0.2917**
Zn	0.40	<0.0001	0.21	0.0047

Table 16. Results of the two-way ANOVA for metals. Land use classes with different letters have significantly different levels of metals.

Analyte	Overall *p* value	Class *p* value	Site *p* value	Duncan's Multiple Range Test (Class) REF	GC	RLD	RHD	COM
Al	0.0438	0.8403	0.0018					
Cd	0.1117	0.3732	0.0262					
Cu	0.0394	0.0870	0.0422					
Fe	0.8306	0.8958	0.3185					
Pb	0.0068	0.0090	0.0604	AB	A	AB	AB	B
Zn	0.0056	0.0083	0.0538	A	A	A	A	B

Abbreviations: reference (REF), golf course (GC), low-density residential (RLD), high-density residential (RHD), commercial (COM).

Table 17. Results of multiple regression analysis on the dependent variable metal sediment levels.

Metal[a]	df[b]	SS[b]	MS[b]	F	p>F	Parameter	Value	SE[b]	Model r^2
Copper[c]									
Regression	4	247418	61854	8.07	0.0093	Intercept	-58.58	153.95	
Error	7	53639	7663			Drainage Area	44.57	8.18	
Total	11	301057				Pond Surface Area	-516.71	96.80	0.821
Lead, model 1[d]									
Regression	1	55.89	55.89	25.14	0.0005	Intercept	2.051	0.548	
Error	10	22.23	2.22			Pond Surface Area	4.571	0.912	0.715
Total	11	78.13							
Lead, model 2[e]									
Regression	1	47.78	47.78	15.74	0.0027	Intercept	1.375	0.783	
Error	10	30.36	3.03			Drainage Area	0.350	0.088	0.611
Total	11	78.13							
Zinc[f]									
Regression	4	106363	26591	4.64	0.0381	Intercept	-66.56	133.18	
Error	7	40139	5741			Drainage Area	25.32	7.07	
Total	11	146502				Pond Surface Area	-155.68	83.74	0.726

[a]Only models showing the best model r^2 are shown. Possible entries into the model as independent variables included drainage area, pond surface area, permit age, percent impervious cover in the drainage area. A 0.1 level of significance was required for a parameter's entry into a model.
[b]df = degrees of freedom, SS = sum of squares, MS = mean square, SE = standard error
[c]Cu = Drainage Area-Pond Surface Area; model r^2 = 0.821.
[d]Pb = Pond Surface Area; model r^2 = 0.715.
[e]Pb = Drainage Area; model r^2 = 0.611.
[f]Zn = Drainage Area-Pond Surface Area; model r^2 = 0.726.

Table 18. Summary of pesticide and PBDE detections in stormwater ponds sediments. An unknown chlorinated compound with a retention time of 14.7 minutes was also detected in Ponds 24, 26, 38 and 70.

Reference Number	Description	Pesticides/PBDE Concentrations (ng/g dry weight) Inlet	Center
R1	Willow Swamp Road Pond	DDE (6.1) DDD (17.0)	no detects
R2	Dill Plantation	DDE (37.0) DDD (94.1)	DDE (33.4) DDD (69.7)
80	Indigo Run	ODZ[b]	chlorpyrifos (7.51) ODZ[b]
62	Arrow Head Country Club	no detects	no detects
57	Traditions Golf Course	no detects	chlorpyrifos (11.4) ODZ[b]
89	Daniel Island	chlorpyrifos (10.3) HCB[a]	chlorpyrifos (7.6)
31	Pawleys Place at Village Inn	γ-chlordane (1.5)	no detects
109	Whitehall Plantation	HCB[a]	chlorpyrifos (10.4)
24	Ashton Glenn	chlorpyrifos (2.8)	fonofos (10.0) chlorpyrifos (6.5)
38	Ricefields	no detects	no detects
7	Sable Palm Apartments	no detects	dichlorvos (25.9)
44	Cantebury at Carolina Forest	no detects	chlorpyrifos (27.7)
48	Tanger Outlet	no detects	chlorpyrifos (17.2)
70	Wal-Mart at Folly Road	chlorpyrifos (24.9)	chlorpyrifos (7.1)
26	Myrtle Beach Chevrolet	no detects	no detects
68	Riverland Woods	chlorpyrifos (27.3) DDE (69.7) DDD (65.5) endosulfan sulfate (6.2) 2,2',4,4',5-pentaBDE (30.0) 2,2',4,4',6-pentaBDE (72.8)	chlorpyrifos (8.6) DDE (41.2) DDD (39.6)
83	Toys R Us	α-chlordane (1.3) DDE (17.8) DDD (40.1)	α-chlordane (1.1) DDE (18.7) DDD (21.6)
87	Nascar Speed Track	no detects	no detects

[a]HCB=hexachlorobenzene (tentative identification, not quantified)
[b]ODZ=oxadiazon/Ronstar (tentative identification, not quantified)

Table 19. Results of multiple regression analysis on the dependent variable chloropyrifos sediment levels.

Pesticide[a]	*df*[b]	SS[b]	MS[b]	F	*p*>F	Parameter	Value	SE[b]	Model r^2
Chloropyrifos[c]									
Regression	1	260.49	260.49	16.56	0.0022	Intercept	3.23	1.46	
Error	10	157.31	15.73			Pond Surface Area	9.87	2.43	0.623
Total	11	417.81							

[a]Only models showing the best model r^2 are shown. Possible entries into the model as independent variables included drainage area, pond surface area, permit age, percent impervious cover in the drainage area. A 0.1 level of significance was required for a parameter's entry into a model.
[b]*df* = degrees of freedom, SS = sum of squares, MS = mean square, SE = standard error
[c]Chlorpyrifos = Pond Surface Area; model r^2 = 0.623.

Table 20. Summary of metal ranges in stormwater ponds from the literature. Previous studies examined were Dechesne *et al*., 2004; Sparling *et al.,* 2004; Barraud *et al*., 2005; and Casey *et al*., 2006.

Metal	Previous studies (mg/kg)	Current Study (mg/kg)
Al	4,542-15,799	927-26,977
Cd	0.45-4.1	<1.5-4.2
Cr	10.5-83.1	<8-56.1
Cu	2.5-355.7	<5-589.6
Fe	8,934-38,406	520-15,050
Pb	4.8-930	<1-13.8
Zn	17.3-2,605	<1-572.7

Table 21. Comparison of fecal coliform levels among various aquatic sediments and municipal sludge.

Habitat	Location	Fecal Coliform Levels (units*)	Notes	Reference
Estuarine sediments	New Orleans, LA	16.8-87.5 MPN/g	Background levels	Jeng *et al.*, 2005
Constructed wetlands	Tucson, AZ	50-110 MPN/g	Wastewater treatment	Karim *et al.*, 2004
Recreational beaches	Lake Huron, MI	1-10 CFU/g		Alm *et al.*, 2002
River	Flagstaff, AZ	0-11,000 CFU/g	Cattle grazing, municipal outfalls	Crahill *et al.*, 1999
Lake	Lake Texoma, TX/OK	35,000-500,000 CFU/g	Near marinas	An *et al.*, 2002
Dredge material	Upper Mississippi River	82-4,200 MPN/g		Grimes, 1980
Sludge	Seattle, WA	1,400,000-1,900,000 CFU/g		Edmonds, 1976
Sludge	Winnepeg, MNB, Canada	6,600,000-21,000,000 MPN/g		Bujoczek *et al.*, 2001
Stormwater detention ponds	coastal South Carolina	254-19,400 MPN/g		This study

CFU=colony forming unit
MPN=most probable number

*CFU values are derived from direct plate counts, whereas MPN values are derived from statistical estimates. However, both values are generally considered numerically equivalent.

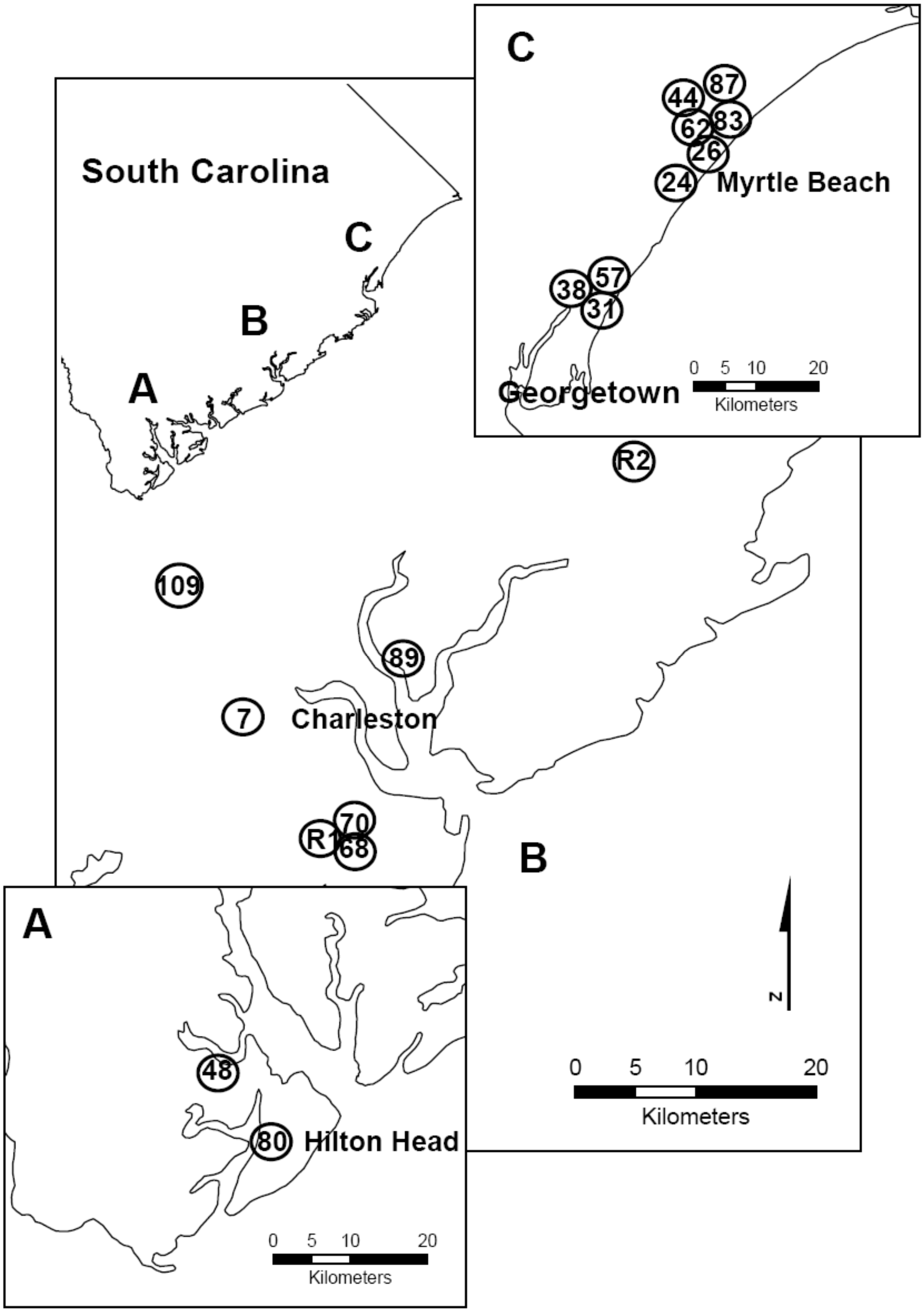


Figure 1. Map showing location of stormwater and reference ponds sampled in coastal South Carolina. Pond reference numbers are from a baseline water quality study conducted by SCDHEC-OCRM in 2006.

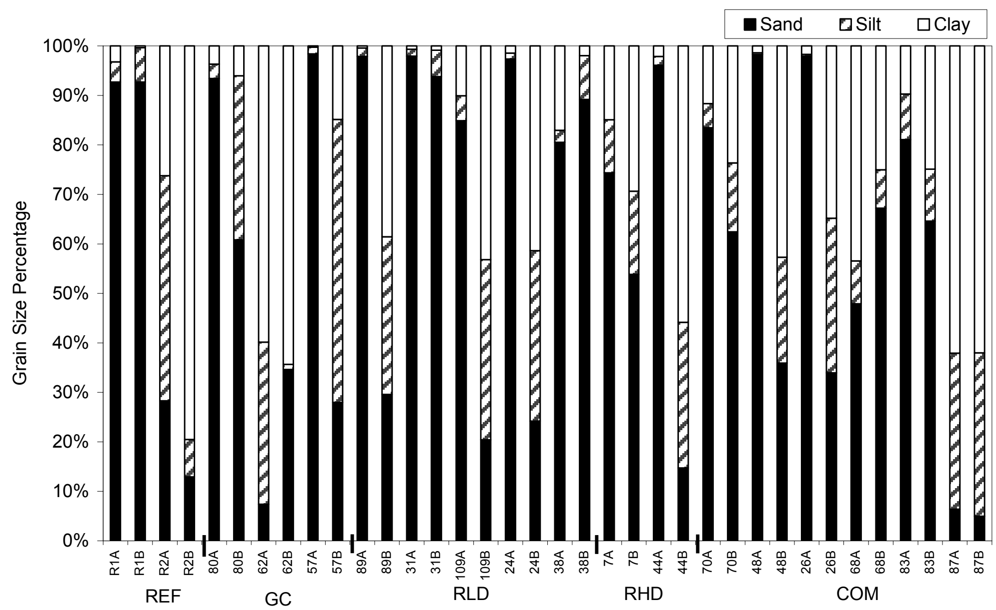


Figure 2. Grain size of sediments from the reference ponds and the four classes of stormwater detention ponds. Designations following pond reference numbers refer to sites with ponds: A=pond inlet, B=pond center. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

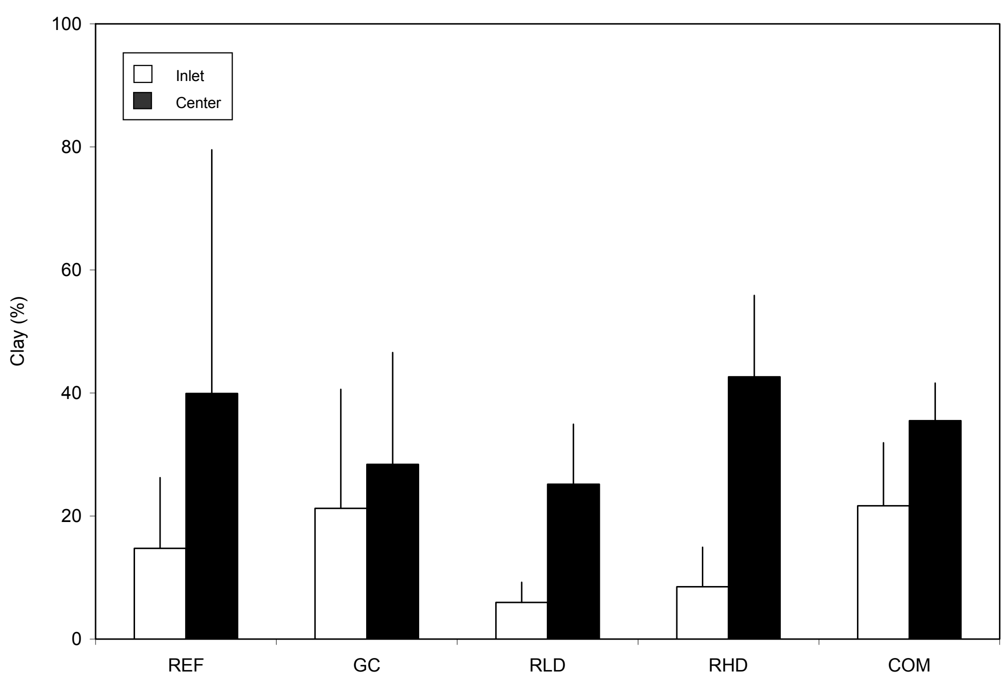


Figure 3. Average clay content (%) of the sediments at the inlet and center of the reference ponds and the four classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

