2014

The Effects of Aeration on Phytoplankton Community Composition and Primary Production in Stormwater Detention Ponds near Myrtle Beach, SC

Lauren Hehman
University of South Carolina - Columbia

THE EFFECTS OF AERATION ON PHYTOPLANKTON COMMUNITY COMPOSITION AND PRIMARY PRODUCTION IN STORMWATER DETENTION PONDS NEAR MYRTLE BEACH, SC

by

Lauren M. Hehman

Bachelor of Science
University of South Carolina, 2012

Submitted in Partial Fulfillment of the Requirements

For the Degree of Master of Science in

Marine Science

College of Arts and Sciences

University of South Carolina

2014

Accepted by:

Tammi Richardson, Director of Thesis

Erik Smith, Director of Thesis

James Pinckney, Reader

Lacy Ford, Vice Provost and Dean of Graduate Studies

DEDICATION

To my mom and Sam, without your encouragement, support, and love I would have not have perused my dreams and to my friends for always finding ways to make me smile and never letting me give up.

ACKNOWLEDGEMENTS

I would like to thank my co-advisors and committee members. Erik Smith, thank you for all of your advice, patience, and guidance in the lab and field. I truly would not have the drive to reach my full potential and be the researcher that I am today without your help. Tammi Richardson, thank you for being a wonderful and enthusiastic teacher. Without your love and support I would not have the amazing opportunities and passion for science as I do now. Thank you Jay Pinckney for your knowledge and willingness to help with HPLC and statistics as well as for the opportunities in your lab that became valuable to this research.

Thank you Ashley Riggs for teaching me, guiding me, and being my role model and Michelle Evans for being my caring support and encouragement. You both were my company every step of the way and without all of your hard work and determination as well, this research would not have been possible.

Thank you to Belle W. Baruch Institute for Marine and Coastal Sciences for lab support and to Angie Defore, Susan Denham, Tracy Buck, Amy Willman, Cathy Ngyuen and Cody Smith (REU student) for all of your dedication and hard work in the field and lab. Also, thanks to fellow graduate students Bridget Bachman and Doug Bell for all of your advice, laughter, and motivation.

ABSTRACT

Stormwater detention ponds are a common approach to managing stormwater runoff in coastal South Carolina. While effective at preventing flooding, they can be sites of intense phytoplankton blooms that result from excess nutrients inputs. This can lead to water quality degradation within ponds from oxygen depletion, fish kills, health concerns from harmful algal blooms (HABs), and water quality deterioration in adjacent coastal waters when pond discharges. The most common management practice to rid the ponds of excess phytoplankton biomass is the addition of copper-based algaecides. While temporarily effective, these algaecides require regular re-application and lead to artificial cycles of productivity and decomposition within the ponds. The installation of water column aerators (fountains or bubblers) have been proposed as an alternative to control phytoplankton biomass by breaking down thermal stratification and oxygenating the bottom waters which could promote benthic nutrient sequestration and enhance denitrification. The use and effectiveness of aerators has never been researched in the shallow ponds typical of coastal South Carolina.

For this thesis, a Before-After-Control-Impact (BACI) experimental design was used to quantify the effects of aeration on water quality in stormwater ponds near Myrtle Beach, SC. Sets of two ponds in two residential developments were sampled for nutrients, phytoplankton biomass, community composition, and rates of primary productivity over two years. Aerators were installed in one pond of each pair after the first year. We hypothesized that aeration would 1) increase bottom water dissolved

oxygen concentrations and thereby increase nutrient retention in sediments and enhance nitrification leading to reduction in water column nutrients and phytoplankton biomass; and 2) increased mixing would shift phytoplankton community composition from potentially harmful cyanobacteria and dinoflagellates towards a community dominated by diatoms. While aeration did enhance mixing and bottom water dissolved oxygen concentrations, overall it had no significant effect on nutrient concentrations, rates of primary productivity, phytoplankton biomass or significantly alter phytoplankton community composition. This research has important implications for developing best management practices and improving coastal water quality in South Carolina as all of these ponds flow directly or indirectly into the coastal zone and are thus potential sources of both chemical and biological pollutants.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1: INTRODUCTION

The coastal region of South Carolina has some of the highest rates of urbanization in the country and the impacts of this development are of great environmental concern (Van Dolah *et al.,* 2008). One potential environmental hazard is stormwater runoff from developed land. The most common approach for controlling runoff is the construction of stormwater detention ponds. These ponds serve primarily to contain runoff for short time periods in order to reduce peak flow (Zhu *et al.,* 2004) and can be effective in minimizing local flooding (Drescher *et al.,* 2007, DeLorenzo *et al.,* 2012) before the water is discharged into a tidal creek or an estuary.

Detention ponds are the most frequently used stormwater management practice in coastal South Carolina (Drescher *et al.,* 2007). The ponds can be found in residential developments and golf courses and can have associated benefits such as acting as a reservoir for pollutants and nutrients that are mobilized in stormwater runoff thus providing time for breakdown of contaminants into products less harmful (Mallin and Wheeler, 2000; Lewitus *et al*., 2003). Homeowners can also use these ponds to improve landscape aesthetics and lead to an increase in property values.

Ponds associated with urbanization can often have high amounts of impervious infrastructure around them resulting in high rates of nutrient loading to the ponds. Typically in South Carolina, the stormwater is directed into the pond through a drainpipe. The lack of a vegetated buffer zone around the ponds exacerbates nutrient loading and results in increased eutrophication, oxygen depletion, bacterial contamination, and fish

kills (Lewitus *et al.*, 2008; Serrano and DeLorenzo, 2008; DeLorenzo and Fulton, 2009). High concentrations of nutrients can lead to excessive phytoplankton growth rates (e.g. Glibert *et al.*, 2001) and could lead to the ponds acting as natural "incubators" for harmful algal blooms (HABs) (Lewitus *et al.*, 2003). Freshwater ponds are particularly susceptible to cyanobacterial blooms that can be toxic, and are always associated with poor water quality and loss of aesthetic value (Paerl, 1988).

Reducing phytoplankton biomass in stormwater ponds can be achieved through various physical, biological, and chemical mitigation strategies. These include vegetative buffers (for removal of nutrients before the water flows into the pond), nutrient inactivation, or bio-manipulation of the food web by introduction of herbivorous fish (Cooke and Kennedy, 2001). The implementation of these strategies in coastal South Carolina stormwater ponds is difficult to achieve since residential pond management practices are often driven by the need for landscape aesthetics. One of the most common management practices to rid ponds of excessive biomass accumulation is through the use of algaecides, like copper sulfate ($CuSO_4$) a relatively inexpensive chemical that can be found at almost any garden supply stores. Algaecide treatments temporarily rid the pond of phytoplankton however in almost all cases phytoplankton re-growth occurs (Le Jeune *et al.*, 2006) and repeated treatment is needed and thus can be expensive. Also, even very low concentrations of copper sulfate will kill the zooplankton (grazers), leading to an even greater 'rebound' of phytoplankton biomass once the chemical leaves the system (Cooke and Kennedy, 2001). The use of copper-based algaecides can also alter community structure (Le Jeune *et al.*, 2006) and can create an artificial cycle of productivity vs. decomposition which can exacerbate low DO impairment further

downstream by stimulating bacterial activity within the ponds. Cyanobacteria have also been shown to be sensitive to copper-sulfate treatments over other members of the phytoplankton community (e.g. Padovesi-Fonseca and Philomeno, 2004) but treatments on cyanobacteria, like *Anabaena sp.*, demonstrated by Gibson (1972) could also lead to benthic fauna stress and food web alterations with heavy copper concentrations settling into the sediments. Copper has been found to be the contaminant of highest concern in South Carolina stormwater pond sediments (Crawford *et al.*, 2010).

Artificial aeration has been proposed by water resource managers as an alternative to the use of toxic algaecides. They have a long history of being used to improve water quality in drinking water sources, lakes, and reservoirs (e.g. Beutel and Horne, 1999; Cooke and Kennedy, 2001). Aerators can also be implemented in a pond through various designs (Boyd, 1997) such as "bubblers or fountains". Diffusion aerators work to induce mixing of the entire water column by bringing deep, low oxygenated waters up to the surface and allowing for a diffusion of oxygen from the atmosphere into the water before the water flows back into the benthos. Circulation of the water column increases oxygen-transfer efficiency and acts to eliminate stratification/ stagnant waters (Boyd, 1997). Creating an oxic benthos can alter biological and chemical processes within the sediments. It can promote the sequestration of ortho-phosphate in iron-based sediments (Boström *et al.*, 1988), inhibit the release of reduced compounds like iron, (Wetzel, 2001) manganese, and sulfide (Beutel and Horne, 1999), and stimulate nitrification and denitrification processes (Rysgaard *et al.*, 1994). Therefore aeration could lead to a reduction in total water column nutrients. Oxic conditions within the pond can also promote desirable conditions for the biological community such as an increase in suitable

habitat for fish (Müller and Stadelmann, 2004) and migrating zooplankton (Field and Prepas, 1997). It can also lead to shifts in phytoplankton community composition from cyanobacteria (e.g. Jungo *et al.*, 2001) to a more desirable taxon by mixing the water column. Under anoxic conditions at the sediment interface, an increased internal phosphorus loading can occur from release of iron-based sediments further enhancing phytoplankton growth (Wetzel, 2001).

The increased prevalence of cyano-HABs is becoming a major concern both coastally and in freshwater systems (Anderson *et al.*, 2002; Pinckney *et al*., 2001; Smith, 2003). Therefore, there is a need to understand and develop best management strategies to improve coastal water quality.

In this study, I compared water column nutrient concentrations, total phytoplankton biomass, primary production, and community composition before and after the installation of diffusion aeration systems in stormwater detention ponds typical of coastal South Carolina. My goal was to quantitatively evaluate their effectiveness with respect to reducing phytoplankton biomass and primary production. I hypothesized that aeration would 1) increase bottom water DO concentrations, 2) increase nutrient retention in the sediments leading to an overall decrease in water column nutrients and 3) decrease phytoplankton biomass and rates of primary productivity. I also hypothesized that aeration and enhanced mixing would alter phytoplankton community composition by 1) decreasing potentially harmful cyanobacteria and dinoflagellates and 2) increasing the relative and absolute abundance of diatoms. This research has broad-scale implications for improving stormwater management practices in South Carolina, as many of these stormwater ponds flow directly or indirectly into the coastal ocean and thus have serious

implications on coastal water quality as being sources of both chemical and biological pollutants.

CHAPTER 2: METHODS

Study Site

A pair of ponds in each of two residential communities of coastal South Carolina were set up as a before-after control-impact (BACI) experimental design (Smith, 2002) (Fig. 2.1). Two of the ponds (SBA, SBB) were located in South Bay residential community in Surfside Beach, SC (33.36º N, 79.00º W) while the other two (IOA, IOB) were located in the Inlet Oaks residential community in Murrells Inlet, SC (33.31ºN, 79.03º W). The pond pairs were chosen because of their proximity and replication to one another. The ponds in Surfside Beach are about 500 m^2 and 1.5 meters deep while the Inlet Oaks ponds are 3220 m^2 and about 3.1 meters deep respectively. At each location, the two ponds were connected by a series of pipes with the control pond upstream from the impacted pond and outflow of the ponds occurred through a pipe that only released water when the water level was high. Ponds were sampled from April 2012 through December 2013. Sampling occurred bi-weekly during the summer months when ambient temperatures were highest (April-September) and monthly during the winter, (October-March) (Reynolds, 1984).

Aerators were provided by Vertex Water Features (Pompano Beach, FL, USA) and were installed on May 31, 2013 in one in South Bay Ponds (SBA) and one of the Inlet Oaks ponds (IOA) (Fig. 2.2). The aerators were fitted according to each of the ponds volume and surface area specifications to achieve water column turnover at a rate of twice per day. A Custom Mini 1 XL2 with two diffuser pads was installed in SBA and a

PondLyfe 2, which features one diffuser placed on opposite sides of the pond was installed in IOA.

Sampling

From all ponds, triplicate, independent samples of surface water were collected using pre-acid washed (10% HCl) 1-litre amber plastic bottles. Water (~15 litres) was also collected in a pre-acid washed (10% HCl) cubitainer for biological oxygen demand (BOD) measurements (data not shown) and for determination of rates of primary productivity. Each bottle was rinsed with sample water before the final samples were collected. Secchi depth was measured in the middle of each pond, a handheld YSI 650 MDS was used to collect water column profiles of dissolved oxygen (DO), temperature, and conductivity, and a submersible scalar quantum sensor (LI-COR Biosciences, Lincoln, Nebraska, USA) was used to measure water column profiles of photosynthetically available radiation (PAR) at each of the sites. Bottles were placed in a cooler on ice and cubitainers were placed in a dry cooler before being transported back to the Belle W. Baruch Institute for Marine and Coastal Sciences (Georgetown, SC).

One YSI 6600 V2 sonde was placed just below (0.5m) the surface of each of the four ponds. In the South Bay ponds, a YSI or Hydrolab OS5X sonde was also placed about 0.5 meters from the bottom. The sondes continuously recorded DO, temperature, and conductivity. They were removed every three weeks for calibration and maintenance.

Measurements

Nutrients, DOC, Suspended Solids, CHN, CDOM

For CHN concentrations, 100 mL of sample water were poured through a pre-combusted 0.21 GF/F Whatman filter and the collected particulate matter was placed in a drying oven for 48 hours before being placed in the -80°C freezer until analysis. For total nitrogen (TP) and total phosphorus (TP), 5 mL of sample water were placed into pre-acid washed and pre-combusted test tubes for analysis. Sample water for dissolved organic carbon (DOC), total dissolved nitrogen (TDN), total dissolved phosphorus (TDP), colored dissolved organic matter (CDOM), and nutrient concentrations were collected by filtering a sample through a weighed and pre-combusted 0.47 mm GF/F Whatman filter. The particulate matter collected on the filter was placed in a drying oven for 48 hours and used to calculate total suspended solids, organic suspended solids, and inorganic suspended solids (TSS, OSS, and ISS).

For TDN and TDP, 5 mL of filtrate were placed into pre-acid washed (10% HCl) and pre-combusted test tubes. For DOC, 20 mL of filtrate were placed into a 25 mL pre-combusted and pre-acid washed glass vial with screw cap and septa and was acidified with 200 mL of 10% HCl. For CDOM, 20 mL of filtrate were placed into a 20 mL pre-acid washed glass scintillation vials. For nutrients, 30mL were placed into a 30 mL plastic bottle. All of these filtrates were placed in a fridge placed at 5°C until analyzed and nutrient bottles were placed in the -80ºC freezer until analyzed.

Primary Productivity

Rates of photosynthesis vs. irradiance (P-E) were determined from the incorporation of ^{14}C bicarbonate using the small-volume incubation method of Lewis and Smith (1983). For these measurements, I used 30 µL $NaH^{14}CO_3$ (57 µCiµmol^{-1}; MP Biomedicals, Solon, OH, USA) added to 30 mL of sample water for a final activity of 1 µCi mL^{-1}. Three "total count" vials were prepared using 20 µL of sample water in 7 mL scintillation vials with 250 µL of phenethylamine and 5 mL of Ecolume scintillation cocktail (MP Biomedicals, Solon, OH, USA) to determine the total activity of the sample. One mL of the remaining radioactive sample was dispensed into each of 24 vials and placed into a photosynthetron (Lewis and Smith 1983) for 30 minutes at *in situ* temperatures. Irradiance in the photosynthetron ranged from ~1 to 1800 µmoles quanta $m^{-2}s^{-1}$ as measured with a 4π scalar quantum sensor (Biospherical Instruments Inc., San Diego, CA, USA) before and after incubation period to obtain average values of irradiance. At the end of the incubation period, 50 µL of buffered formalin was added to each sample. Samples were then acidified with 200 µL of 50% HCl to remove unincorporated ^{14}C and were placed on a shaker table overnight to complete de-gassing. Five mL of Ecolume was then added and the samples were counted on a Packard Tri-Carb3100TR Liquid Scintillation Analyzer. Disintegrations per minute (dpm) were converted to Chl-specific rates of primary productivity (P^{Chl}, µgC[µgChl^{-1}] h^{-1}) using the equation of Knapp *et al.* (1996).

Dissolved inorganic carbon concentrations (DIC) were measured directly to account for isotope dilution. Samples were filtered through a Whatman sterile syringe filter (0.45 µm) and placed in fridge 5^{o}C overnight. DIC samples were taken out of the

fridge and brought to room temperature while standards were made using Na_2CO_3 Samples of 150 µl were then injected into a 20 mL flask of H_2SO_4 flask and then pushed through a LICOR LI-7000 CO_2/ H_2O analyzer using N_2 gas, and the concentration of CO_3 was measured.

Chlorophyll and biomarker photopigment concentrations were determined by High Performance Liquid Chromatography (HPLC) according to Pinckney *et al.* (1996) (see below). Filters for HPLC were placed in aluminum foil packets and transported on dry ice to the University of South Carolina (Columbia, SC) Estuarine Ecology Lab (EEL) and stored immediately in a -80^{0}C freezer until analyzed.

Analytical Methods

Suspended Solids, Nutrients, CHN, DOC, CDOM

Filters for suspended solids were removed from the drying oven and the amount of particulate matter on the filter was re-weighed to determine TSS. Filters were then combusted at 450 ºC to remove organic material and the filters were then re-weighed to determine ISS. OSS concentrations were determined by the difference.

TN and TP samples were processed using standard methods for the examination of wastewater, with the digestion of whole-water by persulfate oxidation (Glibert *et al.,* 1977) which reduces nitrogen contamination followed by analysis using a Technicon II AutoAnalyzer. Concentrations of ammonium at each of the sites were determined using a modification of a phenylhypochlorite method (Solorzano 1969) and. Soluble reactive phosphorus (SRP), nitrate, and nitrite were all determined using the Technicon II AutoAnalyzer. Dissolved organic nitrogen (DON) and dissolved organic phosphorus

(DOP) were determined by the difference between the total and the inorganic N and P concentrations.

Particulate nitrogen (PN) and particulate organic carbon (POC) concentrations were measured on a Costech CHN elemental analyzer after vapor-phase acidification to remove inorganic carbon (Hedges and Stern, 1984).

DOC concentrations were measured by being injected and pushed through a Shimadzu TOC-V_{CPN} Analyzer using CO_2 gas and combusted (450ºC) using the methods of Benner and Strom (1993).

Colored Dissolved Organic Matter (CDOM) was measured by spectrophotometry using a dual beam Shimadzu UV-VIS 2450 spectrophotometer with a 5-cm quartz glass cuvette and deionized water as a blank. Absorbance was measured at 355 nm and converted into absorbance coefficients according to the equation:

$$a_{CDOM}(\lambda) = \frac{2.3A(\lambda)}{\ell}$$

where $\alpha_{CDOM}(\lambda)$ is the spectral absorbance coefficient (m^{-1}), $A(\lambda)$ is the absorbance at 355 nm, and ℓ is the optical path length of cuvette in meters.

Primary Productivity

PE curves were fitted with a non-linear photosynthesis-irradiance equation (Platt *et al*. 1980) using *JMP 8* software to estimate the chlorophyll-specific initial slope α ($\mu gC[\mu gChl]^{-1}\ h^{-1}[\mu molphoton\ m^{-2}\ s^{-1}]^{-1}$) , the chlorophyll specific maximal rate of photosynthesis P_m^{chl} ($\mu gC[\mu gChl^{-1}]h^{-1}$), and the photoinhibition term β^{chl} ($mgC[\mu g\ chl\alpha^{-1}]h^{-1}[\mu molphoton\ m^{-2}s^{-1}]^{-1}$].

$$P(z) = Chl\alpha * P_m^{chl}\left(1 - e^{\frac{-\alpha E(z)}{P_m^{chl}}}\right) * e^{\frac{-\beta E(z)}{P_m^{chl}}}$$

where E(z), the irradiance at depth, is defined as $E(z) = E_0(t) * e^{-kd*z}$ where E_0 and k_d are the surface irradiances and z is depth. Meteorological data at the NOAA/NERR weather station at Oyster Landing collects continuous measurements of total PAR. These values were used to estimate the daily rate of primary production with depth $P(_z)$, every 0.25 m, as a function of surface irradiance (Platt *et al.* 1980). The rates of total PAR were measured as the total flux averaged over a 15 minute logging interval and used the values from the average of the two days preceding and the day of the sampling event. The averages of gross primary production (GPP) were then integrated from sunrise to sunset using U.S. Naval Observatory sunrise/sunset tables (Washington, DC).

Chlorophyll-α concentrations were determined by filtering 10 mL of sample water onto a 0.21 GF/F Whatman filter and then placing filters in 90% acetone at 4 °C for 24 h. The extracts were analyzed fluorometrically using a Turner Trilogy Fluorometer based on the methods of Welschmyer (1994).

HPLC/ CHEMTAX

Samples for HPLC analysis were lyophilized for 24 h at -50° C, placed in 90% acetone (0.75 ml), sonicated, and extracted at -20° C for 24 h. Filtered extracts (200 µl) were injected into a Shimadzu HPLC equipped with a monomeric (Rainin Microsorb-MV, 0.46 x 10 cm, 3 µm) and a polymeric (Vydac 201TP54, 0.46 x 25 cm, 5 µm) reverse-phase C18 column in series. A nonlinear binary gradient consisting of the solvents 80% methanol: 20% 0.50 M ammonium acetate and 80% methanol: 20% acetone was used for pigment separations (Pinckney *et al*., 1996). Absorption spectra and

chromatograms (440 ± 4 nm) were acquired using a Shimadzu SPD-M10av photodiode array detector. Pigment peaks were identified by comparison of retention times and absorption spectra with pure standards (DHI, Denmark). The synthetic carotenoid β-apo-8'-carotenal (Sigma) was used as an internal standard.

Contributions of individual algal groups to total community composition and to each size class were determined by chemical taxonomy using CHEMTAX software (Mackey *et al*., 1996). The absolute contributions of algal group as the total chlorophyll-*a* (in μgL^{-1}) concentration was derived and used to calculate the relative contributions as the proportion of total chlorophyll-*a* by each group so sum of the contributions at each site equal one. The initial pigment ratios were compiled and evaluated using a freshwater algal matrix proposed by Schlüter *et al.* (2006).

Statistical Analysis

All the nutrient concentrations of were reported in mass units converting $\mu molL^{-1}$ to mgL^{-1} by multiplying the concentrations by the molar ratios then diving by 1000. All statistics were evaluated using *SPSS 21* software and relative CHEMTAX derived phytoplankton community composition values and nutrient concentrations were log transformed with outliers removed. The relationship between log Chl-*a* and log GPP to other variables (e.g. nutrients) were determined using multiple linear regressions. A Randomized Complete Block Analysis of Variance (RCB ANOVA) with period (before, after), location (control, impact) and site as the blocking factor was performed on the differences between the control and the impacted sites to determine significant differences. A multivariate analysis of variance (MANOVA) with the differences

between the control and impacted and period (two levels-before, after) was used to determine the significant differences between the factor and their interactions at each site pair as well. A paired t-test of surface and bottom DO concentrations at each of the sites before and after aeration was also performed to determine a significant difference with aeration. A p-value of 0.05 was used for hypothesis testing.

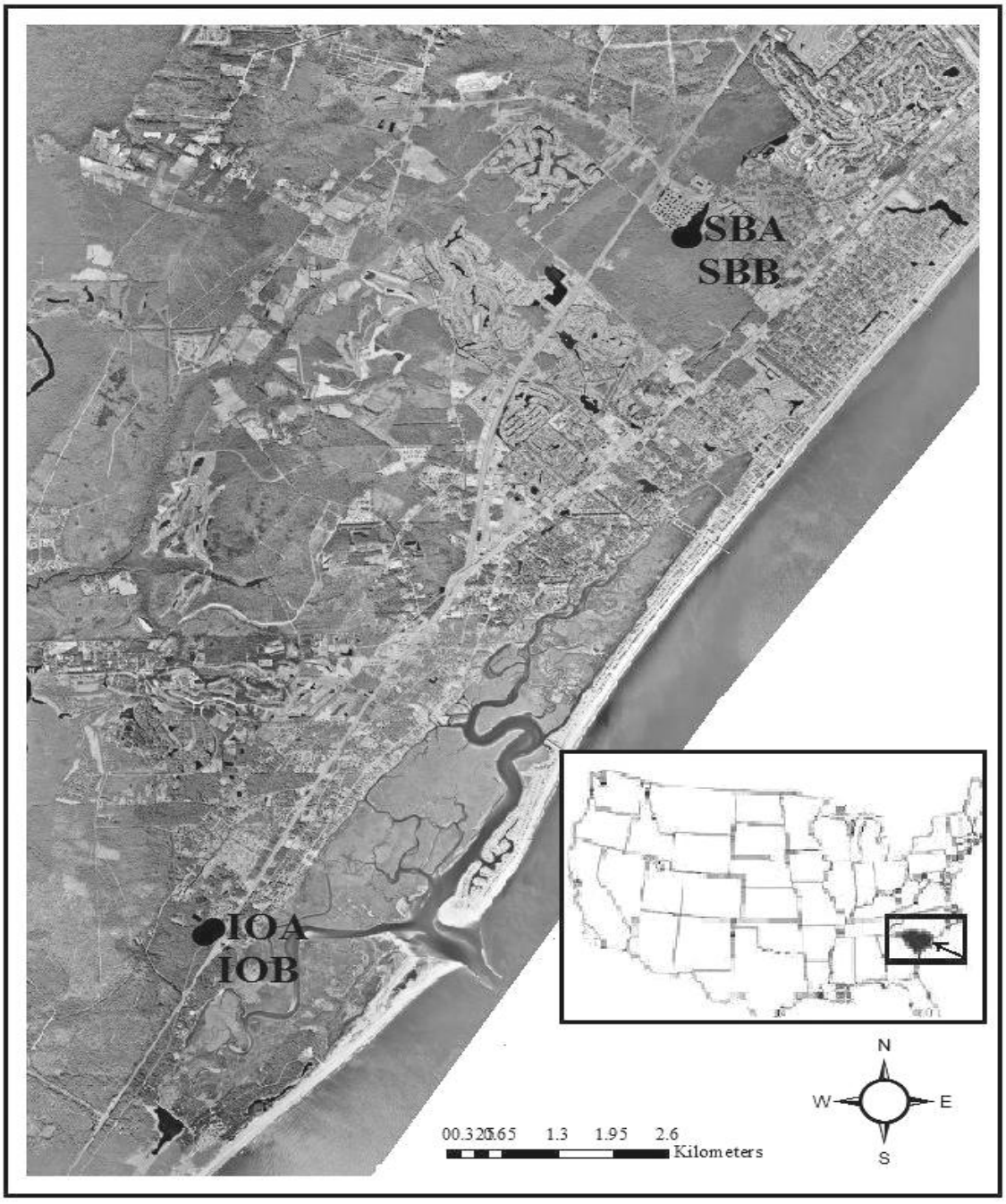

Figure 2.1. Map image showing the location of the study sites**.** Sampling sites are indicated by the circles (SBA, SBB, IOA, IOB). The impacted sites were fitted with aerators March 31, 2013 in SBA and IOA. GIS data layers were provided by SC DNR (https://www.dnr.sc.gov/pls/gisdata/download_data.login).