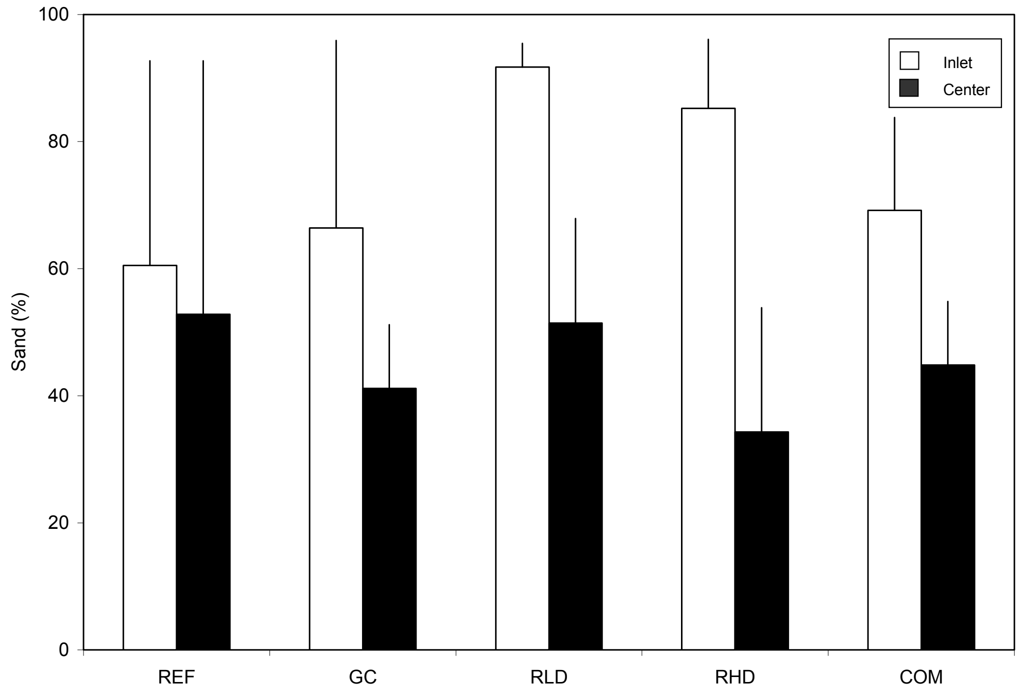


Figure 4. Average sand content (%) of the sediments at the inlet and center of the reference ponds and the four classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

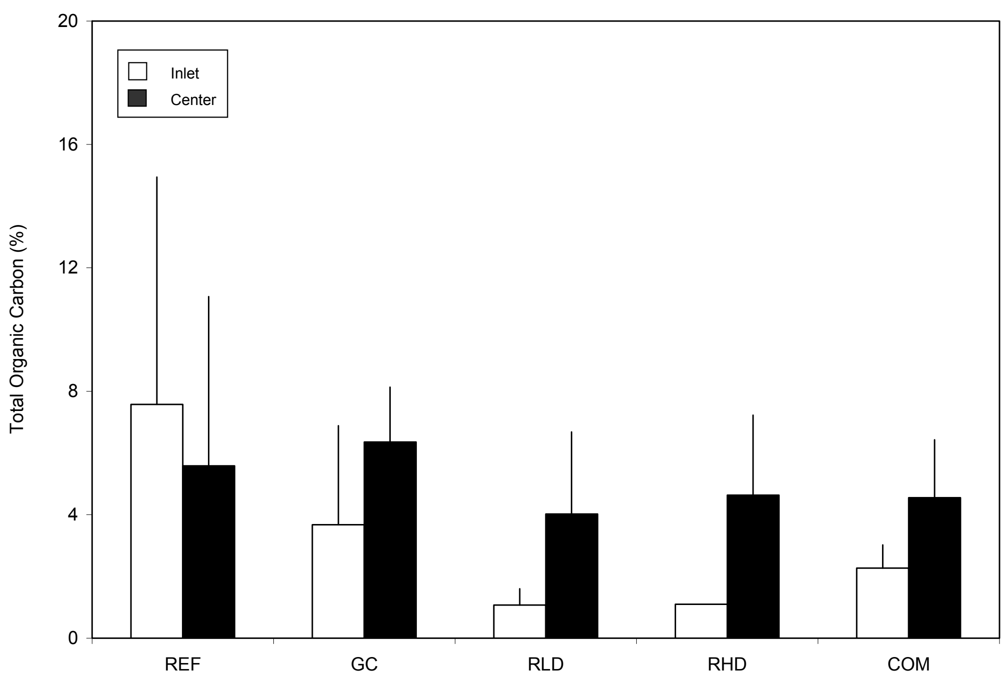


Figure 5. Average total organic carbon (%) of the sediments at the inlet and center of the reference ponds and the four classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

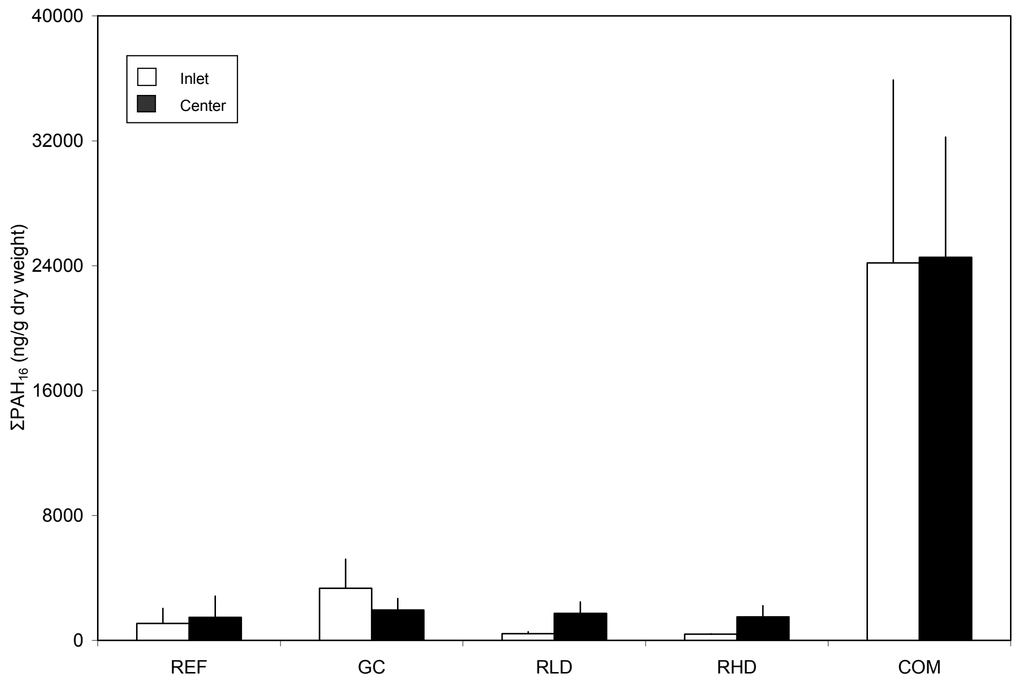


Figure 6. Average ΣPAH_{16} levels in the sediments at the inlet and center of the reference ponds and the four classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

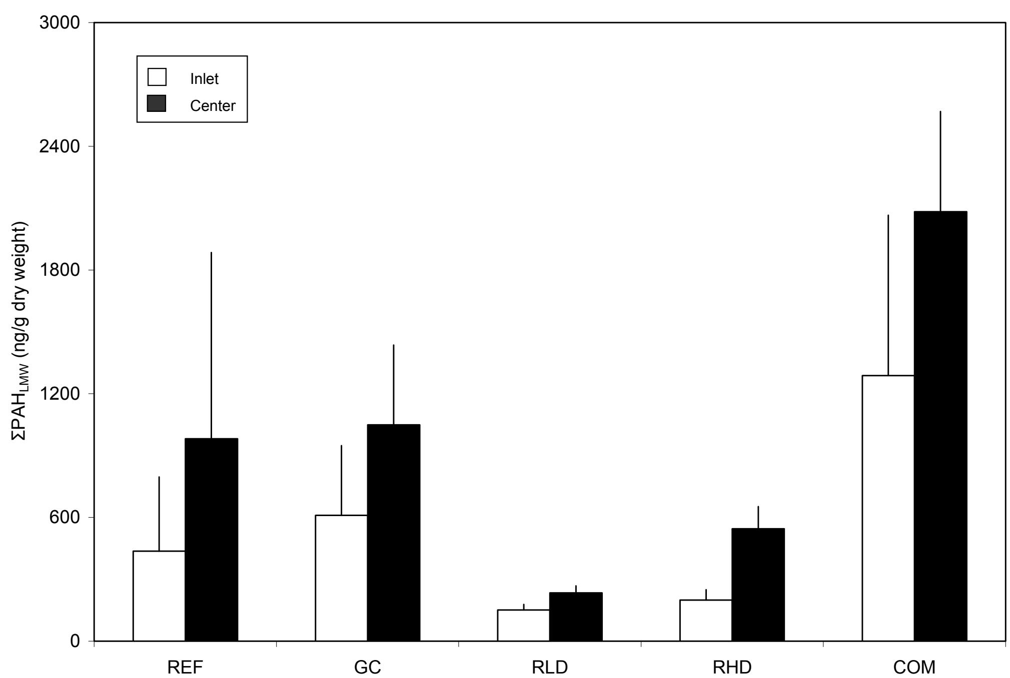


Figure 7. Average ΣPAH_{LMW} levels in the sediments at the inlet and center of the reference ponds and the four classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

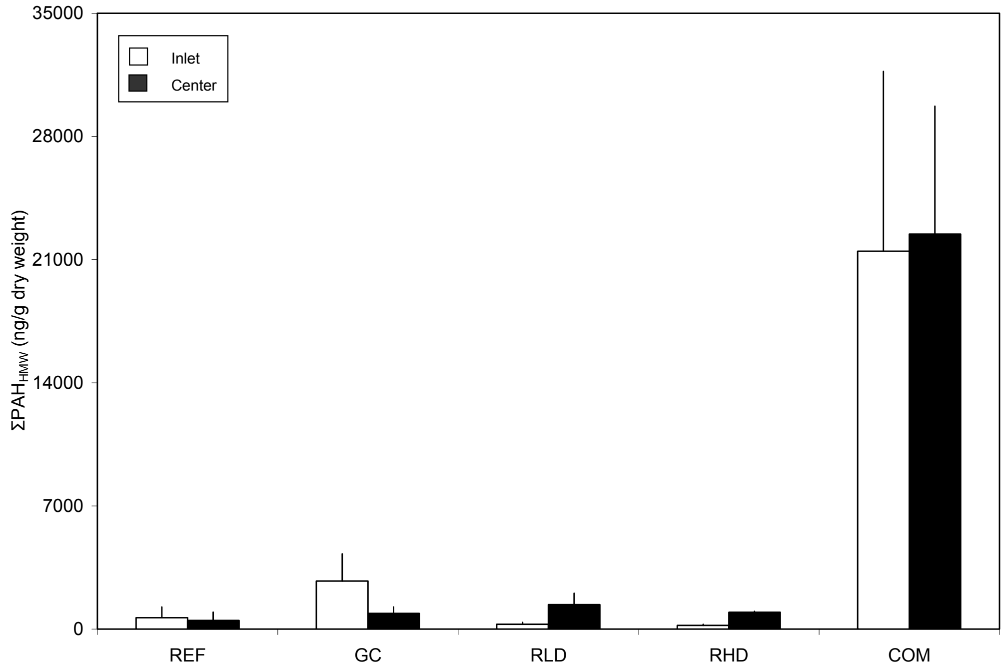


Figure 8. Average ΣPAH_{HMW} levels in the sediments at the inlet and center of the reference ponds and the four classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

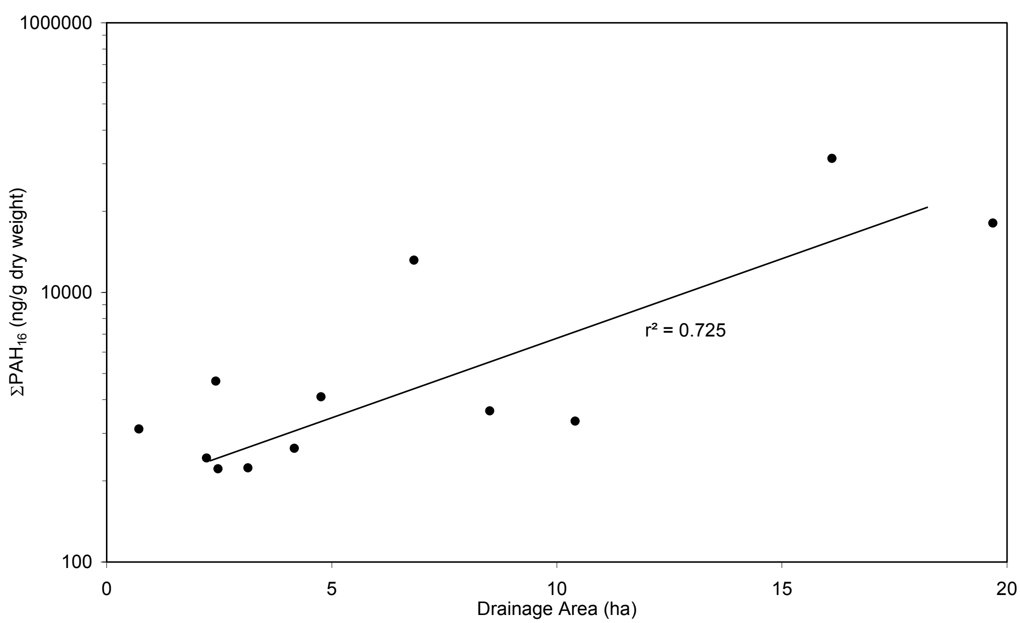


Figure 9. Relationship between ΣPAH_{16} levels in the sediments and drainage area of stormwater detention ponds.