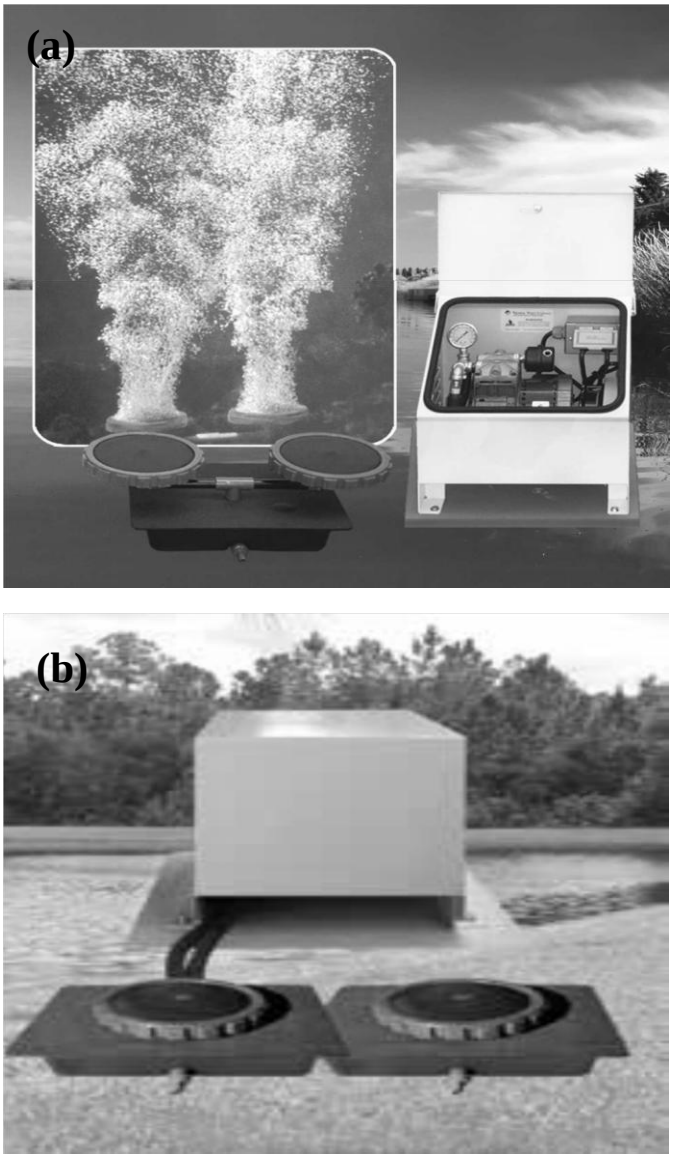

Figure 2.2. Aerators installed in (a) SBA, Custom Mini 1 XL2 and (b) IOA, PondLyfe 2 provided by Vertex Water Features on March 31, 2013.

CHAPTER 3: RESULTS

Nutrient concentrations, phytoplankton biomass, community composition, and rates of primary productivity varied and were significantly different between sites (Pillai's Trace = 0.865, $F_{1,32} = 7.313$, $p = 0.000$, power = 1.000) and period (defined as before and after aeration) (Pillai's Trace = 0.665, $F_{1,32} = 2.264$, $p < 0.05$, power = 0.969), as determined by RCB ANOVA (Table 3.1). However, TSS ($F_{1,59} = 6.873$, $p < 0.05$), OSS ($F_{1,59} = 4.537$, $p < 0.05$), DOP ($F_{1,59} = 4.918$, $p < 0.05$), light attenuation coefficient (kd) ($F_{1,59} = 4.194$, $p < 0.05$), chrysophytes and diatoms ($F_{1,59} = 6.585$, $p < 0.05$), and dinoflagellates ($F_{1,59} = 19.296$, $p < 0.05$) were the only significantly different treatments before and after aeration. All other nutrient concentrations, total phytoplankton biomass and gross primary productivity were not significantly different before and after aeration ($p > 0.05$). A MANOVA of South Bay and Inlet Oaks with period revealed that nutrient concentrations, total phytoplankton biomass, community composition, and gross primary productivity in South Bay (Pillai's Trace = 0.965, $F_{1,29} = 1.985$, $p > 0.05$, power = 0.142) or in Inlet Oaks (Pillai's Trace = 0.969, $F_{1,29} = 2.264$, $p > 0.05$, power = 0.154) were overall not significantly different (Table 3.2, 3.3).

Physical

DO concentrations remained high in the surface of the water column at each of the sites but then decreased during the summer month which corresponded to the

highest water temperatures (between 25- 30º C) (Fig. 3.1). During the first year of study, in all of the ponds, thermal stratification occurred. Bottom water DO concentrations were significantly different and lower (Fig. 3.2) than at the surface in all of the sites, SBA ($t=3.516$, $df=28$, $p < 0.01$), SBB ($t=4.356$, $df=28$, $p < 0.01$), IOA ($t=5.762$, $df=28$, $p < 0.01$), and IOB ($t=6.631$, $df=28$, $p < 0.01$) as demonstrated with a paired t-test. DO concentrations often reached points of hypoxia (< 2 mgL^{-1}) during 25- 30º C (summer) temperatures (Fig. 3.1). During the winter months (November through January), DO concentrations in the bottom water rose and matched those of the surface except in IOB (Fig. 3.1d) where concentrations remained low. During March and April 2013, DO concentrations in the bottom water began to decrease from surface waters as temperatures warmed and the ponds thermally stratified.

Once aerators were installed at SBA and IOA (Fig. 3.1, a and c), bottom water DO concentrations increased. Do concentrations remained close to those of the surface water even when temperatures were high (25- 30º C) and thermal stratification was broken down. As seen in a paired t-test, bottom water DO concentrations with aeration were not significantly different than surface concentrations in SBA ($t=1.969$, $df=33$, $p > 0.05$). Concentrations were significantly different in IOA ($t=7.520$, $df=33$, $p < 0.01$) but were much higher than in 2012 (Fig. 3.2). In SBB and IOB ponds, the bottom water DO remained low during summer of 2013 (Fig. 3.1, b and d) and often reached critical levels. DO concentrations in surface and bottom water were significantly different in SBB ($t=7.709$, $df=33$, $p < 0.01$) and IOB ($t=12.776$, $df=33$, $p < 0.01$) and remained very low in bottom waters (Fig. 3.2).

Suspended Solids, Nutrients, CHN, CDOM

In South Bay (Table 3.2), there was a significant effect before and after aeration for TN ($F_{1,29} = 2.248$, $p < 0.05$), TDN ($F_{1,29} = 0.016$, $p < 0.05$), DOP ($F_{1,29} = 8.081$, $p < 0.05$), and CDOM ($F_{1,29} = 0.084$, $p < 0.05$). All other nutrient concentrations, total phytoplankton biomass and productivity were not significantly different with aeration ($p > 0.05$). In Inlet Oaks (Table 3.3), there was a significant effect before and after aeration with TSS ($F_{1,29} = 14.238$, $p < 0.05$), OSS ($F_{1,29} = 9.226$, $p < 0.05$), ISS ($F_{1,29} = 8.797$, $p < 0.05$), PN ($F_{1,29} = 7.550$, $p < 0.05$), POC ($F_{1,29} = 8.756$, $p < 0.05$), kd ($F_{1,29} = 4.353$, $p < 0.05$). All other values of nutrient concentrations did not vary significantly vary between before and after aeration ($p > 0.05$).

TSS, OSS, and ISS remained high during the summer months (June through September) and low during the winter months (November through February) (Fig. 3.3). OSS made up the majority of TSS values except in IOB during 2012 when values of TSS were primarily ISS. Concentrations of TSS, OSS, and ISS remained relatively the same throughout the study in SBA (Fig. 3.3) and even with SBB had a high peak during 2012 in TSS and OSS (Fig. 3.3, a and b) the means between the control and impacted site remained similar with aeration (Table 3.2). However, with aeration in Inlet Oaks, IOA increased in values of TSS, OSS, and ISS (Fig. 3.3) while IOB remained similar throughout the study and mean concentrations were different (Table 3.3).

SBB had high TN (> 3.5 mgL^{-1}) and TDN (> 1 mgL^{-1}) (Fig. 3.4) concentrations during summer 2012, however during 2013, concentrations were much lower. SBA had relatively constant concentrations of TN and TDN throughout the study ranging between 0.4 - 1 mgL^{-1}. The South Bay sites had higher concentrations of TN and TDN during

2012 because of high values in SBB than the mean concentrations in 2013 with aeration (Table 3.2). The Inlet Oaks sites remained more variable in both sites with TN ranging from 0.5 - 1.3 mgL^{-1} during 2012 and 2013, except in July 2013 when IOA that had the highest value of TN in ~1.5 mgL^{-1} (Fig. 3.4a). TDN in Inlet Oaks during 2012 ranged from 0.25 – 1.1 mgL^{-1} but in 2013 only ranged between 0.25 - 0.75 mgL^{-1} (Fig. 3.4b). IOA had the highest concentration of TDN in October 2012 (~ 1.1 mgL^{-1}). Mean concentrations of TN and TDN in the Inlet Oaks sites remained similar before and after aeration (Table 3.3).

NH_4^+ concentrations (Fig. 3.5a) ranged from 0.002 - 0.9 mgL^{-1} in all of the sites in 2012 but only from 0.002 - 0.2 mgL^{-1} in 2013 after installation of the aerator except in IOA. IOA had NH_4^+ concentrations ~ 0.3 mgL^{-1} in October 2013. NO_x values ranged from about 0 - 0.25 mgL^{-1} and were typically higher during the colder months (October through February) and were the highest overall in SBA and IOA throughout most of the study. NO_3^- (Fig. 3.5b) concentrations composed the majority of the NO_x speciation. Highest values of NO_3^- were in IOA and had the highest concentration in August 2012 (~0.27 mgL^{-1}) during August. During August 2013 however SBA had the highest concentration (~0.26 mgL^{-1}). The NO_2^- concentrations were highly variable ranging from 0.0003 - 0.025 mgL^{-1} during both years (Fig. 3.5c) with SBA and IOA being the highest concentrations in both years. IOB concentrations of NH_4^+ (< 0.2 mgL^{-1}) and NO_3^- (< 0.05 mgL^{-1}) remained close to minimum detection limits and NO_2^- (< 0.01 mgL^{-1}) remained low throughout the study (Fig. 3.5).

DIN concentrations ranged from ~0.01 - 1 mgL^{-1} (Fig. 3.6a) with highest values occurring in SBA and IOA in both years. Values remained fairly low in IOB during the

study and never > 0.2 mgL^{-1}. DIN concentrations remained the highest during winter months (October through February) but overall lower in 2013. Contributions of DON values were typically similar during both years of study (Fig. 3.6b) but SBB had the highest values of DON (> 0.6 mgL^{-1}) which occurred in summer 2012 and the lowest values of occurring in IOA during winter 2012 (< 0 mgL^{-1}) because TDN values were less than measured DIN.

TP concentrations ranged from 0.01 - 0.13 mgL^{-1} in 2012, but in 2013 ranged between 0.02 - 1.30 mgL^{-1} within the sites (Fig. 3.7a). SBA TP concentrations remained relatively constant (~0.05 mgL^{-1}) throughout the study. SBB TP concentrations remained below 0.15 mgL^{-1} decreased in overall in 2013 and never exceeding ~0.05 mgL^{-1}. Inlet Oaks sites had a higher influx of TP during 2013 with concentrations often reaching > 0.8 mgL^{-1} during April in IOB and both sites in August. TDP remained relatively constant in SBA and IOA ranging between 0 - 0.025 mgL^{-1} but IOA had higher values during 2013 (Fig. 3.7b). SBB had a large influx of dissolved TP (~0.58 mgL^{-1}) during late August 2012 but remained lower throughout 2013. IOB had peaks occurring in October 2012 and August 2013 at ~0.045 mgL^{-1}. The mean differences between TP and TDP were similar before and after aeration at South Bay and Inlet Oaks (Table 3.2, 3.3).

SRP concentrations were the most stable in SBA and ranging from ~0.001 to 0.02 mgL^{-1} (Fig. 3.8a) in 2012 except for a peak (> 0.02 mgL^{-1}) occurring in August but values were lower with aeration (< 0.01 mgL^{-1}). SBB had concentrations (< 0.01 mgL^{-1}) in both years except in July 2012 (>0.02 mgL^{-1}) and March 2013 (> 0.03 mgL^{-1}). In IOA concentrations were low in 2012 (< 0.01) but with aeration had peaks > 0.02 mgL^{-1} but remained < 0.015 mgL^{-1} during the remainder of 2013. IOB had the highest values during

(> 0.05 mgL^{-1}) in 2012 but lower in 2013 never exceeding 0.02 mgL^{-1}. DOP ranged from ~0.001 to 0.02 mgL^{-1} (Fig. 3.8b) except in SBB where values > 0.02 mgL^{-1} in August and October 2012. The SBB DOP concentrations were similar to TDP concentrations (Fig. 3.7b) and highest peaks occurring in August 2012 but were much lower in 2013. Other sites SBA, IOA and IOB had values that remained relatively similar between 2012 and 2013 (< 0.02 mgL^{-1}).