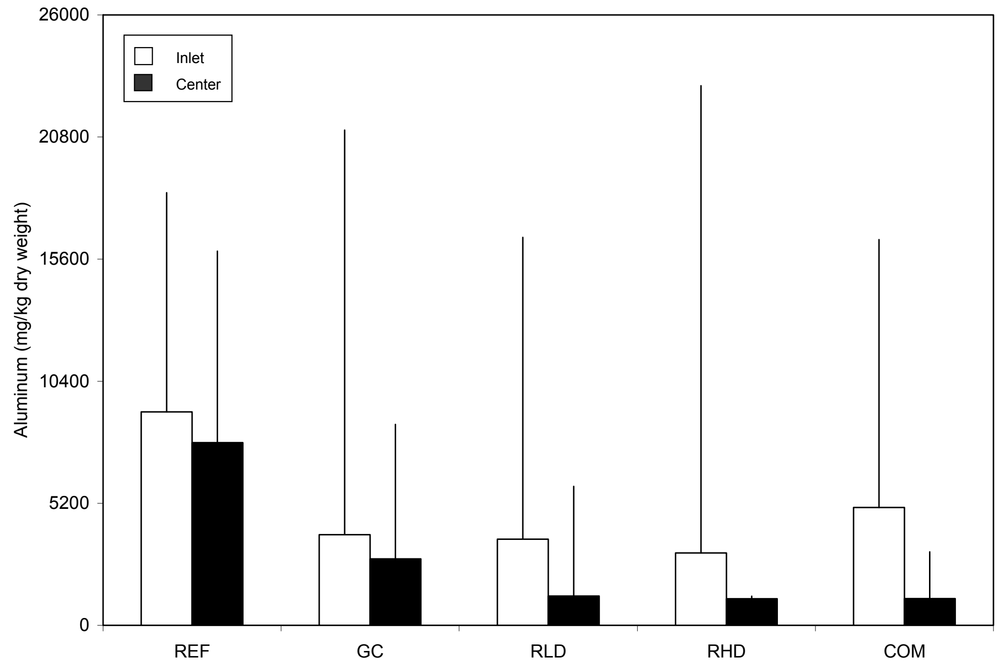


Figure 10. Average aluminum levels in the sediments at the inlet and center of the reference ponds and the four classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

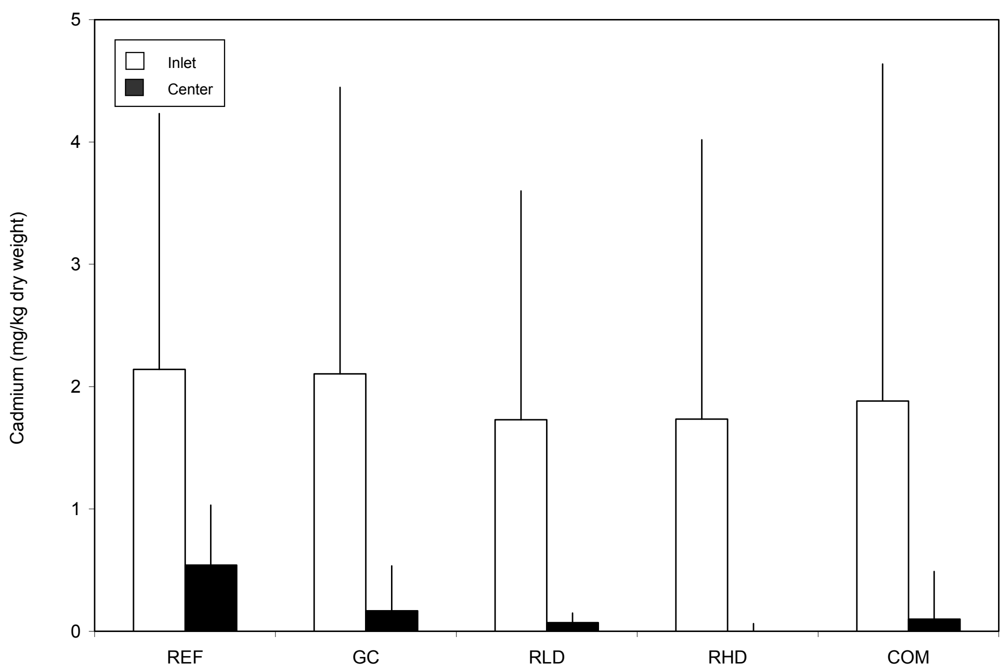


Figure 11. Average cadmium levels in the sediments at the inlet and center of the reference ponds and the 4 classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

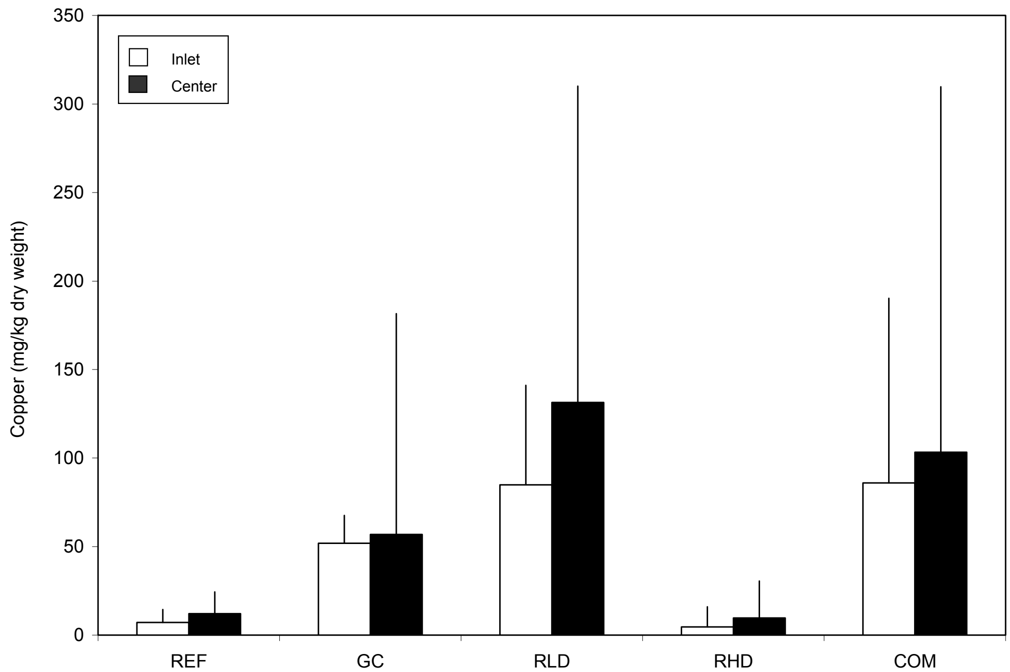


Figure 12. Average copper levels in the sediments at the inlet and center of the reference ponds and the four classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

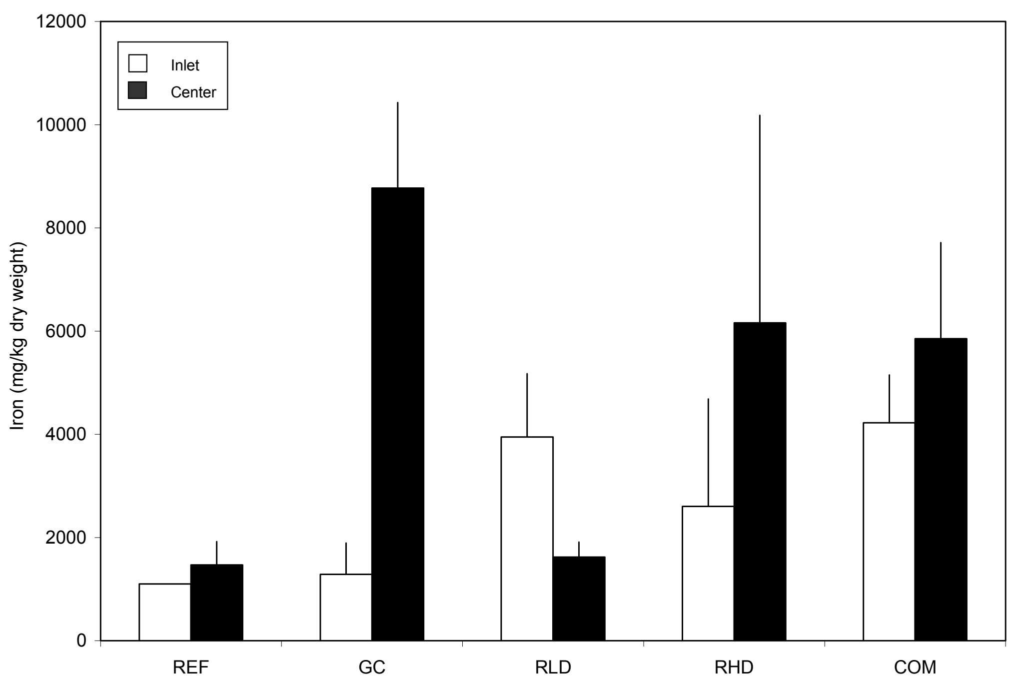


Figure 13. Average iron levels in the sediments at the inlet and center of the reference ponds and the four classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

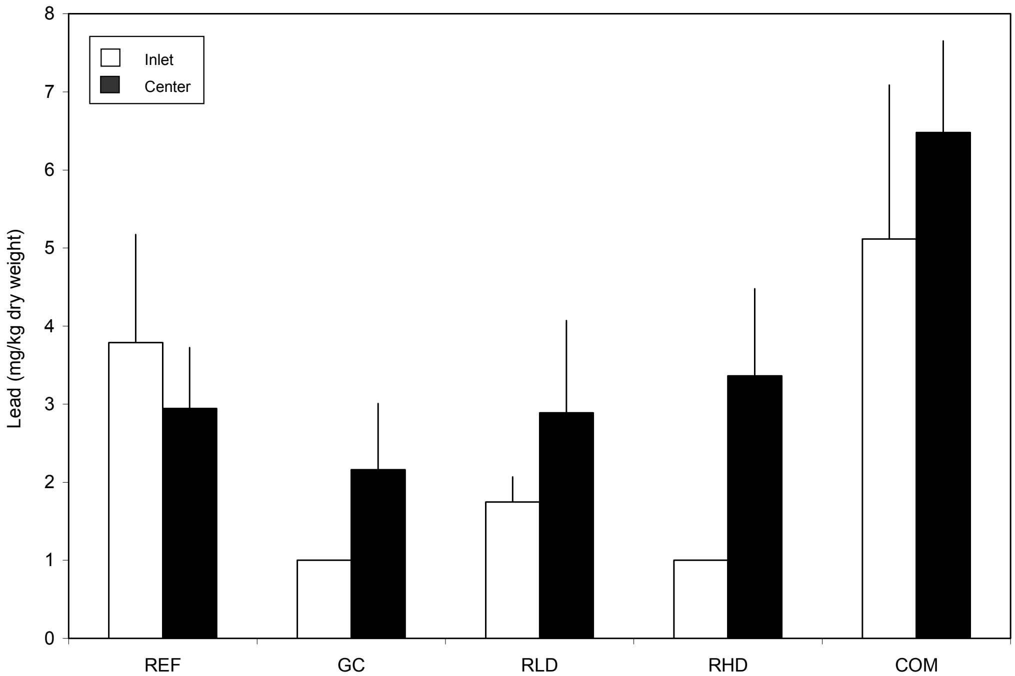


Figure 14. Average lead levels in the sediments at the inlet and center of the reference ponds and the 4 classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

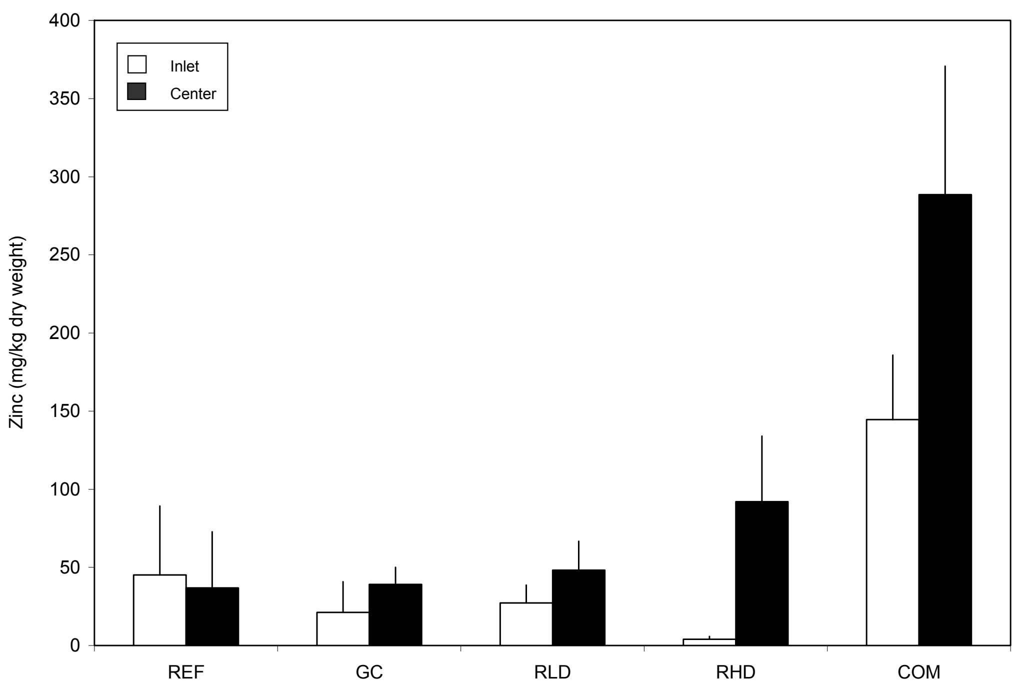


Figure 15. Average zinc levels in the sediments at the inlet and center of the reference ponds and the 4 classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

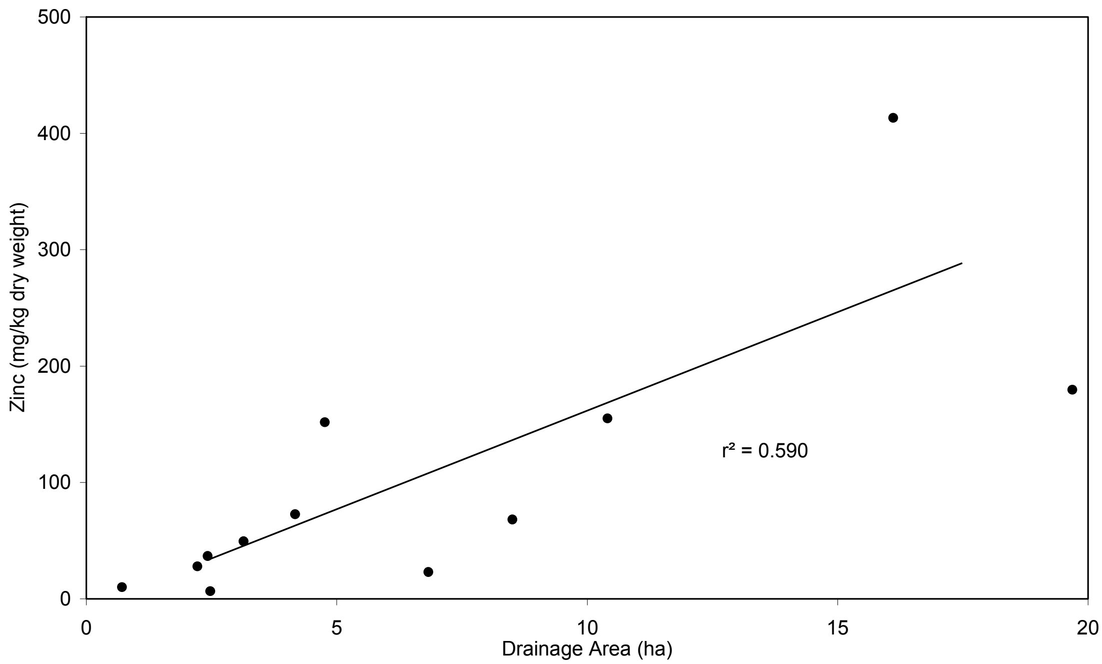


Figure 16. Relationship between zinc levels in the sediments and drainage area of stormwater detention ponds.

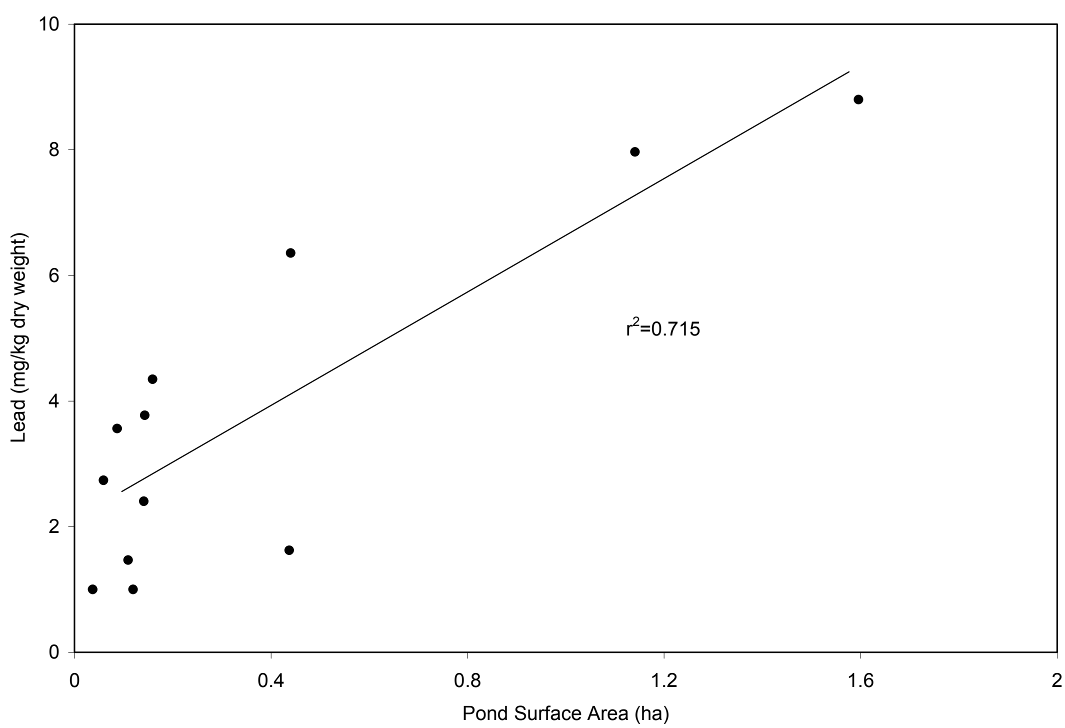


Figure 17. Relationship between lead levels in the sediments and pond surface area of stormwater detention ponds.