During both years of the study, PN remained relatively constant and < ~0.06 mgL^{-1} (Fig. 3.9a) in all of the sites except in SBB and IOA. SBB has the highest valued occurring during summer 2012 (> 1.2 mgL^{-1}) but not > 0.7 mgL^{-1} in 2013. IOA, with aeration, had higher values in 2013 (> 0.9 mgL^{-1}) but not in 2012. Similar trends were seen with POC concentrations (Fig. 3.9b) in the water column. SBA remained relatively constant during both years even with aeration. Values were the highest (> 0.6 mgL^{-1}) in SBB during summer 2012 but all were < 6 mgL^{-1}during 2013. However, with aeration IOA had the highest values overall with values > 6 mgL^{-1}.

DOC concentrations (Fig. 3.10a) ranged between 5 - 17 mgL^{-1}. Trends for DOC were similar to CDOM concentrations (Fig. 3.10b) ranging between 4 - 27 mgL^{-1} but typically the highest in IOB and the lowest in IOA. In South Bay, concentrations of DOC and CDOM were more variable in 2012 between sites but with aeration SBA typically had higher values than SBB (Fig. 3.10). In Inlet Oaks, DOC and CDOM concentrations were greater in IOB than IOA except in December 2012 and February 2013 where concentrations of IOA increased and IOB decreased. Light attenuation coefficient (k_d) varied between 2012 and 2013 within sites (Table 3.2). The highest coefficient values occur during the summer (June through August) and the lowest values occurred during

the winter (November through February). The highest k_d values occurred in SBB, during summer 2012, when Secchi depth was about 0.2 - 0.3 m and in IOA when Secchi depth was about 0.4 m.

Total primary production, biomass, and Community Composition

Chlorophyll-*a* concentrations (Fig. 3.11a) were highest during summer (June through August) and low during winter months (November through March). In South Bay, chl-*a* mean differences were similar before and after aeration (Table 3.2) but there was a reduction in SBB during 2013 when concentrations remained < 70 μgL^{-1} (Fig. 3.11a). Concentrations in SBA were typically lower than SBB but were higher (~60 μgL^{-1}) in June 2013. In Inlet Oaks, values were low (< 60 μgL^{-1}) in IOA during 2012 but were higher > 60 μgL^{-1} with installation of the aerator for most of summer 2013. There were high concentrations of chl-*a* (> 120 μgL^{-1}) that occurred in September 2012 and August 2013 in IOB.

Mean differences of GPP before and after aeration in Inlet Oaks were different (Table 3.3) and IOA had higher productivity with aeration. From a MANOVA, Inlet Oaks rates of GPP were significant ($F_{1,29} = 5.049$, $p < 0.05$), but not in South Bay ($p > 0.05$). High productivity (Fig. 3.11b) occurred during the summer months (June through September) with lowest occurring during the winter months (November through March). Rates of GPP were similar in both years of the study in South Bay with SBB typically having higher rates of productivity. Even though chl-*a* (Fig. 3.11a) was high during 2012 in SBB which corresponded with high k_d values (-5.5) (Fig. 3.11c) rates of GPP (Fig. 3.11b) remained similar to 2013. May 2012 was the only time GPP exceeded 1000 mg

$C/m^2/day$ but in 2013 was exceeded by IOA in June through August and in July by IOB where k_d values were again high (Fig. 11c).

According to a least squares liner regression analysis there was a significant relationship between log chl-*a* and log GPP, log TSS, log TN, log TP, log DIN, logTN/TP, logDIN/SRP (n=124, p = 0.000) and SRP (n=124, $p < 0.05$) As values of chlorophyll-*a* increased values of GPP, TSS, TN, TP, and SRP increased (Fig. 3.12) but as values of DIN, logTN/TP, logDIN/SRP (Fig. 3.14 a and b) decreased chlorophyll-*a* increased slightly. TN proved to be a good indicator of chl-*a* (Figure 3.12c) and aeration had a coefficient of determination ($R^2 = 0.492$) which was higher than non-aerated ($R^2 = 0.424$). However, TP proved to be a better indicator of chl-*a* with aeration having a higher ($R^2 = 0.618$) and non-aerated having a lower ($R^2 = 0.497$) coefficient of determination.

According to a least squares linear regression analysis there was a significant relationship between log GPP and log TSS, log TN, log TP, log DIN (n=124, $p < 0.05$) but SRP was not significant (n=124, $p > 0.05$). As rates of GPP increased values of TSS, TP and TN increased (Fig. 3.13) but as values of DIN, logTN/TP, logDIN/SRP (Fig. 3.14 c and d) decreased GPP increased slightly. TSS proved to be the best indicator of GPP rates (Figure 3.13a) and aeration had a coefficient of determination ($R^2 = 0.211$) which was higher than non-aerated ($R^2 = 0.246$) all other values had small coefficient of determination values ($R^2 < 0.1$).

Community Composition from an RCB ANOVA (Table 3.1) showed there was a significant difference in chrysophytes and diatoms ($F_{1,59} = 6.585$, $p < 0.05$), and

dinoflagellates ($F_{1,59} = 19.296$, $p < 0.05$) with period. As determined by MANOVAs, in South Bay the dinoflagellates ($F_{1,60} = 0.508$, $p < 0.012$) (Table 3.2), and in Inlet Oaks chrysophytes and diatoms ($F_{1,29} = 14.924$, $p < 0.05$) (Table 3.3), cryptophytes ($F_{1,29} = 6.080$, $p < 0.05$), and dinoflagellates ($F_{1,29} = 28.750$, $p = 0.00$) were all significantly different with treatment.

All sites were dominated by chlorophytes and euglenophytes (e.g. *Pediastrum sp., Scenedesmus sp., Ankistrodesmus sp., Euglena sp.*) in 2012 until early September (Fig. 3.15) and then chrysophytes and diatoms (e.g. *Synedra sp., Dinobryon sp., Pinnularia sp.*) bloomed during the colder months typically October through February 2013. Late summer cyanobacteria blooms (e.g. *Microcystis sp., Amphizomenon sp., Anabaena sp.*) occurred August through October.

During the summer 2012, in SBA a bloom of dinoflagellates making increasing contributions (19.12 -59.34%) in June through early August (Fig 15a). Chrysophytes and diatoms bloomed late winter (43.81 - 59.72%) and then dinoflagellates and cryptophytes (24.07 – 96.23%) during summer 2013 with increasing contributions of cyanobacteria than in 2012 (2.41 – 28.19%). In SBB, a bloom of cyanobacteria occurred (Fig. 3.15b) in late August (17.89 – 44.36%) accompanied by the dominance of chrysophytes and diatoms (56.63 – 85.49%) during the colder months (November through February). During 2013, the composition shifted to cryptophytes and dinoflagellates (29.30 – 93.43%) (e.g. *Cryptomomas sp., Prorocentrum sp.*) in June through September in both sites.

In IOA there was an increase in cyanobacteria in late summer 2012 (Fig. 3.15c) (14.76 – 26.47%) accompanied by increasing chrysophytes and diatoms. During 2013,

chlorophytes and euglenophytes with greater *Eudorina sp.*, making up a majority of composition (56.70 – 88.53%) but there was a bloom of dinoflagellates and cryptophytes late summer making up 43.29 -85.82% of total composition. IOB was different than IOA in 2013 (Fig. 3.15d) by having high chrysophytes and diatoms (e.g. *Dinobryon sp.*, *Synura sp*.) contributions remaining after winter bloom (52.35 – 90.09%) and continuing through 2013. Chlorophytes and euglenophytes increased as well during the summer. A bloom of Cryptophytes and Dinoflagellates occurred in late April (46.82 - 70.25%)

Total precipiatation 48 hours before a sampling event ranged in both of the sites and during both years of study (Fig. 3.16). The Inlet Oaks sites featured the highest precipitation overall in both years of study. During 2013, with aeration, there was higher precipitation overall (~8.92 in) in South Bay and Inlet Oaks but in 2012 there was (~6.05 in).

Table 3.1: Results of RCB ANOVA where values are the differences between the control and the impact with sites (South Bay/ Inlet Oaks) treated as the blocking effect. Periods are before (no treatment) and after (with aerator installed in impact).

	Period (Treatment)		**ANOVA**	
	Before	**After**	**F**	**p-value**
	Mean ± 1 SE	Mean ± 1 SE		
Chl-*a* (μgL^{-1})	0.20 ± 0.074	0.014 ± 0.077	3.189	0.079
TSS (mgL^{-1})	0.18 ± 0.055	-0.031 ± 0.057	6.873	0.011**
OSS (mgL^{-1})	0.17 ± 0.055	0.0061 ± 0.056	4.537	0.037**
ISS (mgL^{-1})	0.067 ± 0.11	-0.021 ± 0.11	0.338	0.563
DOC (mgL^{-1})	0.080 ± 0.017	0.065 ± 0.018	0.392	0.534
TN Shimadzu (mgL^{-1})	-0.019 ± 0.027	-0.0091 ± 0.027	0.075	0.785
TN (mgL^{-1})	0.091 ± 0.036	0.025 ± 0.037	1.653	0.204
TP (mgL^{-1})	0.19 ± 0.064	0.018 ± 0.066	3.342	0.073
Dissolved TN (mgL^{-1})	0.032 ± 0.024	-0.015 ± 0.024	1.878	0.176
Dissolved TP (mgL^{-1})	0.052 ± 0.067	-0.023 ± 0.069	0.616	0.436
NH_4^+ (mgL^{-1})	-0.44 ± 0.13	-0.32 ± 0.13	0.391	0.534
NOx (mgL^{-1})	-0.59 ± 0.14	-0.67 ± 0.14	0.155	0.695
NO_3 (mgL^{-1})	-0.51 ± 0.27	0.25 ± 0.28	3.829	0.055
NO_2 (mgL^{-1})	-0.24 ± 0.097	-0.061 ± 0.10	1.676	0.200
DIN (mgL^{-1})	-0.52 ± 0.13	-0.50 ± 0.13	0.012	0.912
DON (mgL^{-1})	0.11 ± 0.052	0.083 ± 0.054	0.158	0.692
SRP (mgL^{-1})	0.20 ± 0.094	-0.027 ± 0.098	2.892	0.094
DOP (mgL^{-1})	0.41 ± 0.13	-0.0061 ± 0.14	4.918	0.030**
CDOM (mgL^{-1})	0.14 ± 0.028	0.11 ± 0.029	0.539	0.466
PN (mgL^{-1})	0.16 ± 0.057	0.016 ± 0.059	3.080	0.084
POC (mgL^{-1})	0.16 ± 0.057	0.018 ± 0.058	3.260	0.076
TN:TP	0.27 ± 2.6	4.8 ± 2.9	1.331	0.254
DIN:SRP	-3.7 ± 3.6	7.1 ± 4.01	4.042	0.049
GPP mgC/m^2 /day	0.066 ± 0.078	-0.097 ± 0.081	2.067	0.156
K_d	0.11 ± 0.037	0.0031 ± 0.038	4.194	0.045**
Chlorophytes &Euglenophytes (%)	-0.48 ± 0.23	-0.40 ± 0.24	0.057	0.812
Chrysophytes &Diatoms (%)	-0.62 ± 0.26	0.32 ± 0.26	6.585	0.013**
Cryptophytes (%)	-0.045 ± 0.26	-0.26 ± 0.26	0.352	0.555
Cyanobacteria (%)	-0.19 ± 0.23	-0.52 ± 0.24	1.020	0.317
Dinoflagellates (%)	-0.36 ± 0.33	-1.8 ± 0.24	19.296	0.000**

** denotes a significant p-value < 0.05

Table 3.2: Results of MANOVA of period of South Bay sites where values are the differences between the control and the impact and sites. Periods are before (no treatment) and after (with aerator installed in impact site).

	Period (Treatment)		**ANOVA**	
	Before	**After**	**F**	**p-value**
	Mean ± 1 SE	Mean ± 1 SE		
Chl-*a* (μgL^{-1})	0.31 ± 0.12	0.19 ± 0.12	0.326	0.482
TSS (mgL^{-1})	0.29 ± 0.087	0.22 ± 0.090	0.168	0.573
OSS (mgL^{-1})	0.29 ± 0.086	0.23 ± 0.089	2.237	0.685
ISS (mgL^{-1})	-0.13 ± 0.16	0.21 ± 0.16	2.404	0.146
DOC (mgL^{-1})	0.0081 ± 0.017	-0.031 ± 0.018	0.222	0.132
TN Shimadzu (mgL^{-1})	0.0080 ± 0.034	-0.015 ± 0.035	4.791	0.641
TN (mgL^{-1})	0.19 ± 0.041	0.060 ± 0.042	2.248	0.037**
TP (mgL^{-1})	0.20 ± 0.068	0.048 ± 0.071	9.846	0.145
Dissolved TN (mgL^{-1})	0.10 ± 0.026	-0.016 ± 0.027	0.016	0.004**
Dissolved TP (mgL^{-1})	-0.078 ± 0.11	-0.057 ± 0.12	0.288	0.900
NH_4^+ (mgL^{-1})	-0.27 ± 0.18	-0.13 ± 0.19	0.033	0.595
NOx (mgL^{-1})	-0.31 ± 0.20	-0.36 ± 0.21	3.007	0.857
NO_3 (mgL^{-1})	-0.39 ± 0.34	0.44 ± 0.35	0.383	0.094
NO_2 (mgL^{-1})	-0.12 ± 0.12	-0.012 ± 0.12	0.021	0.541
DIN (mgL^{-1})	-0.31 ± 0.18	-0.27 ± 0.18	2.384	0.885
DON (mgL^{-1})	0.14 ± 0.053	0.021 ± 0.054	0.059	0.133
SRP (mgL^{-1})	-0.059 ± 0.12	-0.10 ± 0.13	4.820	0.809
DOP (mgL^{-1})	0.31 ± 0.12	-0.052 ± 0.12	8.081	0.036**
CDOM (mgL^{-1})	0.015 ± 0.017	-0.054 ± 0.017	0.084	0.008**
PN (mgL^{-1})	0.25 ± 0.095	0.21 ± 0.098	0.012	0.774
POC (mgL^{-1})	0.25 ± 0.090	0.23 ± 0.093	0.260	0.915
TN:TP	0.92 ± 1.4	1.5 ± 1.6	0.086	0.772
DIN:SRP	-4.3 ± 3.6	2.4 ± 4.1	1.487	0.234
GPP mgC/m^2 /day	0.037 ± 0.14	-0.068 ± 0.15	0.889	0.614
K_d	0.22 ± 0.058	0.14 ± 0.060	0.532	0.354
Chlorophytes &Euglenophytes (%)	-0.69 ± 0.44	-0.22 ± 0.46	2.055	0.472
Chrysophytes &Diatoms (%)	1.34 ± 0.49	-0.33 ± 0.50	1.371	0.162
Cryptophytes (%)	-0.16 ± 0.49	-0.98 ± 0.50	0.279	0.251
Cyanobacteria (%)	-0.30 ± 0.44	-0.64 ± 0.46	7.259	0.601
Dinoflagellates (%)	0.31 ± 0.12	0.19 ± 0.12	0.508	0.012**

** denotes a significant p-value < 0.05

Table 3.3: Results of MANOVA of period of Inlet Oaks sites where values are the differences between the control and the impact and sites. Periods are before (no treatment) and after (with aerator installed in impact site).