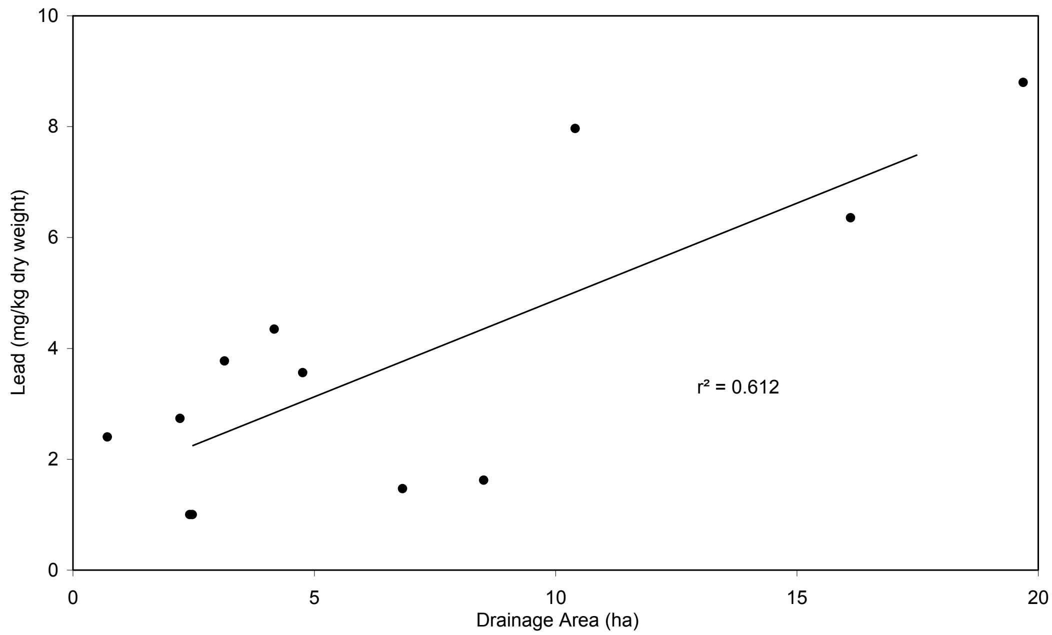


Figure 18. Relationship between lead levels in the sediments and drainage area of stormwater detention ponds.

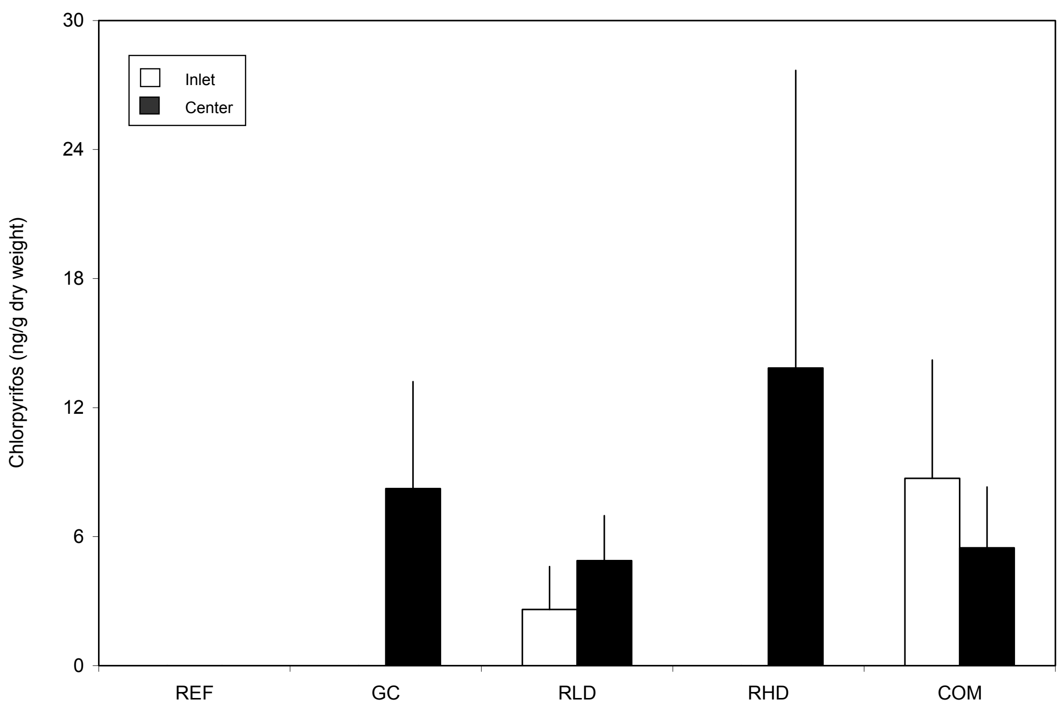


Figure 19. Average chlorpyrifos levels in the sediments at the inlet and center of the reference ponds and the 4 classes of stormwater detention ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

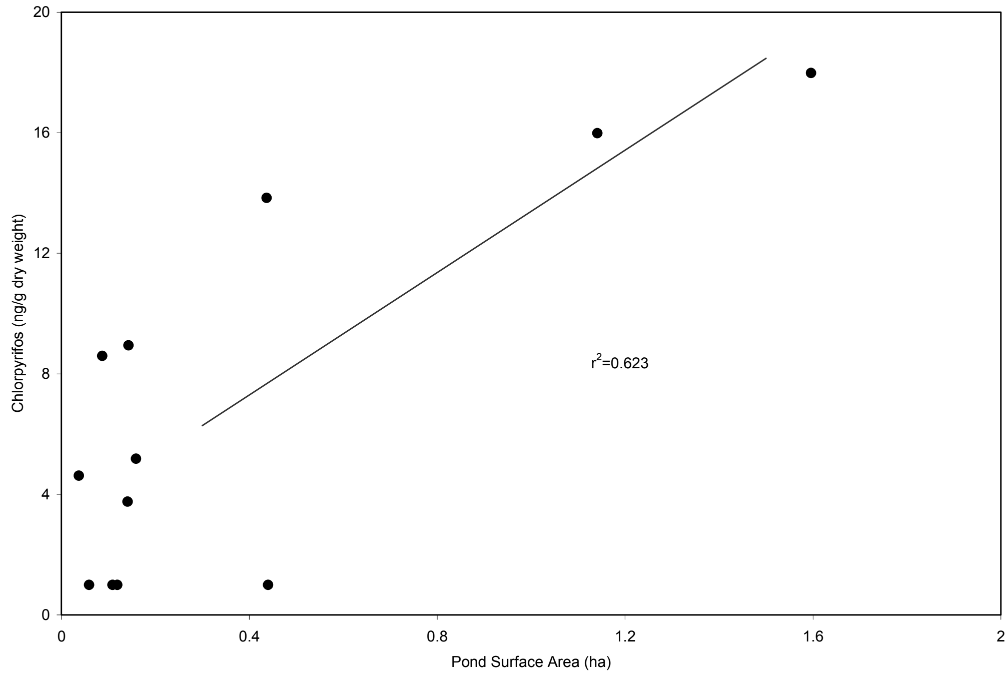


Figure 20. Relationship between chlorpyrifos levels in the sediments and surface area of stormwater detention ponds.

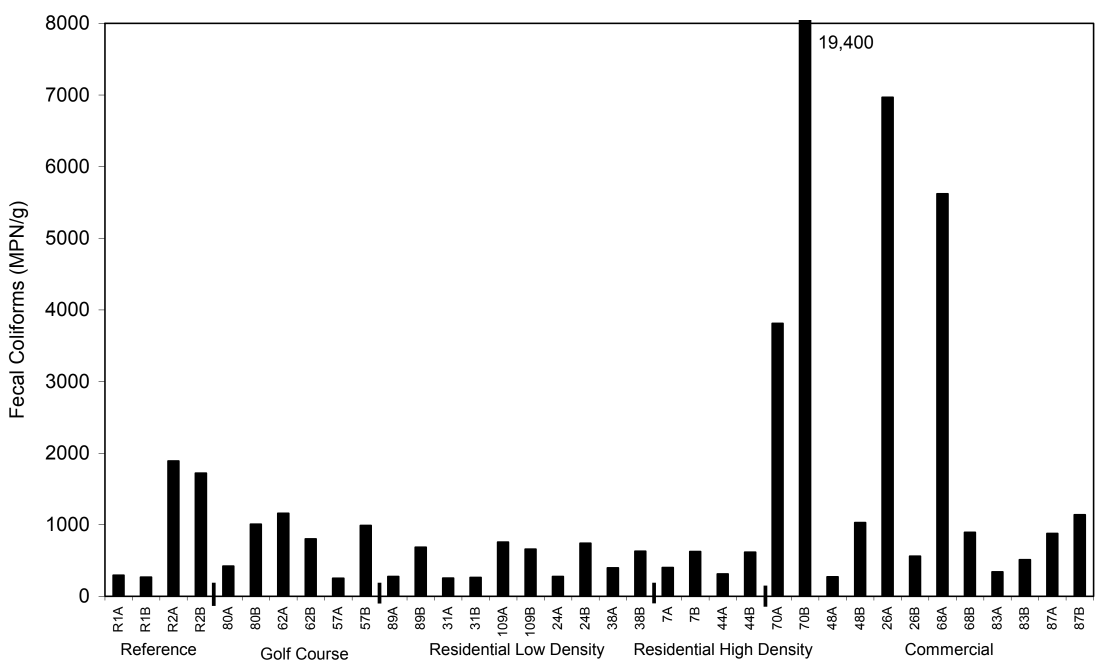


Figure 21. Sediment levels of fecal coliforms from the two reference ponds and 16 stormwater ponds. A=pond inlet, B=pond center.

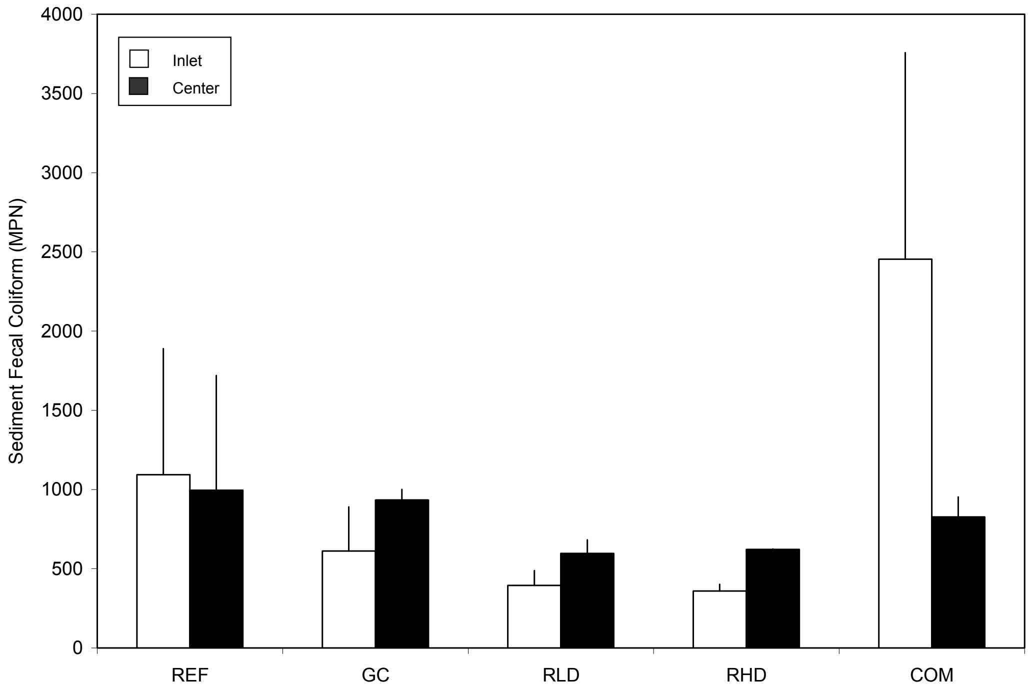


Figure 22. Average sediment fecal coliform from the reference ponds and the four classes of stormwater ponds. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

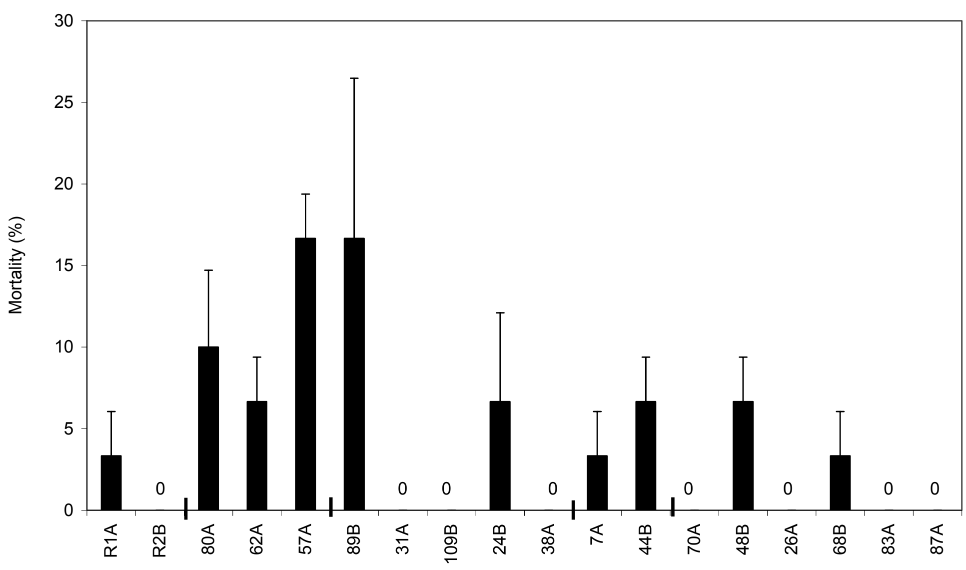


Figure 23. Mortality in the 10-day sediment toxicity tests using the amphipod *Hyalella azteca*. Only one site (either Site A or Site B) per pond was tested. Designations following pond reference numbers refer to sites with ponds: A=pond inlet, B=pond center. Error bars = +1 standard deviation. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

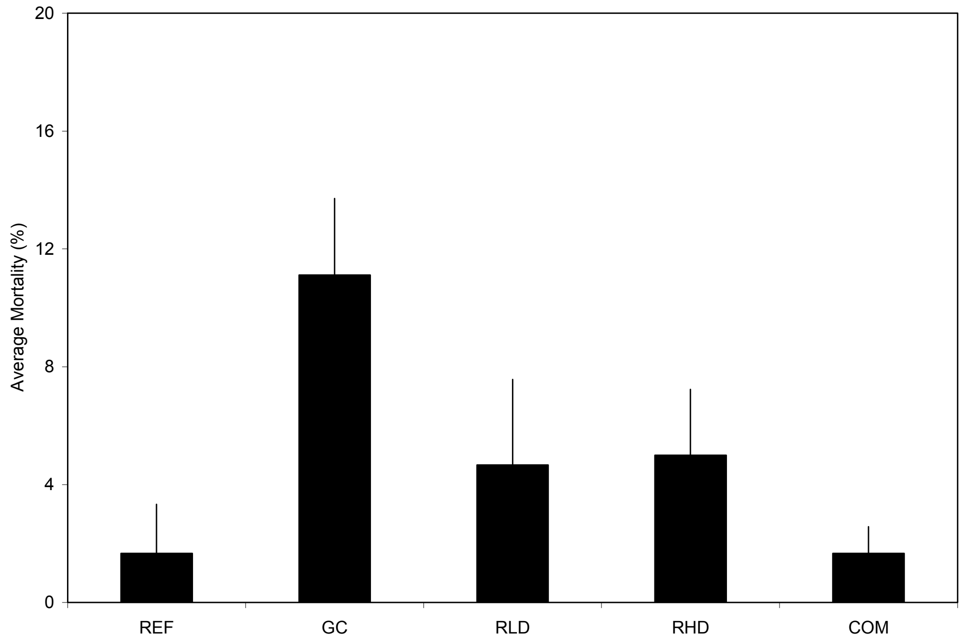


Figure 24. Average mortality in sediments from the reference ponds and the 4 classes of stormwater ponds using the standard 10-day sediment toxicity tests with the amphipod *Hyalella azteca*. Error bars = +1 SE. REF=reference ponds, GC=golf course ponds, RLD=low density residential ponds, RHD=high density residential ponds, and COM=commercial ponds.

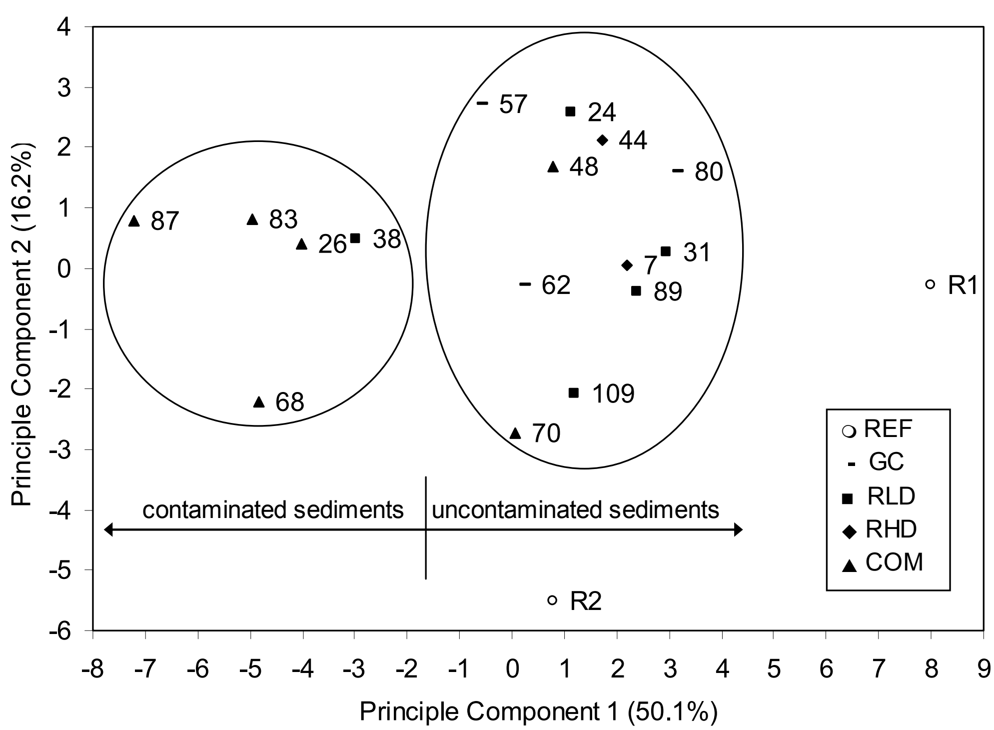


Fig. 25. Relationship between Principal Component 1 and Principal Component 2 for the Principle Component Analysis of chemical and biological contaminants in reference pond and stormwater pond sediments.

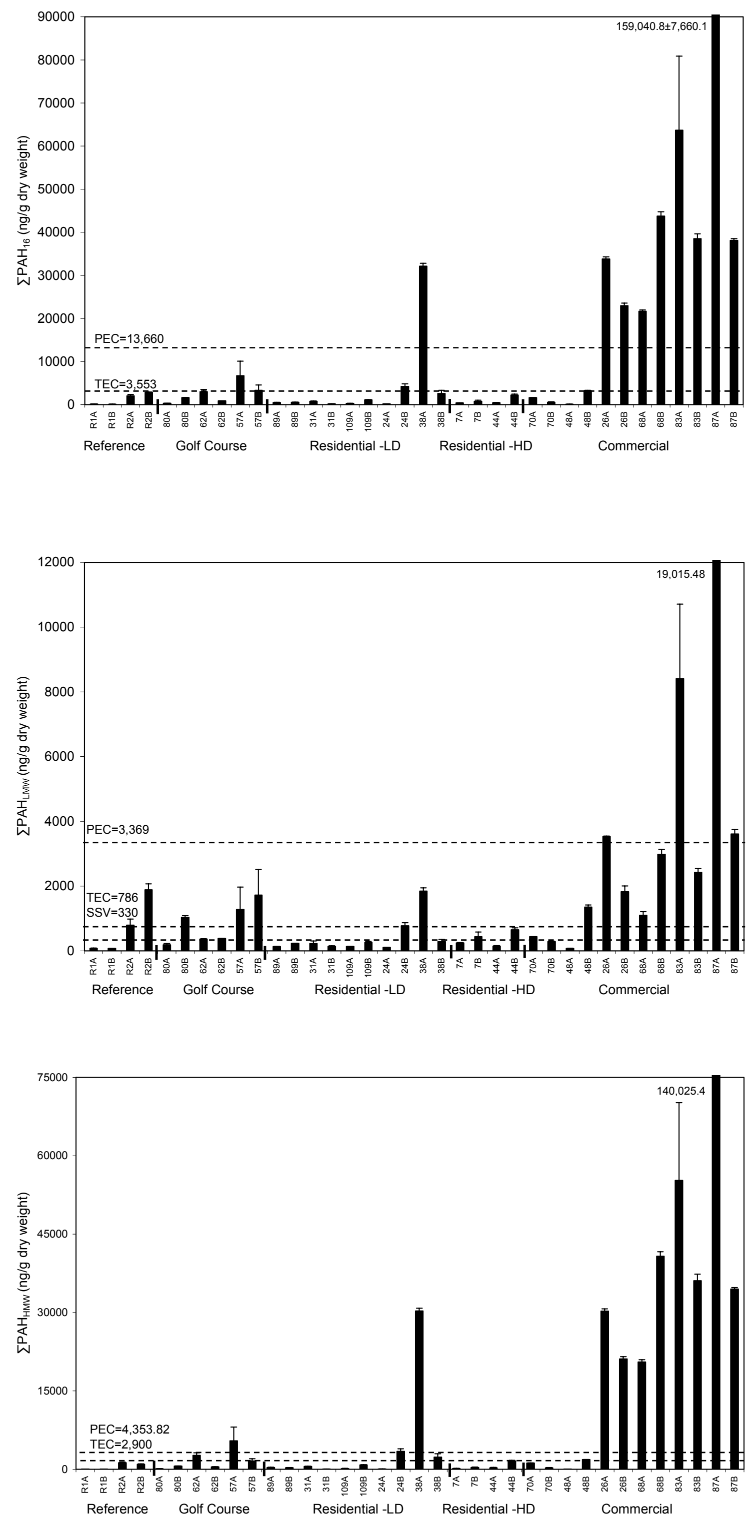


Figure 26. Screening assessment for sediment levels of ΣPAH_{16}, ΣPAH_{LMW} , ΣPAH_{HMW} from the two reference ponds and 16 stormwater ponds. TEC=threshold effect concentration, PEC=probable effect concentration, SSV (sediment screening value). PRG (preliminary remediation goal) values are not available. A=pond inlet, B=pond center. Error bars = +1 SE.

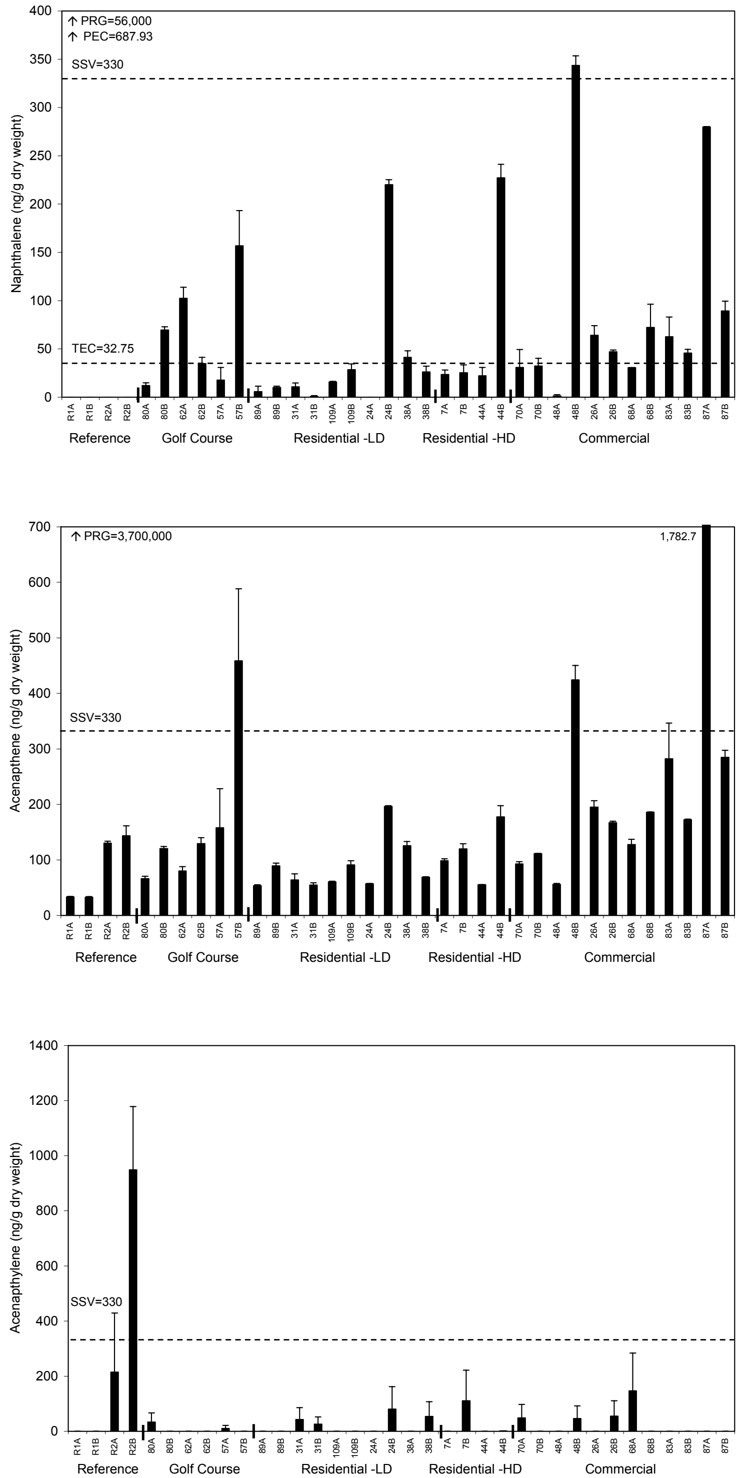


Figure 27. Screening assessment for sediment levels of naphthalene, acenapthene and acenapthylene from the two reference ponds and 16 stormwater ponds. TEC=threshold effect concentration, PEC=probable effect concentration, SSV=sediment screening value, PRG=preliminary remediation goal. A=pond inlet, B=pond center. Error bars = +1 SE.

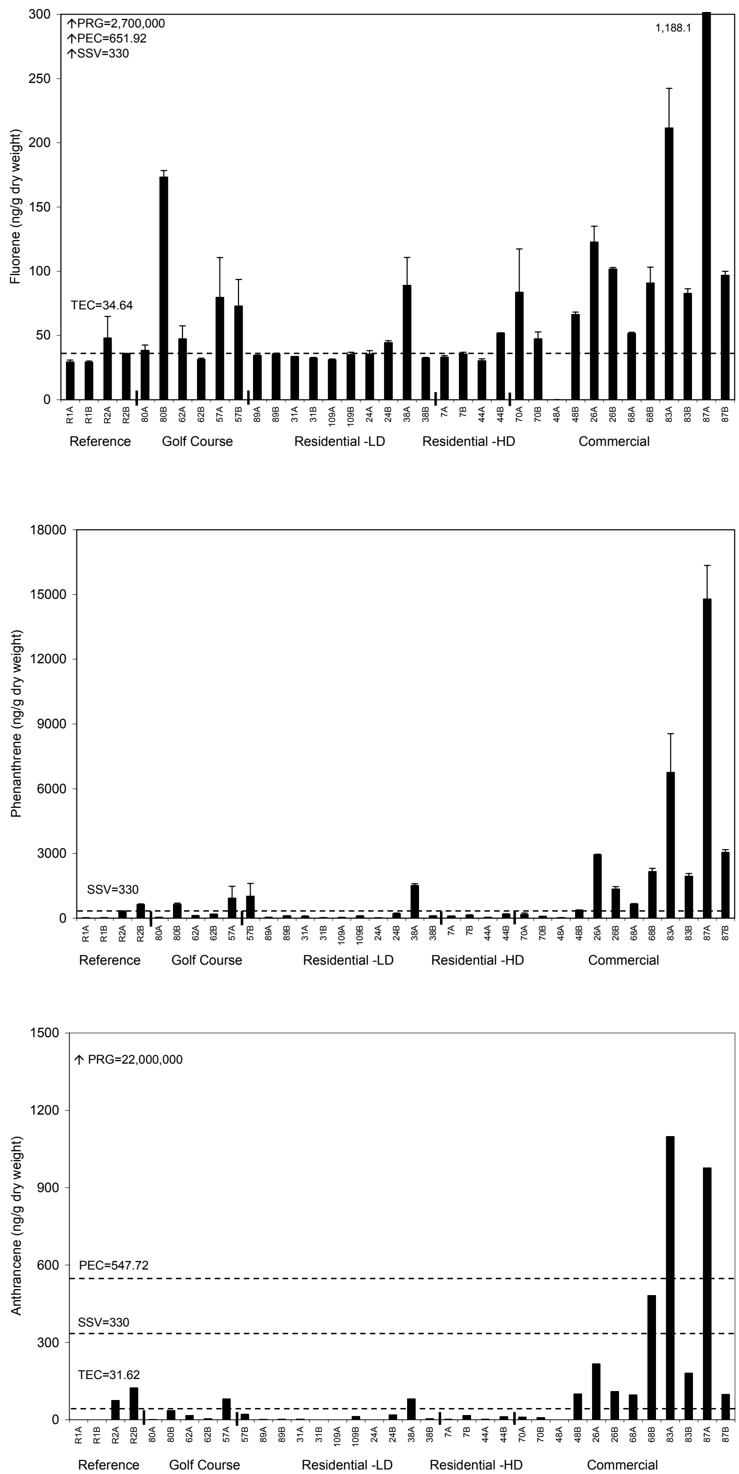
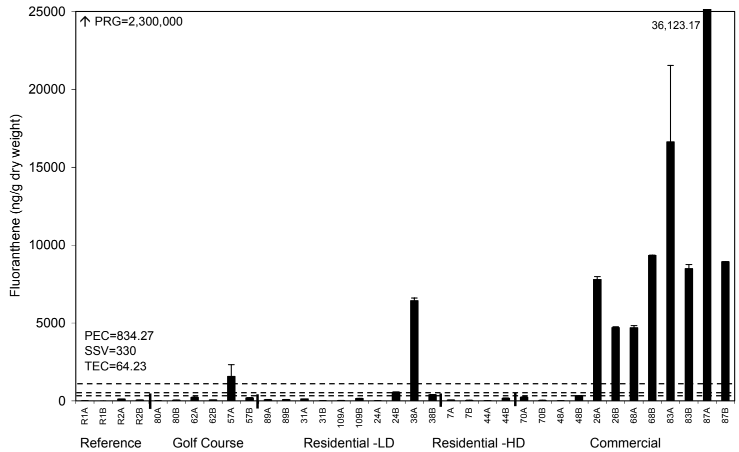
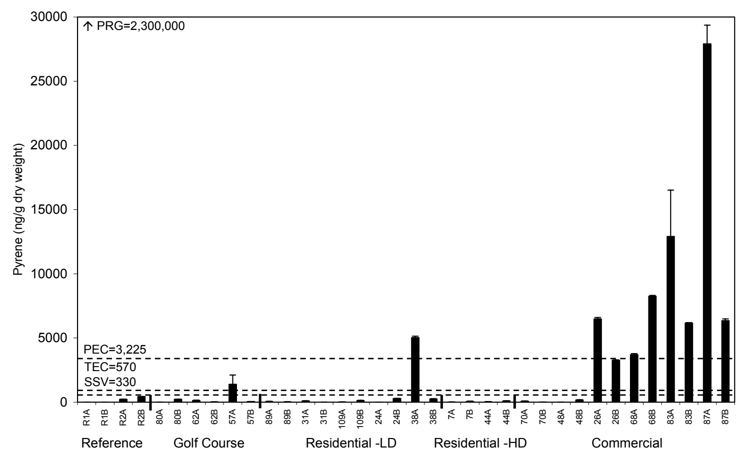


Figure 28 Screening assessment for sediment levels of fluorene, phenanthrene and anthracene from the two reference ponds and 16 stormwater ponds. TEC=threshold effect concentration, PEC=probable effect concentration, SSV=sediment screening value, PRG=preliminary remediation goal. A=pond inlet, B=pond center. Error bars = +1 SE.