	Period (Treatment)		**ANOVA**	
	Before	**After**	**F**	**p-value**
	Mean ± 1 SE	Mean ± 1 SE		
Chl-*a* (μgL^{-1})	0.099 ± 0.094	-0.16 ± 0.097	3.751	0.063
TSS (mgL^{-1})	0.059 ± 0.063	-0.28 ± 0.065	14.238	0.001**
OSS (mgL^{-1})	0.061 ± 0.065	-0.22 ± 0.067	9.266	0.005**
ISS (mgL^{-1})	0.27 ± 0.12	-0.25 ± 0.12	8.797	0.006**
DOC (mgL^{-1})	0.15 ± 0.029	0.16 ± 0.030	0.040	0.842
TN Shimadzu (mgL^{-1})	-0.046 ± 0.041	-0.0021 ± 0.042	0.571	0.456
TN (mgL^{-1})	-0.0071 ± 0.059	-0.011 ± 0.061	0.002	0.961
TP (mgL^{-1})	0.18 ± 0.11	-0.012 ± 0.11	1.447	0.239
Dissolved TN (mgL^{-1})	-0.039 ± 0.038	-0.013 ± 0.039	0.235	0.632
Dissolved TP (mgL^{-1})	0.18 ± 0.066	0.011 ± 0.068	3.284	0.080
NH_4^+ (mgL^{-1})	-0.60 ± 0.18	-0.51 ± 0.19	0.115	0.737
NOx (mgL^{-1})	-0.88 ± 0.20	-0.98 ± 0.20	0.139	0.712
NO_3 (mgL^{-1})	-0.63 ± 0.43	0.054 ± 0.44	1.225	0.278
NO_2 (mgL^{-1})	-0.37 ± 0.16	-0.11 ± 0.16	1.305	0.263
DIN (mgL^{-1})	-0.74 ± 0.18	-0.74 ± 0.19	0.000	0.991
DON (mgL^{-1})	0.087 ± 0.089	0.14 ± 0.092	0.200	0.658
SRP (mgL^{-1})	0.47 ± 0.14	0.048 ± 0.15	4.170	0.050
DOP (mgL^{-1})	0.51 ± 0.24	0.040 ± 0.25	1.906	0.178
CDOM (mgL^{-1})	0.26 ± 0.053	0.27 ± 0.054	0.019	0.891
PN (mgL^{-1})	0.067 ± 0.063	-0.18 ± 0.065	7.550	0.010**
POC (mgL^{-1})	0.083 ± 0.066	-0.20 ± 0.068	8.756	0.006**
TN:TP	-0.37 ± 5.03	7.7 ± 5.2	1.236	0.275
DIN:SRP	-3.06 ± 6.2	11.0 ± 6.4	2.601	0.118
GPP mgC/m^2 /day	0.095 ± 0.068	-0.13 ± 0.070	5.049	0.032**
K_d	0.0021 ± 0.046	-0.14 ± 0.047	4.353	0.046**
Chlorophytes &Euglenophytes (%)	-0.28 ± 0.11	-0.58 ± 0.11	4.172	0.050
Chrysophytes &Diatoms (%)	0.099 ± 0.16	0.98 ± 0.16	14.924	0.001**
Cryptophytes (%)	0.067 ± 0.11	0.454 ± 0.113	6.080	0.020**
Cyanobacteria (%)	-0.071 ± 0.16	-0.41 ± 0.16	2.254	0.144
Dinoflagellates (%)	-0.034 ± 0.16	-1.3 ± 0.17	28.750	0.000**

** denotes a significant p-value < 0.05

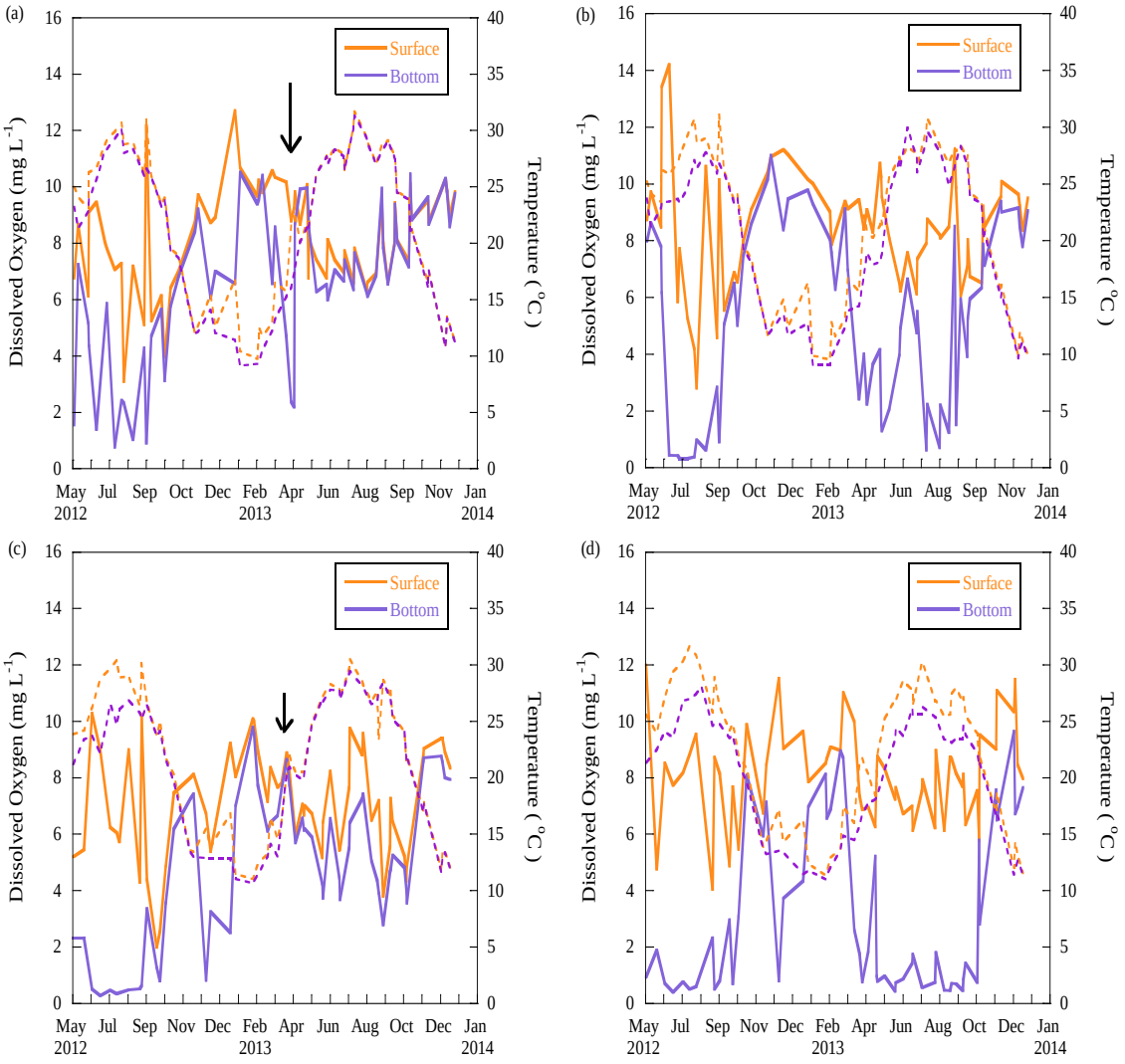

Figure 3.1. Comparison of Dissolved Oxygen (mg L^{-1}) (solid lines) and temperature (ºC) (dashed lines) profiles at (a) SBA, (b) SBB, (c) IOA, (d) IOB. Arrows correspond to the installation of the aerators in SBA and IOA. Surface measurements (orange) are taken at 0.5 m below the surface of the water and bottom (purple) were taken at 0.5 m above the sediment.

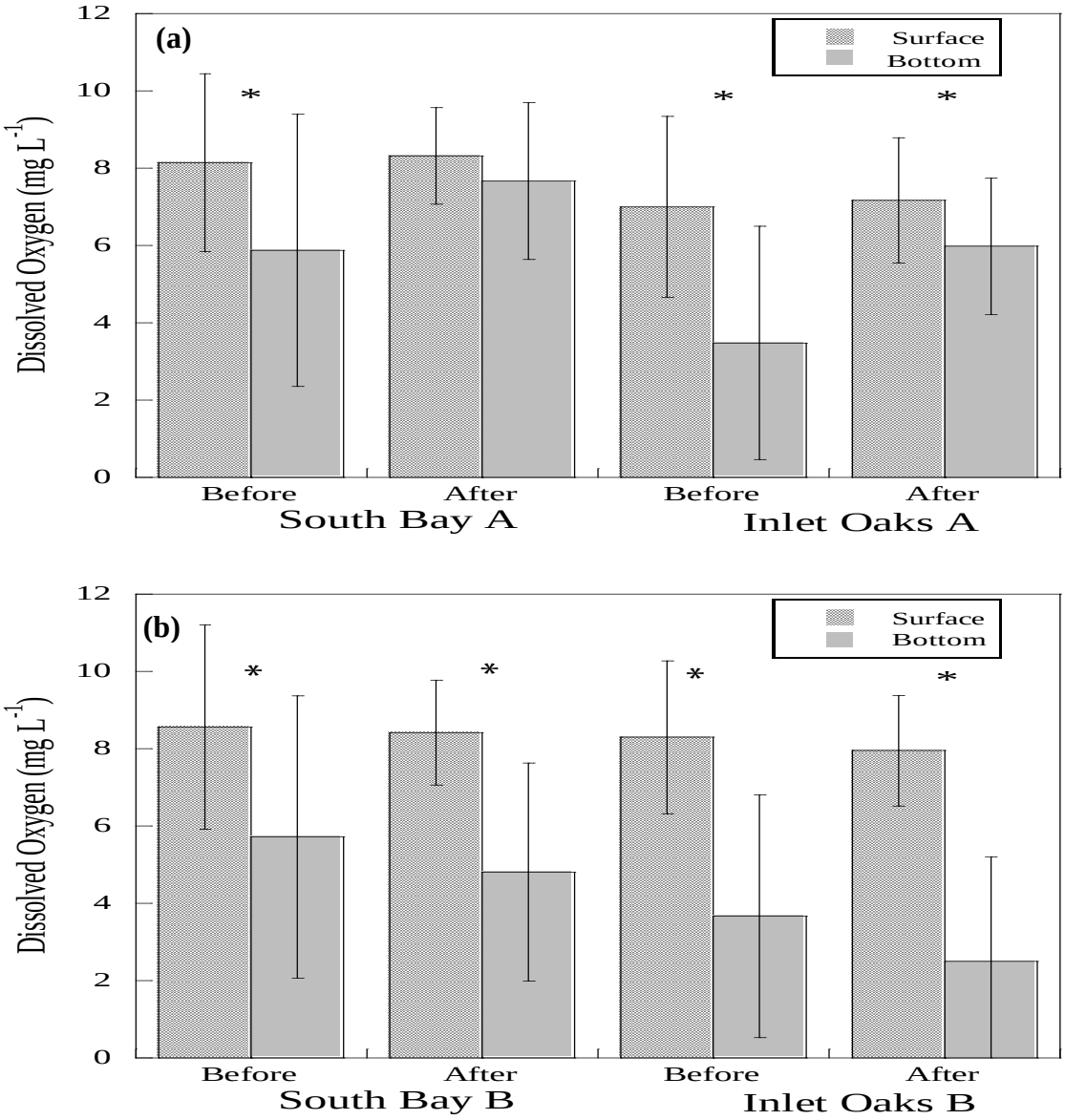

Figure 3.2. Comparison of Dissolved Oxygen (mg L^{-1}) at (a) aerated sites, (b) non-aerated sites before and after treatment for surface (diagonal pattern) and bottom (solid) concentrations. A star (*) denotes a significant difference ($p < 0.01$) between surface and bottom DO concentrations.