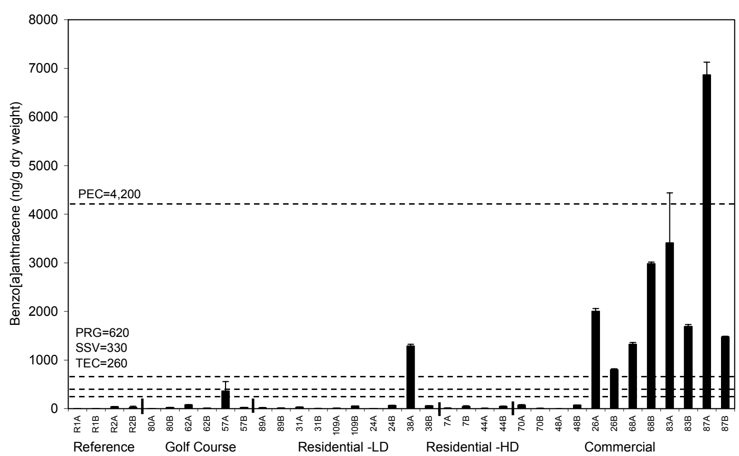


Figure 29. Screening assessment for sediment levels of fluoranthene, pyrene and benzo[a]anthracene from the two reference ponds and 16 stormwater ponds. TEC=threshold effect concentration, PEC=probable effect concentration, SSV=sediment screening value, PRG=preliminary remediation goal. A=pond inlet, B=pond center. Error bars = +1 SE.

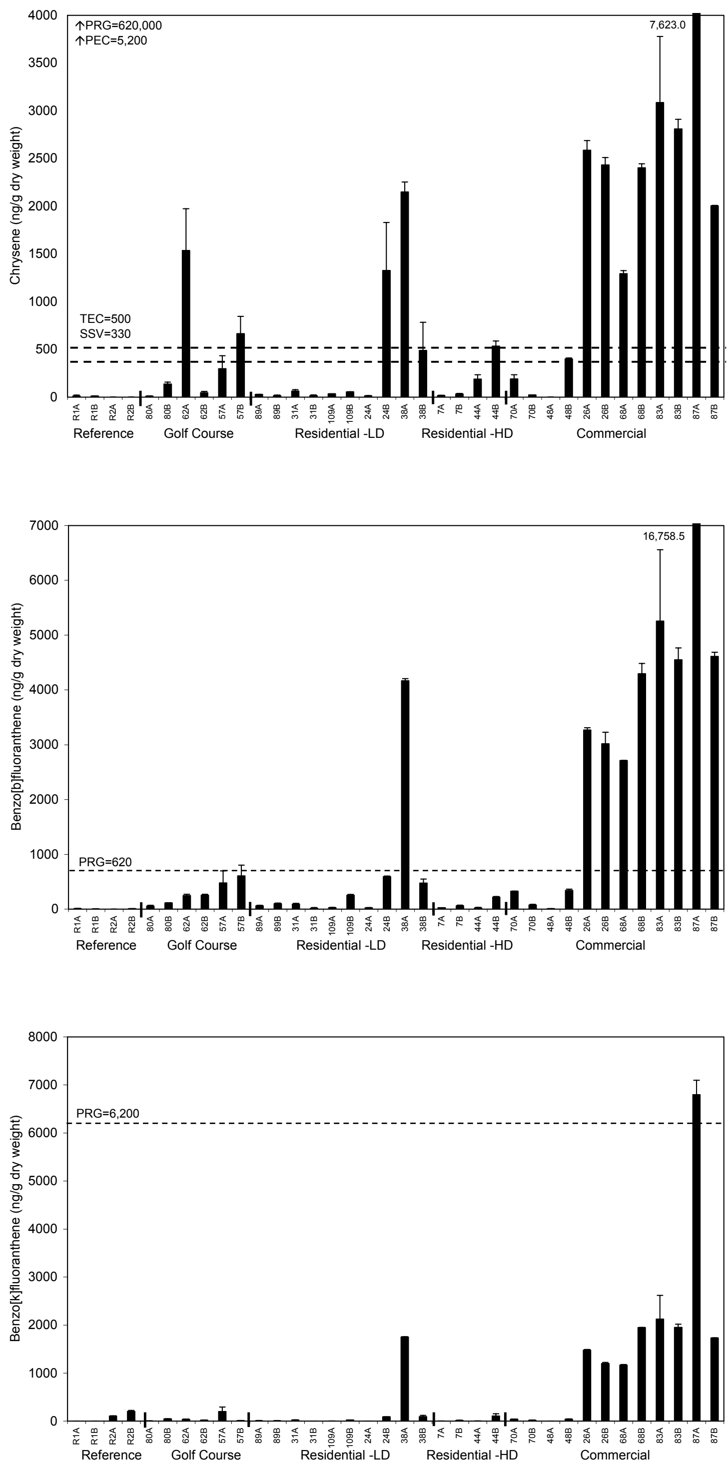


Figure 30. Screening assessment for sediment levels of chrysene, benzo[b]fluoranthene and benzo[k]fluoranthene from the two reference ponds and 16 stormwater ponds. TEC=threshold effect concentration, PEC=probable effect concentration, SSV=sediment screening value, PRG=preliminary remediation goal. A=pond inlet, B=pond center. Error bars = +1 SE.

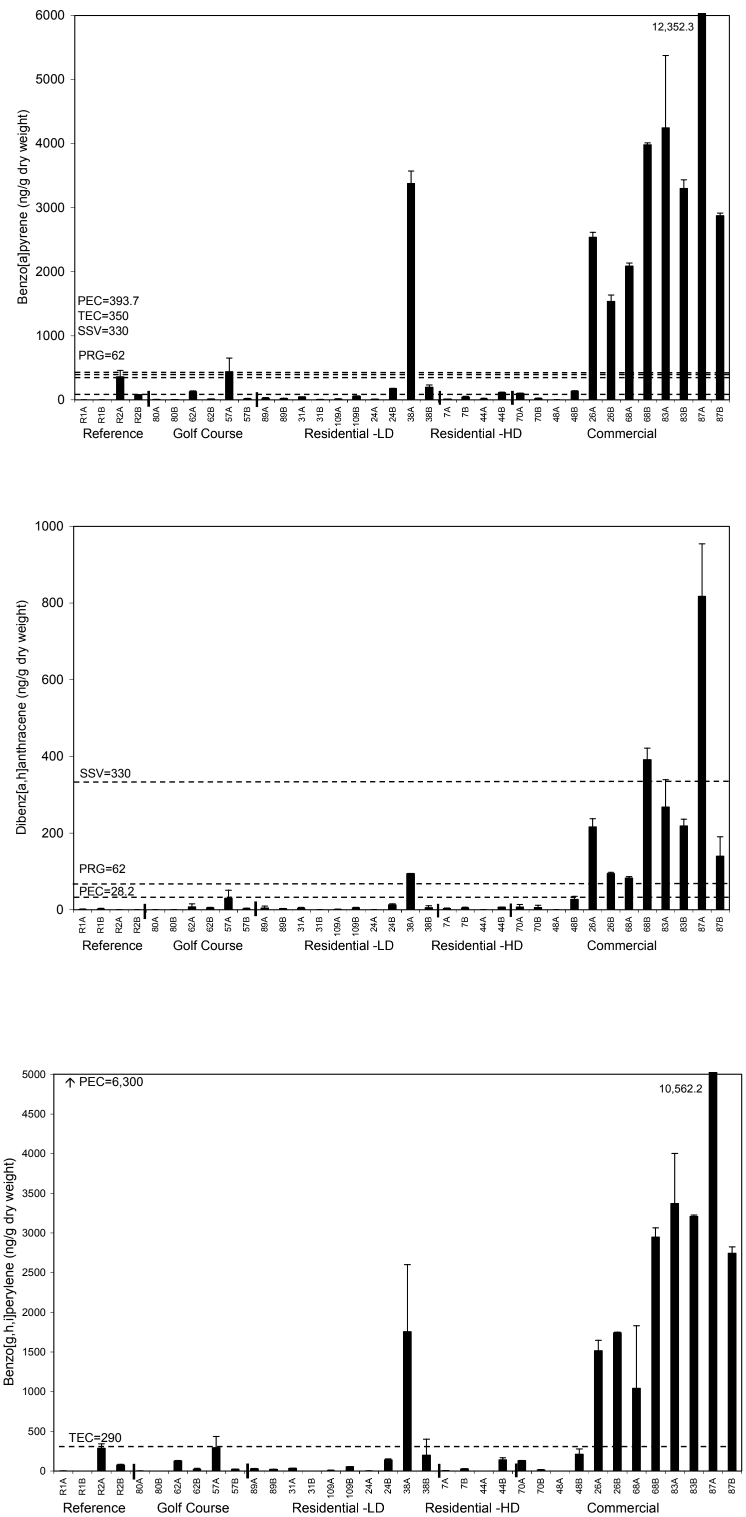


Figure 31. Screening assessment for sediment levels of benzo[a]pyrene, dibenz[a,h]anthracene and benzo[g,h,i]perylene from the two reference ponds and 16 stormwater ponds. TEC=threshold effect concentration, PEC=probable effect concentration, SSV=sediment screening value, PRG=preliminary remediation goal. A=pond inlet, B=pond center. Error bars = +1 SE.

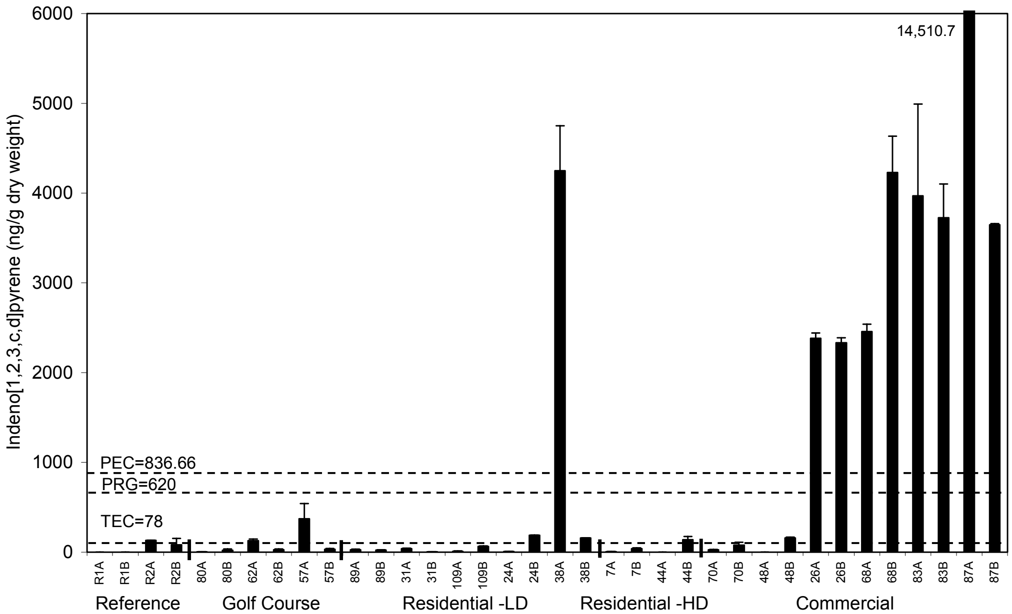


Figure 32. Screening assessment for sediment levels of indeno[1,2,3,c,d]pyrene from the two reference ponds and 16 stormwater ponds. TEC=threshold effect concentration, PEC=probable effect concentration, SSV=sediment screening value, PRG=preliminary remediation goal. A=pond inlet, B=pond center. Error bars = +1 SE.

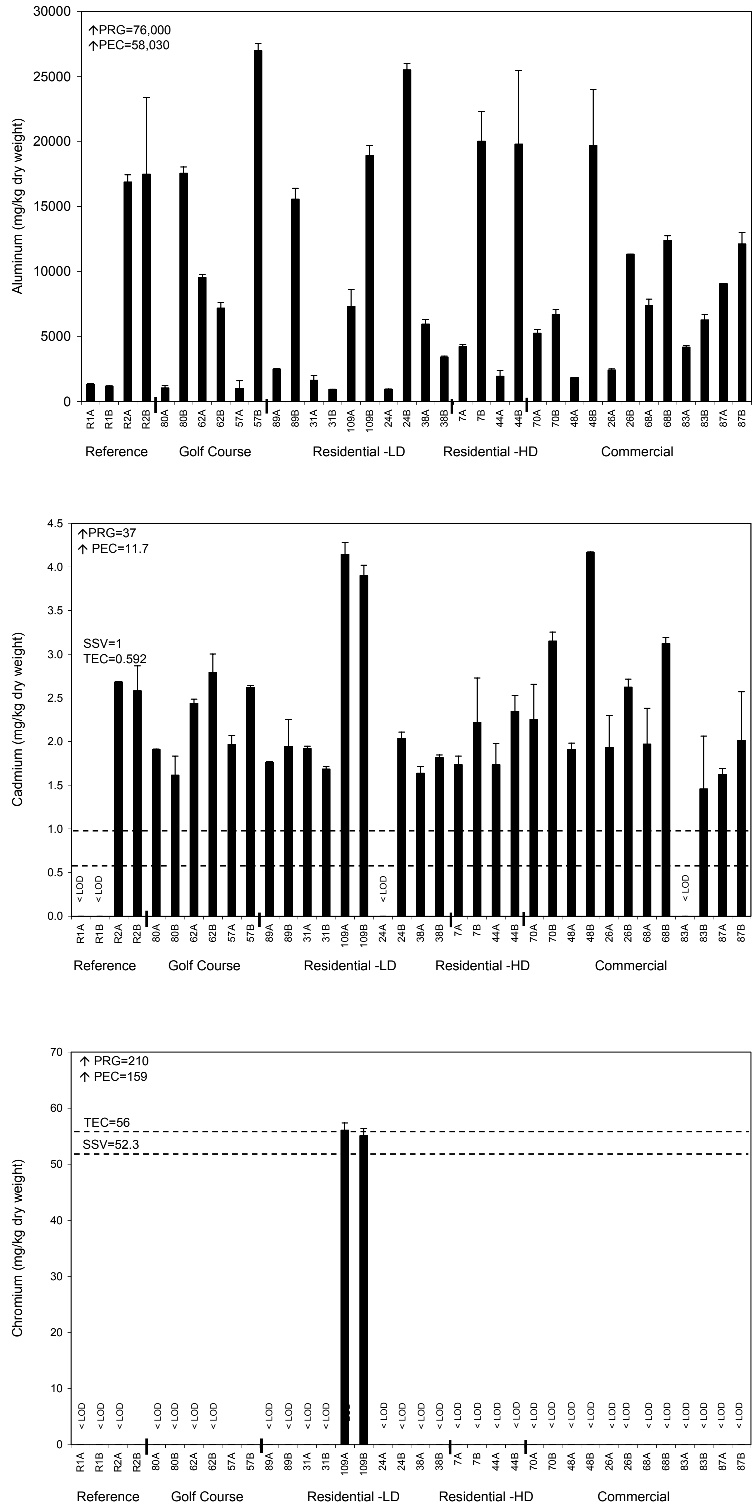


Figure 33. Screening assessment for sediment levels of aluminum, cadmium and chromium from the two reference ponds and 16 stormwater ponds. TEC=threshold effect concentration, PEC=probable effect concentration, SSV=sediment screening value, PRG=preliminary remediation goal. A=pond inlet, B=pond center. Error bars = +1 SE.

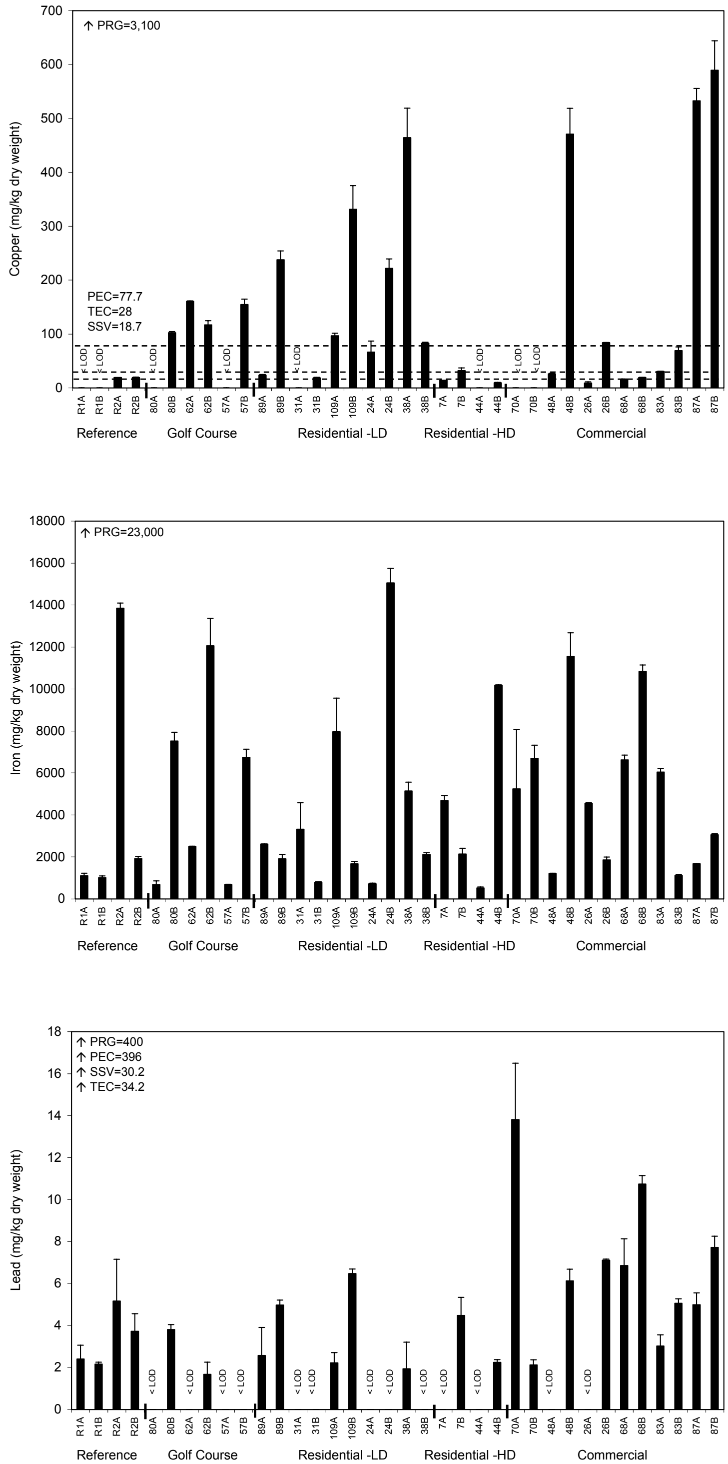


Figure 34. Screening assessment for sediment levels of copper, iron, and lead from the two reference ponds and 16 stormwater ponds. TEC=threshold effect concentration, PEC=probable effect concentration, SSV=sediment screening value, PRG=preliminary remediation goal. A=pond inlet, B=pond center. Error bars = +1 SE.

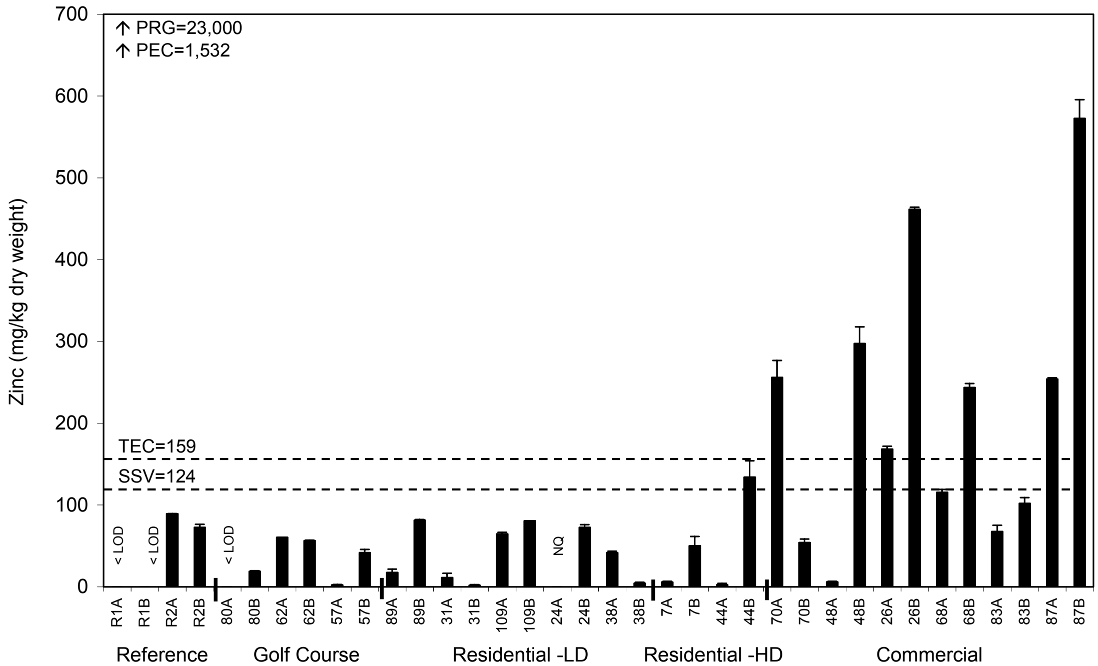


Figure 35. Screening assessment for sediment levels of zinc from the two reference ponds and 16 stormwater ponds. TEC=threshold effect concentration, PEC=probable effect concentration, SSV=sediment screening value, PRG=preliminary remediation goal. A=pond inlet, B=pond center. Error bars = +1 SE.

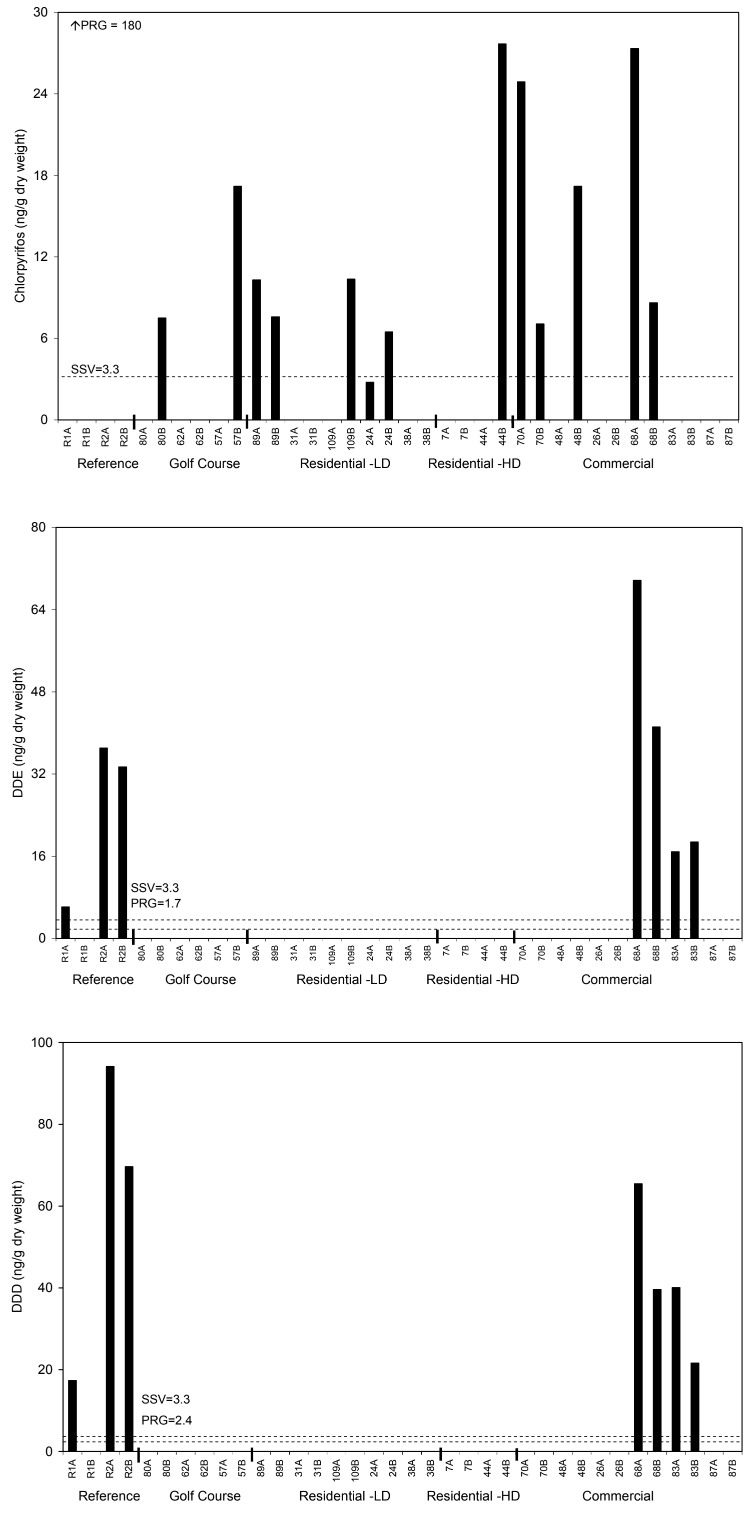


Figure 36. Screening assessment of chlorpyrifos, DDE, and DDD sediment levels from the two reference ponds and 16 stormwater ponds. TEC=threshold effect concentration, PEC=probable effect concentration, SSV=sediment screening value, PRG=preliminary remediation goal. A=pond inlet, B=pond center.

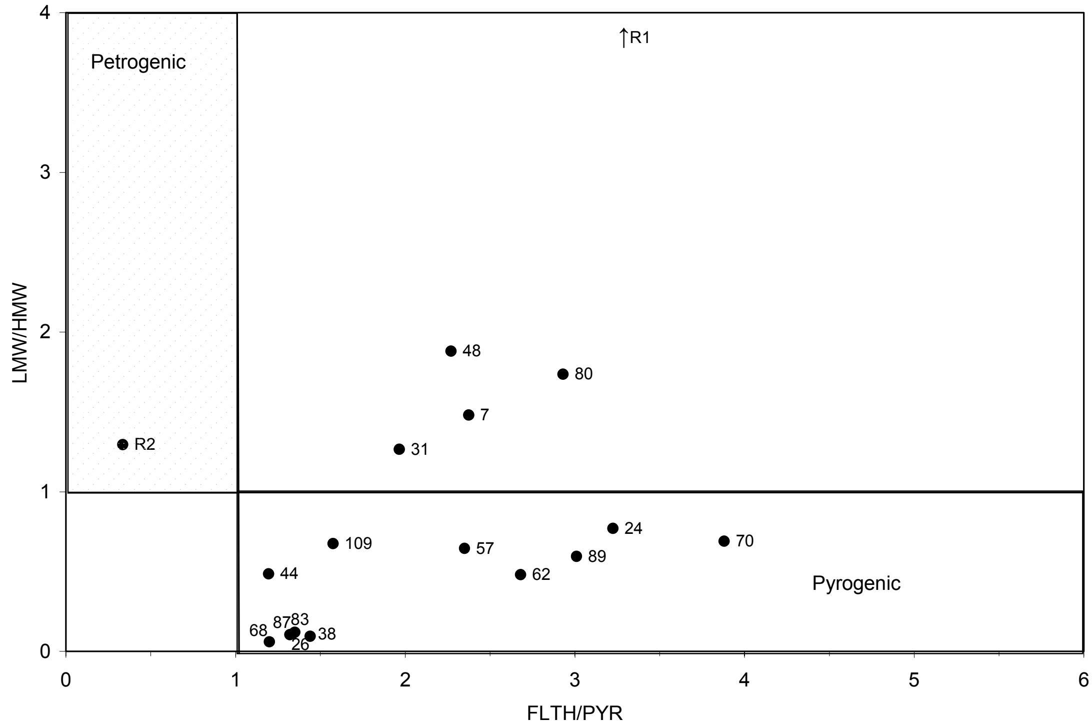


Figure 37. Plot of LMW/HMW ratio to that of FLTH/PYR ratio.

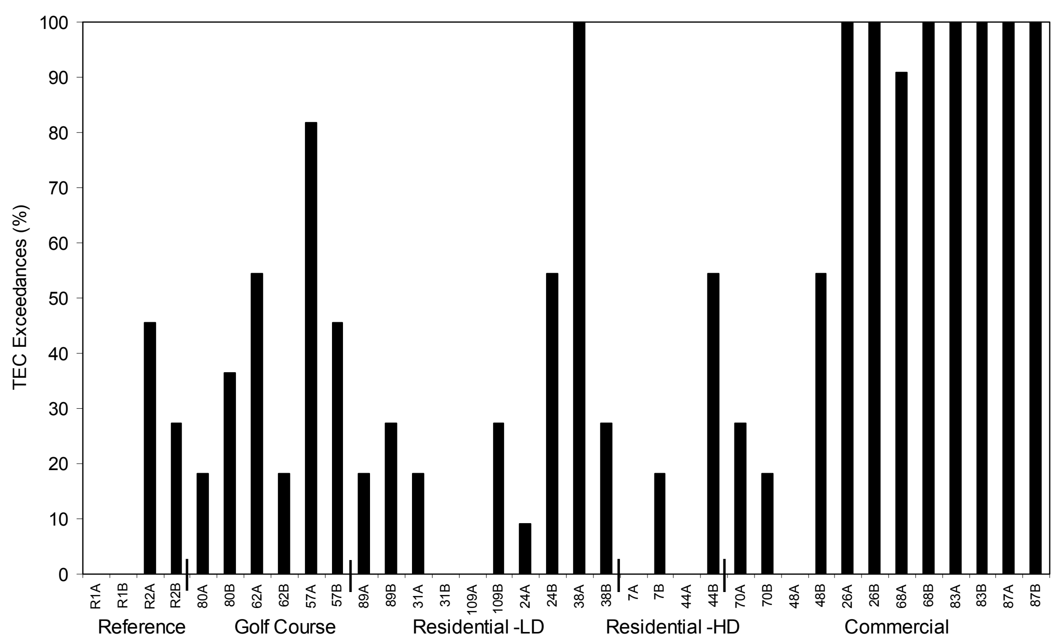


Figure 38. Summary percent exceedances of PAH TEC values in sediments from the reference and stormwater ponds.