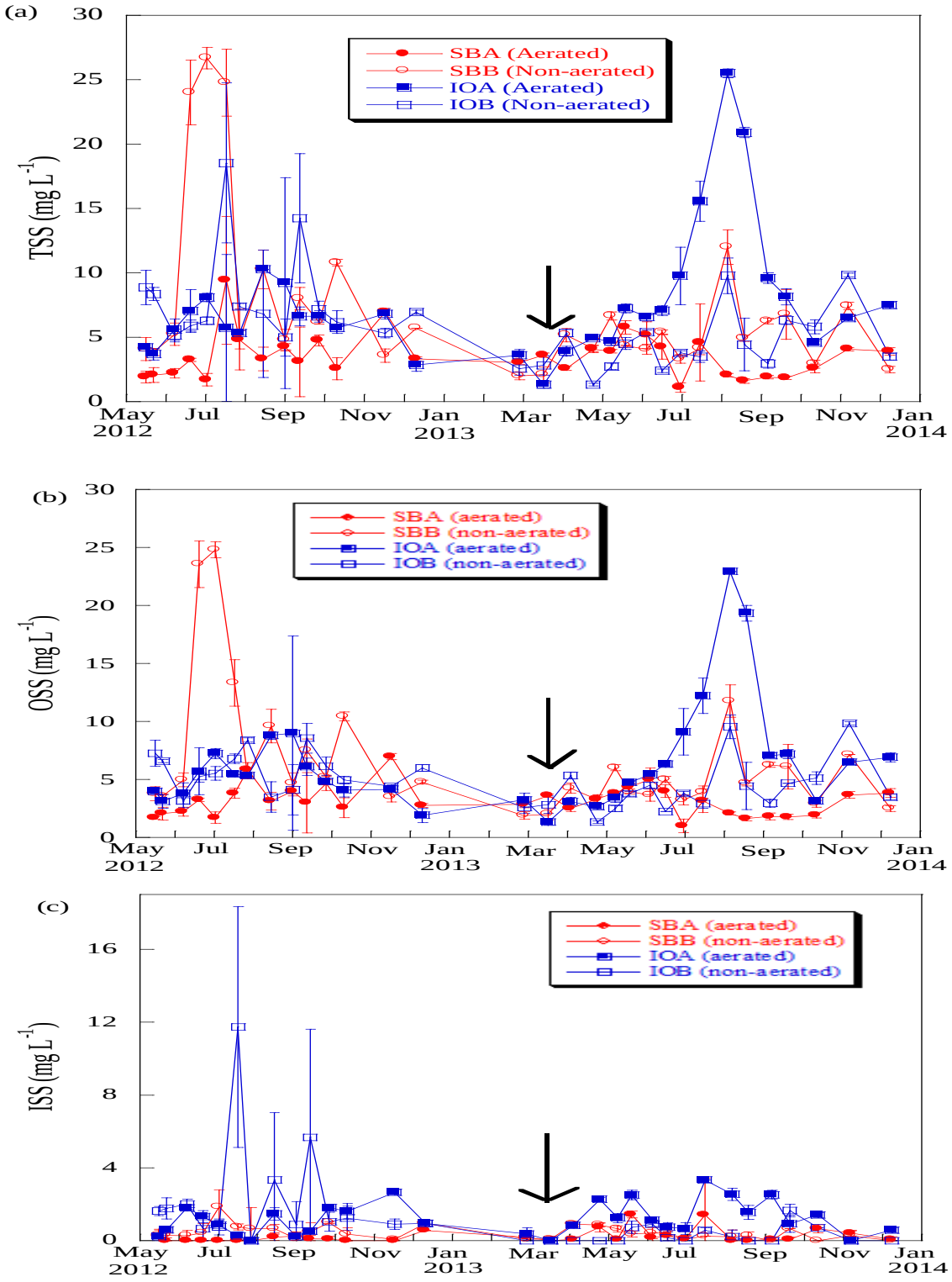

Figure 3.3. Average concentrations of (a) TSS (mg L^{-1}), (b) OSS (mg L^{-1}), (c) ISS (mg L^{-1}) in each of the stormwater ponds. The arrow indicates installation of the aerator and error bars are ± 1 standard deviation of the average.

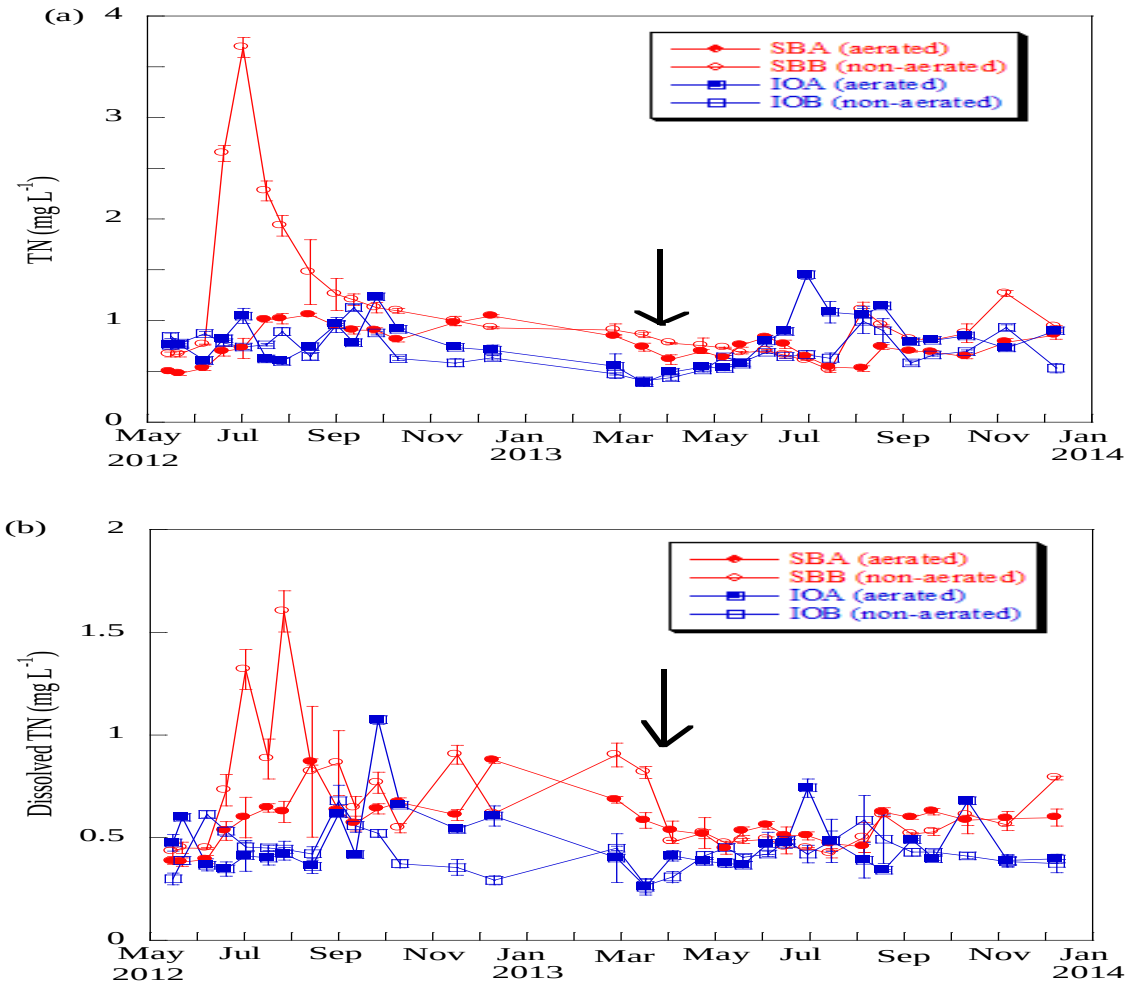

Figure 3.4. Average concentrations of (a) TN (mg L^{-1}) and (b) TDN (mg L^{-1}) in each of the stormwater ponds. The arrow indicates installation of the aerator and error bars are ± 1 standard deviation of the average.

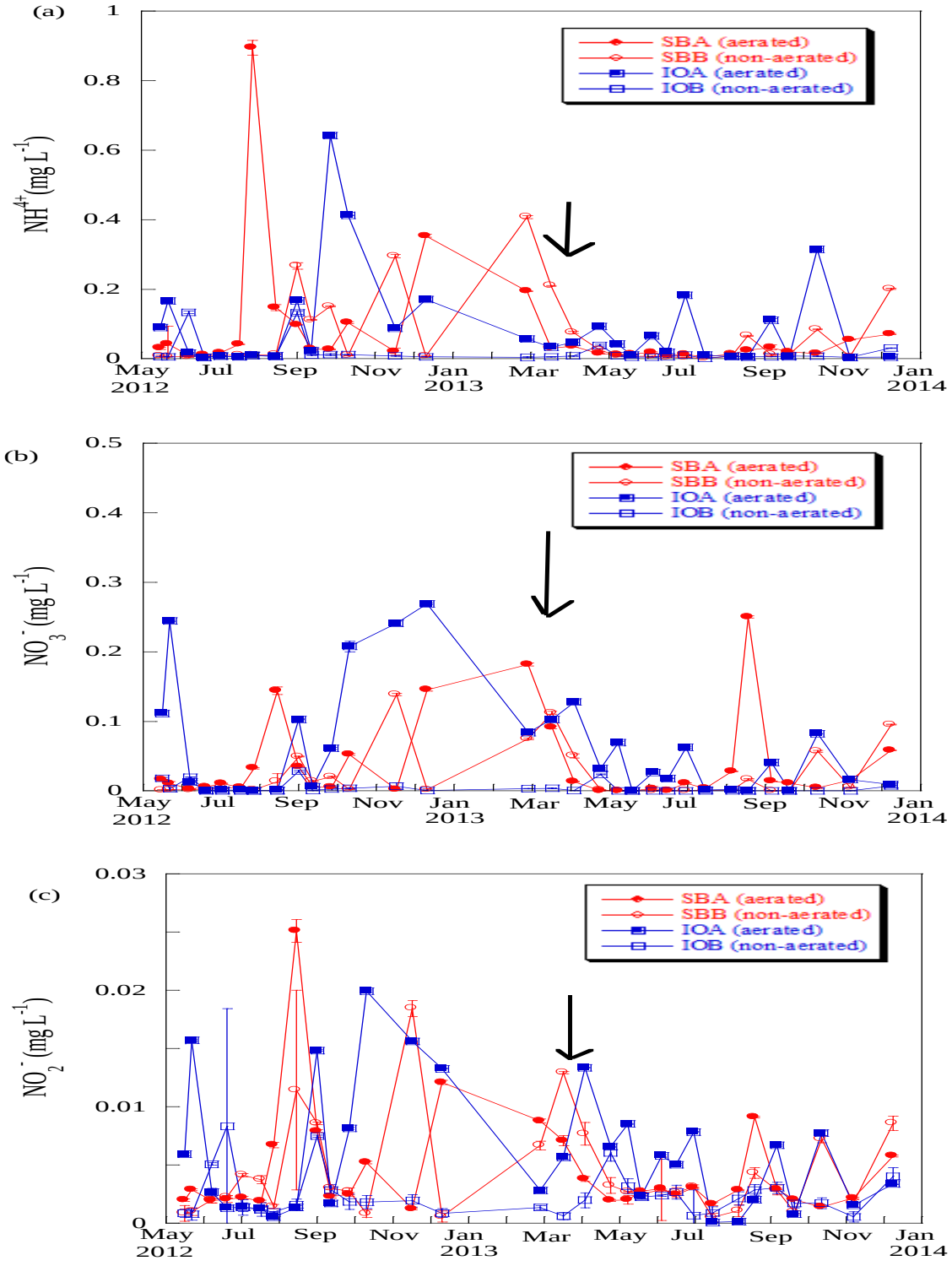

Figure 3.5. Average concentrations of (a) NH_4^+ (mg L^{-1}), (b) NO_3^- (mg L^{-1}), (c) NO_2^- (mg L^{-1}) in each of the stormwater ponds. The arrow indicates installation of the aerator and error bars are ± 1 standard deviation of the average.