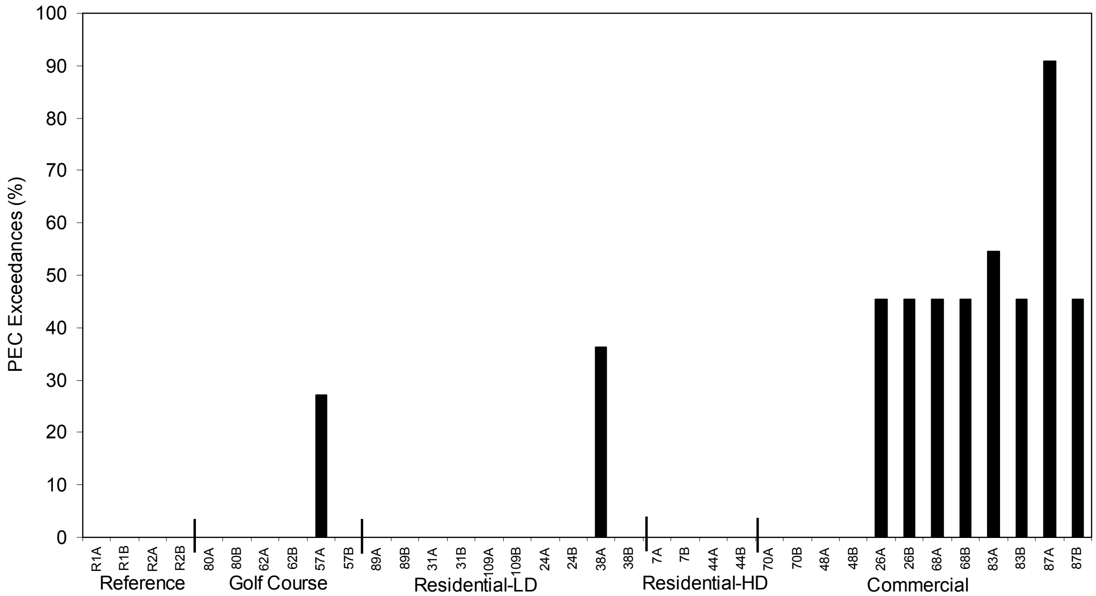


Figure 39. Summary percent exceedances of PAH PEC values in sediments from the reference and stormwater ponds.

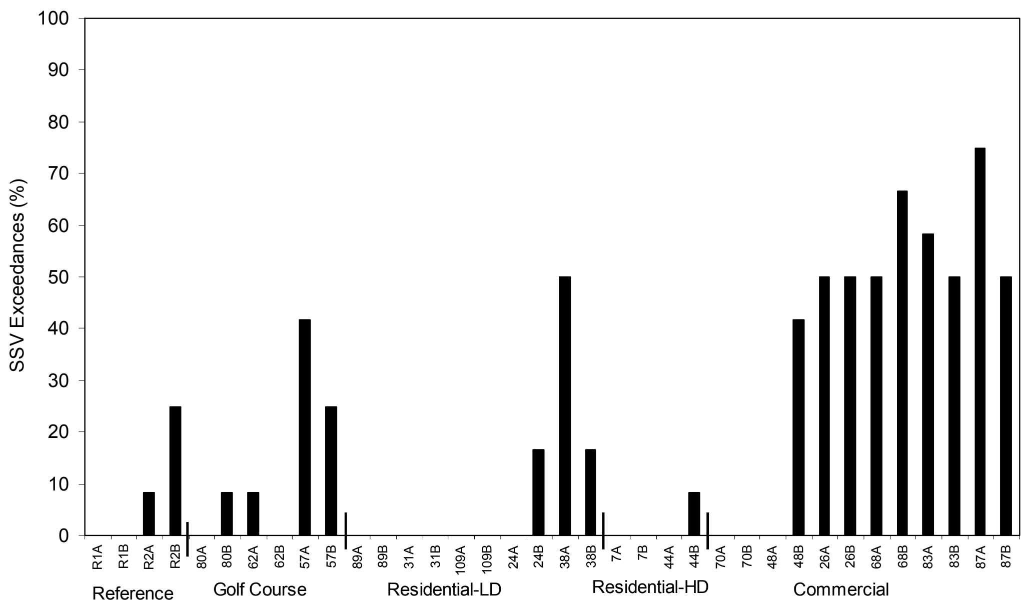


Figure 40. Summary percent exceedances of PAH SSV values in sediments from the reference and stormwater ponds.

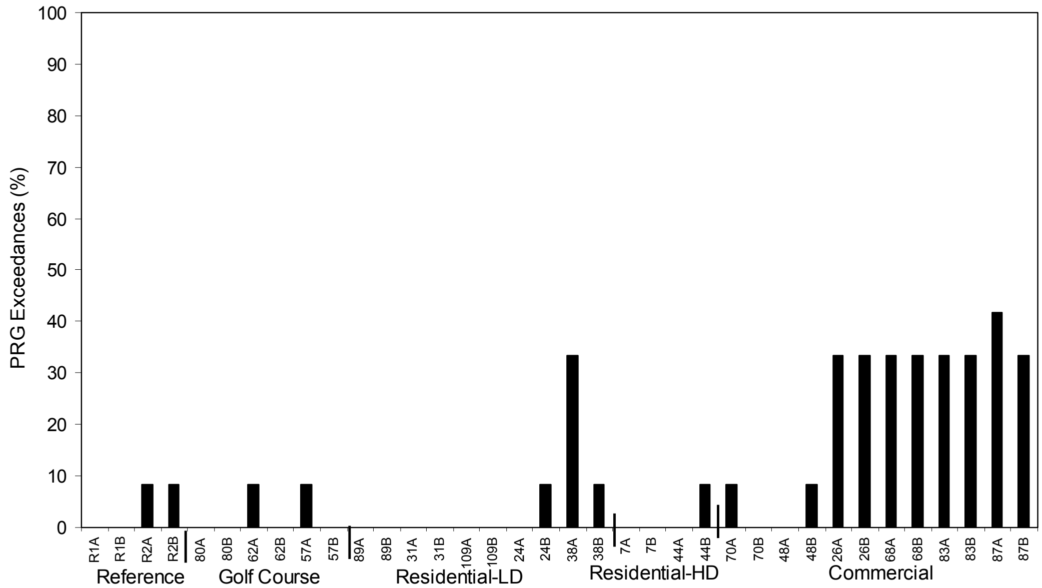


Figure 41. Summary percent exceedances of PAH PRG values in sediments from the reference and stormwater ponds.

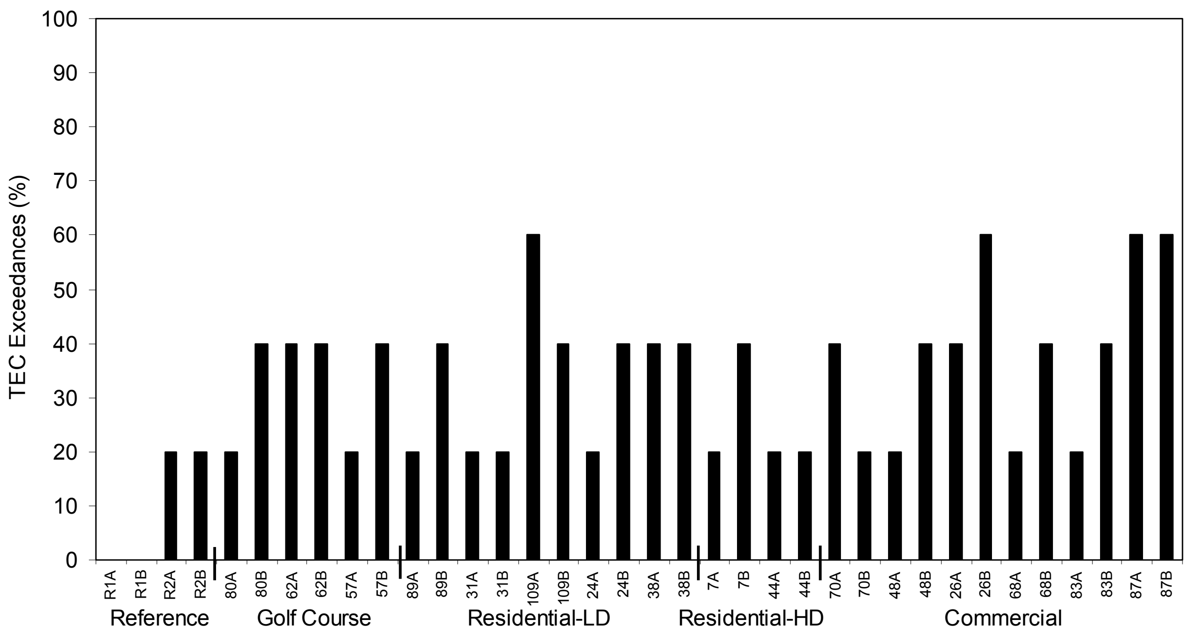


Figure 42. Summary percent exceedances of metal TEC values in sediments from the reference and stormwater ponds.

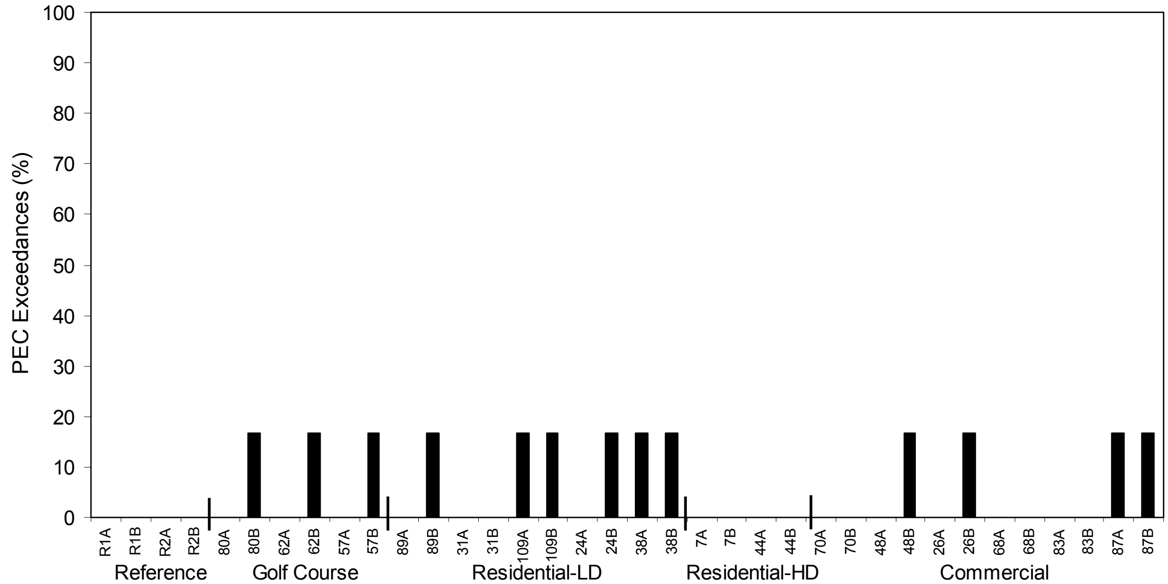


Figure 43. Summary percent exceedances of metal PEC values in sediments from the reference and stormwater ponds.

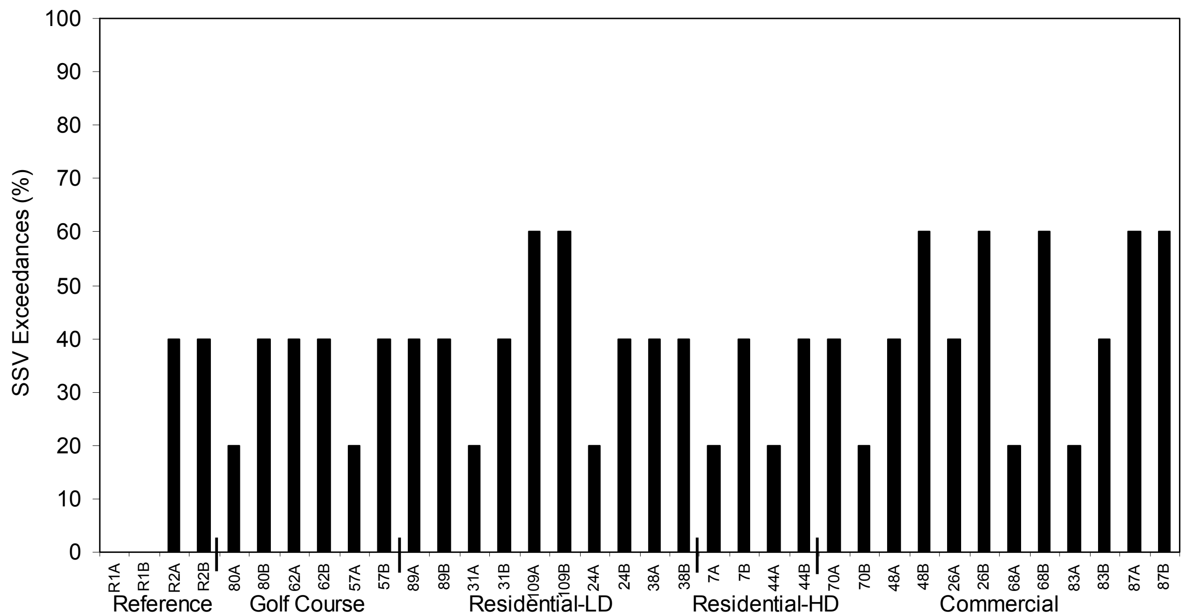


Figure 44. Summary percent exceedances of metal SSV values in sediments from the reference and stormwater ponds.

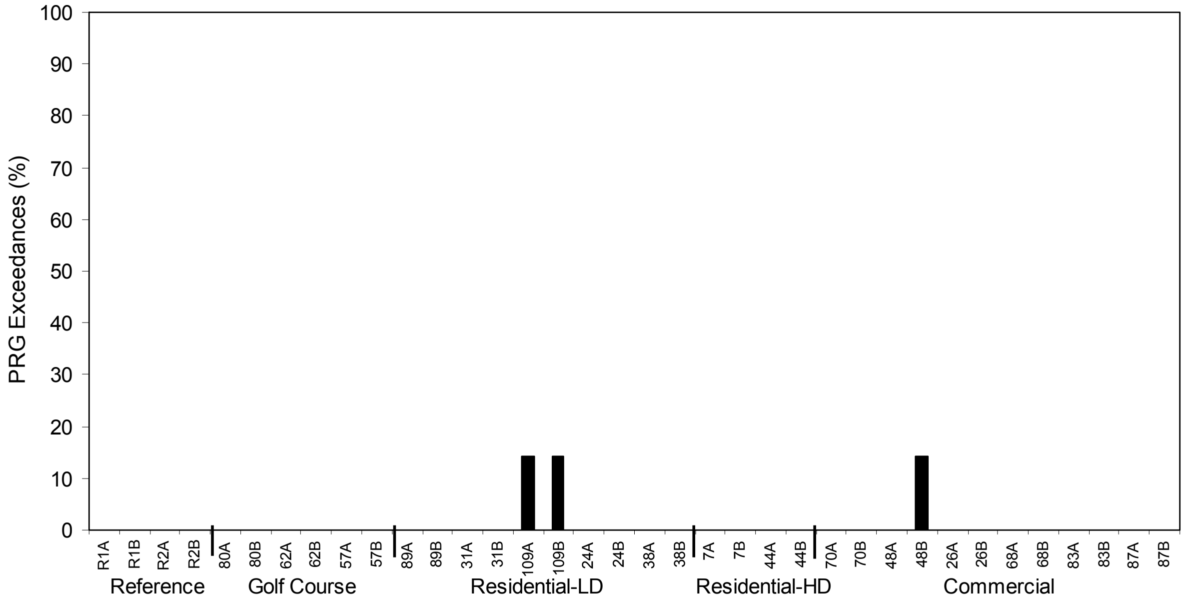


Figure 45. Summary percent exceedances of metal PRG values in sediments from the reference and stormwater ponds.