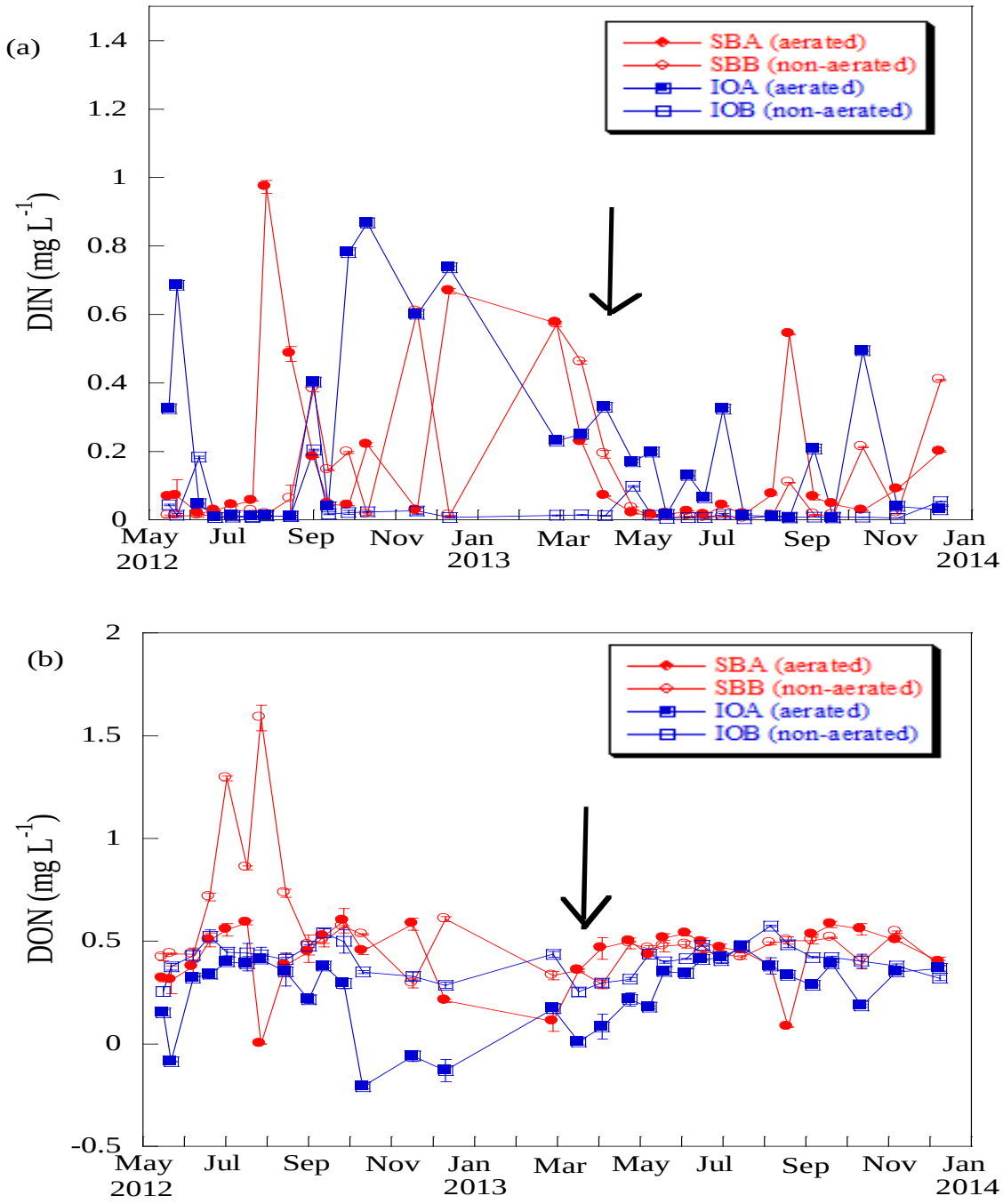

Figure 3.6. Average concentrations of (a) DIN (mg L^{-1}) and (b) DON (mg L^{-1}) in each of the stormwater ponds. The arrow indicates installation of the aerator and error bars are ± 1 standard deviation of the average.

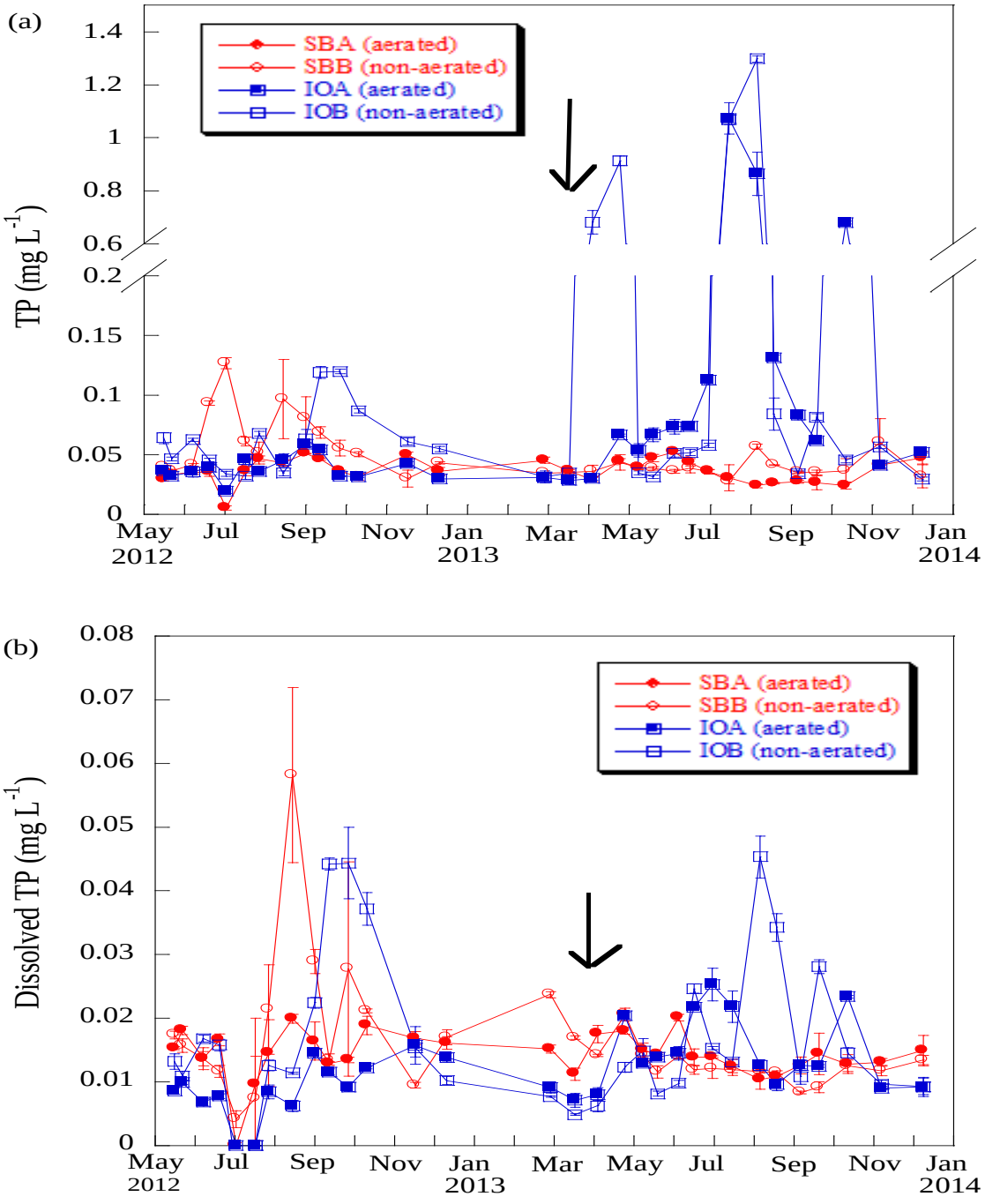

Figure 3.7. Average concentrations of (a) TP (mg L^{-1}) and (b) TDP (mg L^{-1}) in each of the stormwater ponds. The arrow indicates installation of the aerator and error bars are ± 1 standard deviation of the average.

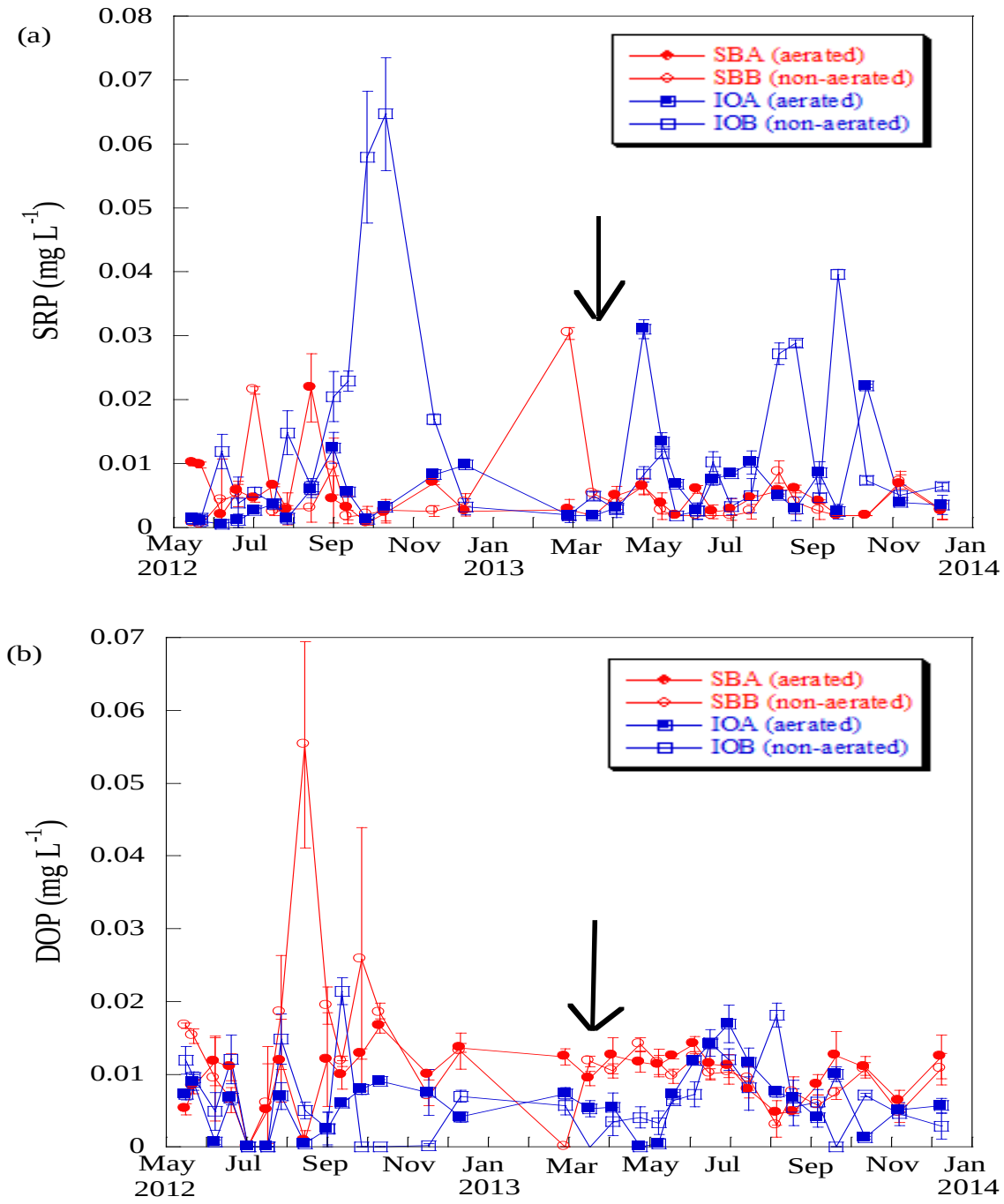

Figure 3.8. Average concentrations of (a) SRP (mg L^{-1}) and (b) DOP (mg L^{-1}) in each of the stormwater ponds. The arrow indicates installation of the aerator and error bars are ± 1 standard deviation of the average.

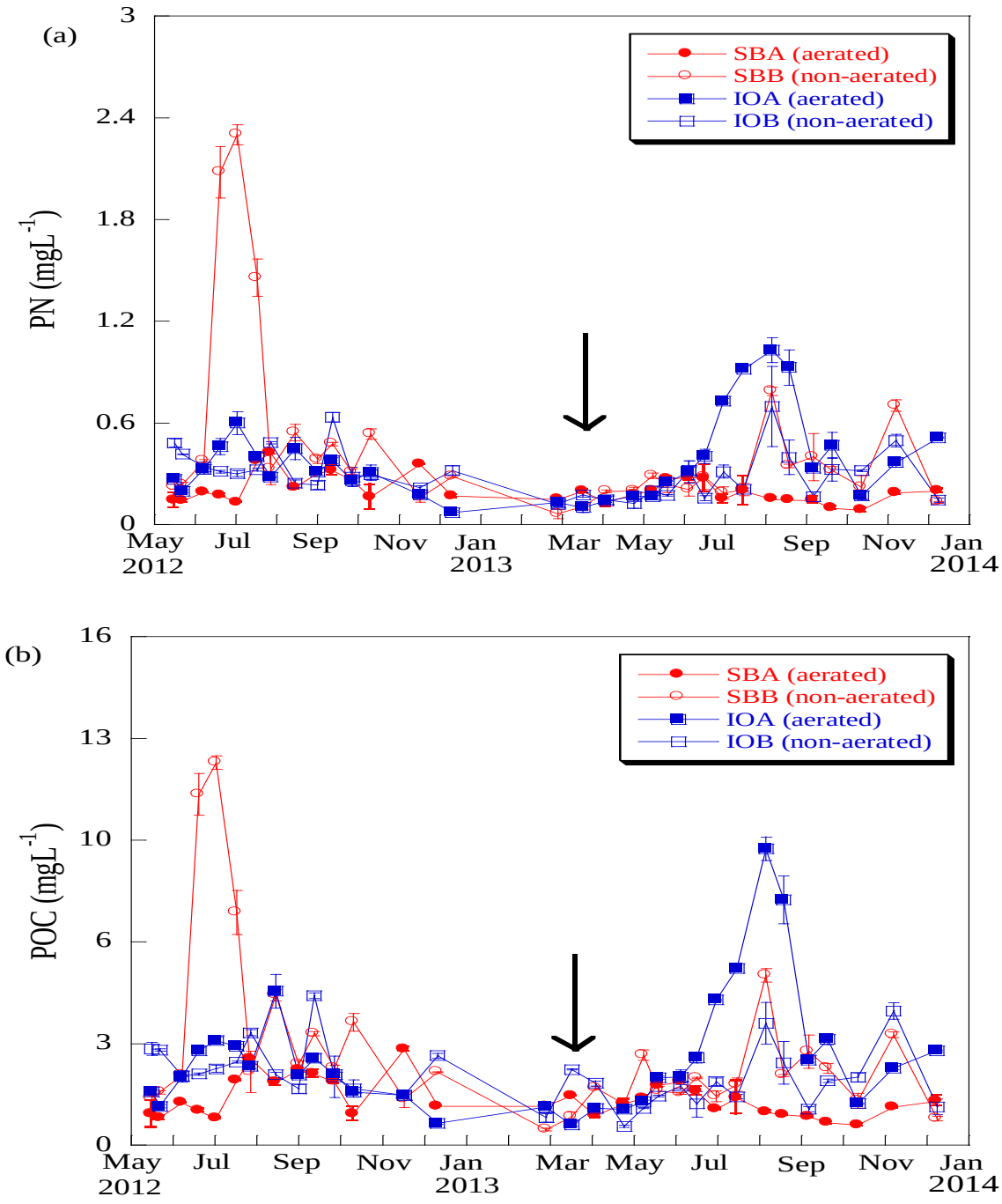

Figure 3.9. Average concentrations of (a) PN (mg L^{-1}) and (b) POC (mg L^{-1}) in each of the stormwater ponds. The arrow indicates installation of the aerator and error bars are ± 1 standard deviation of the average.

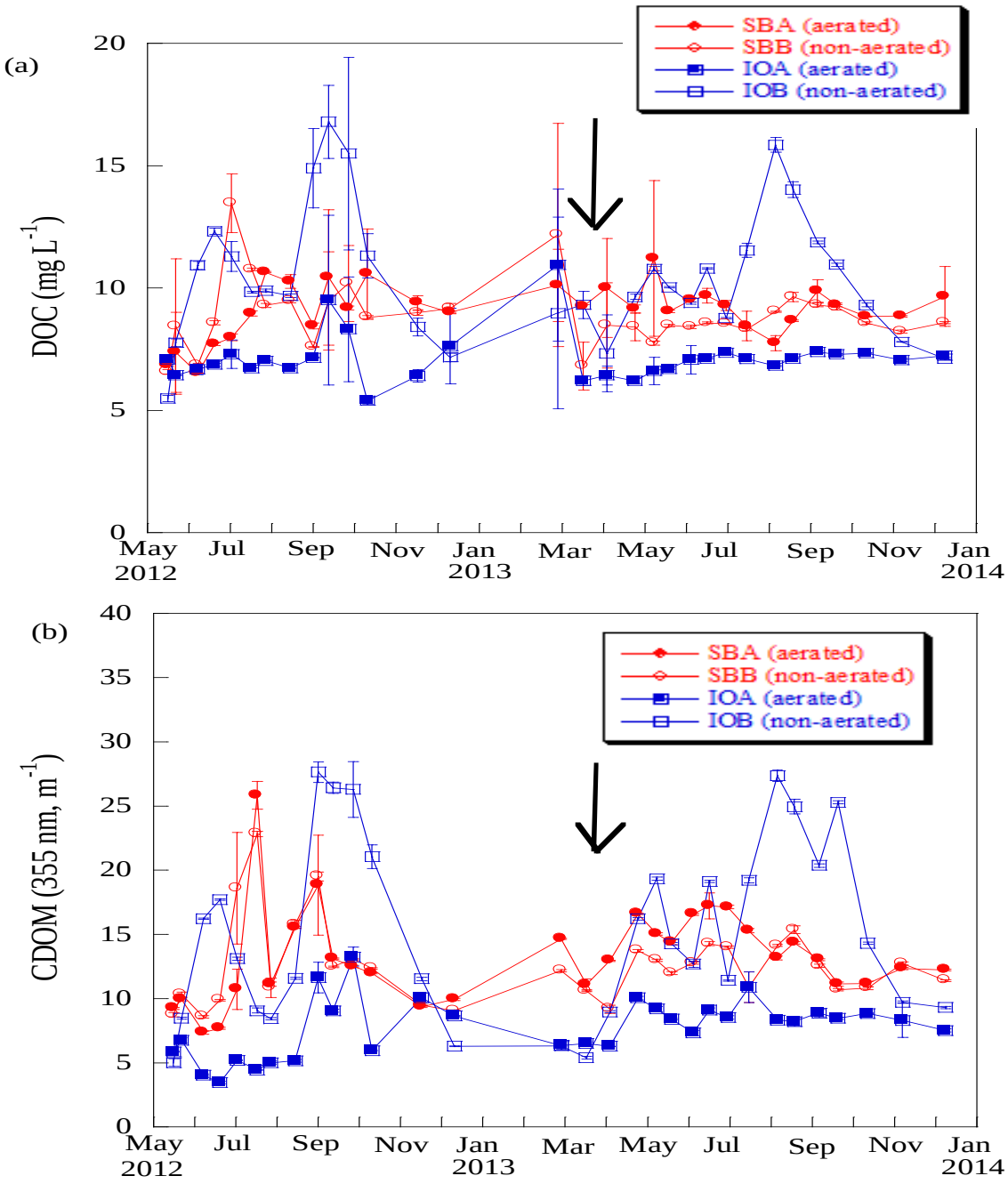

Figure 3.10. Average concentrations of (a) DOC (mg L^{-1}) and (b) CDOM (mg L^{-1}) in each of the stormwater ponds. The arrow indicates installation of the aerator and error bars are ± 1 standard deviation of the average.

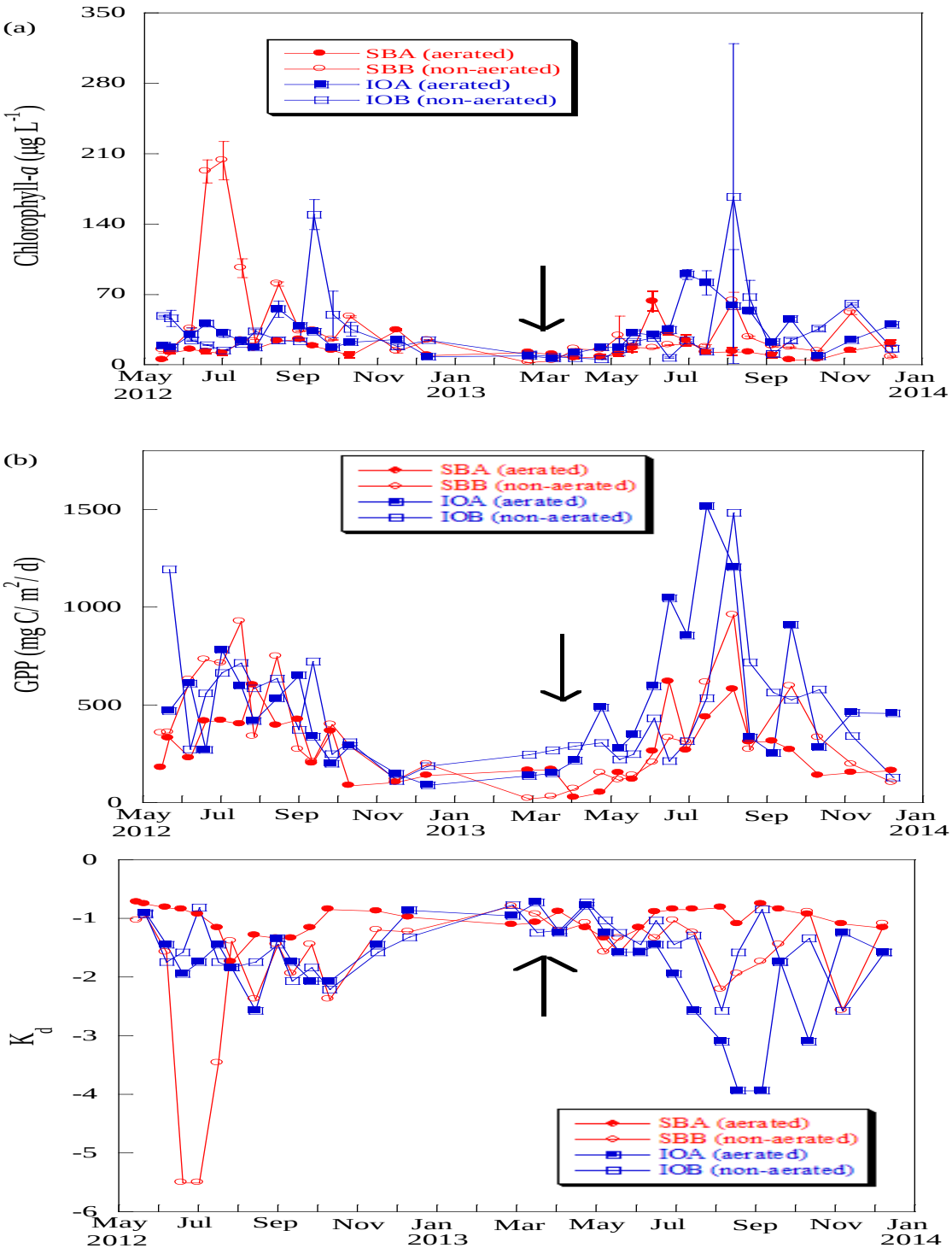

Figure 3.11. Average concentrations of (a) Chlorophyll-*a* (µg L^{-1}) and (b) GPP (mg C/m^2/day) in each of the stormwater ponds. The arrow indicates installation of the aerator and error bars are ± 1 standard deviation of the average.

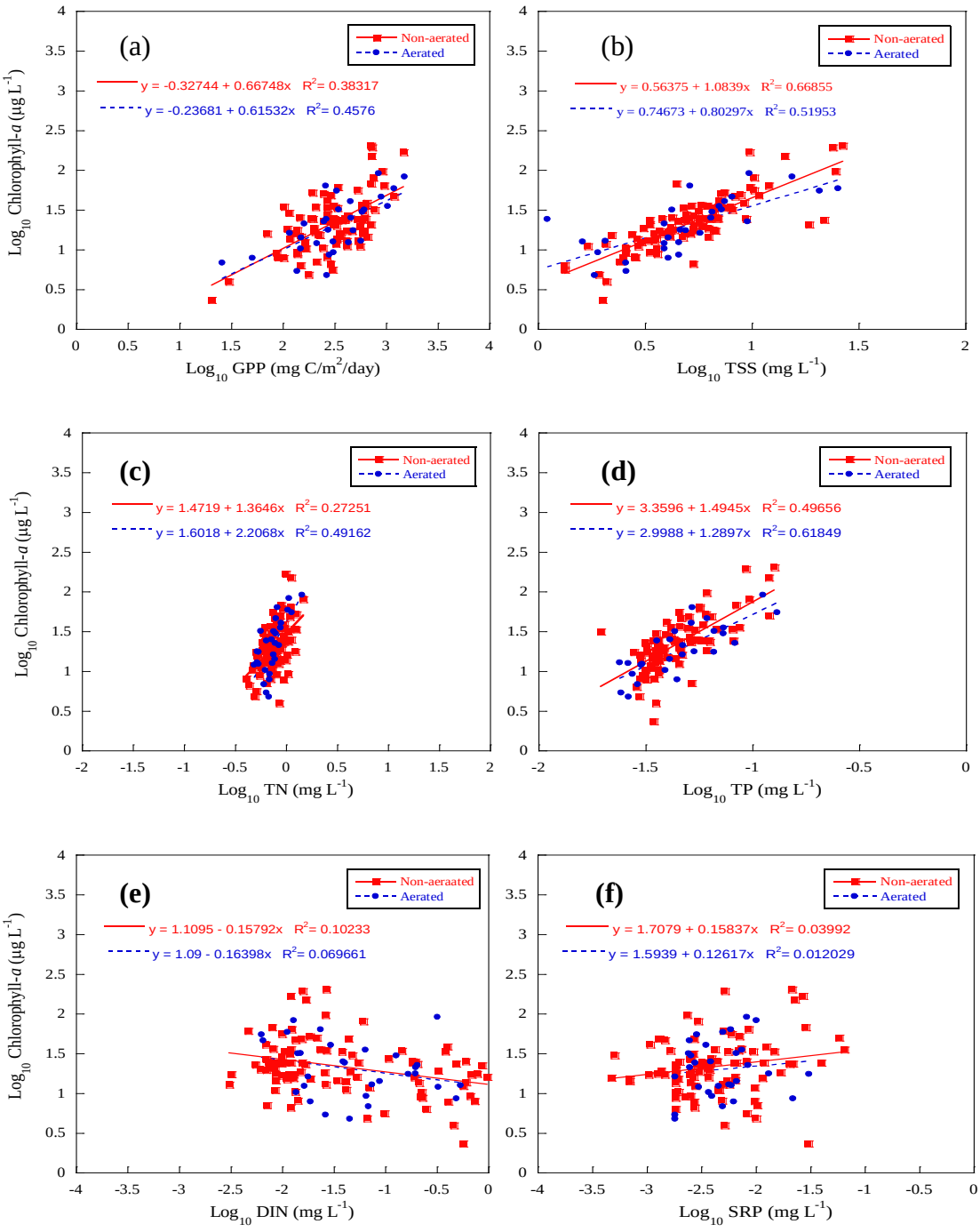

Figure 3.12. Least squares linear regression analysis (a) log chl-*a* (μgL^{-1}) vs. log GPP ($mgC/m^2/day$), (b) log chl-*a* (μgL^{-1}) vs. log TSS (mgL^{-1}), (c) log chl-*a* (μgL^{-1}) vs. log TN (mgL^{-1}), (d) log chl-*a* (μgL^{-1}) vs. log TP (mgL^{-1}), (e) log chl-*a* (μgL^{-1}) vs. log DIN (mgL^{-1}), (f) log chl-*a* (μgL^{-1}) vs. log SRP (mgL^{-1}) in each of the stormwater ponds. The red squares are non-aerated values and blue circles are aerated values.

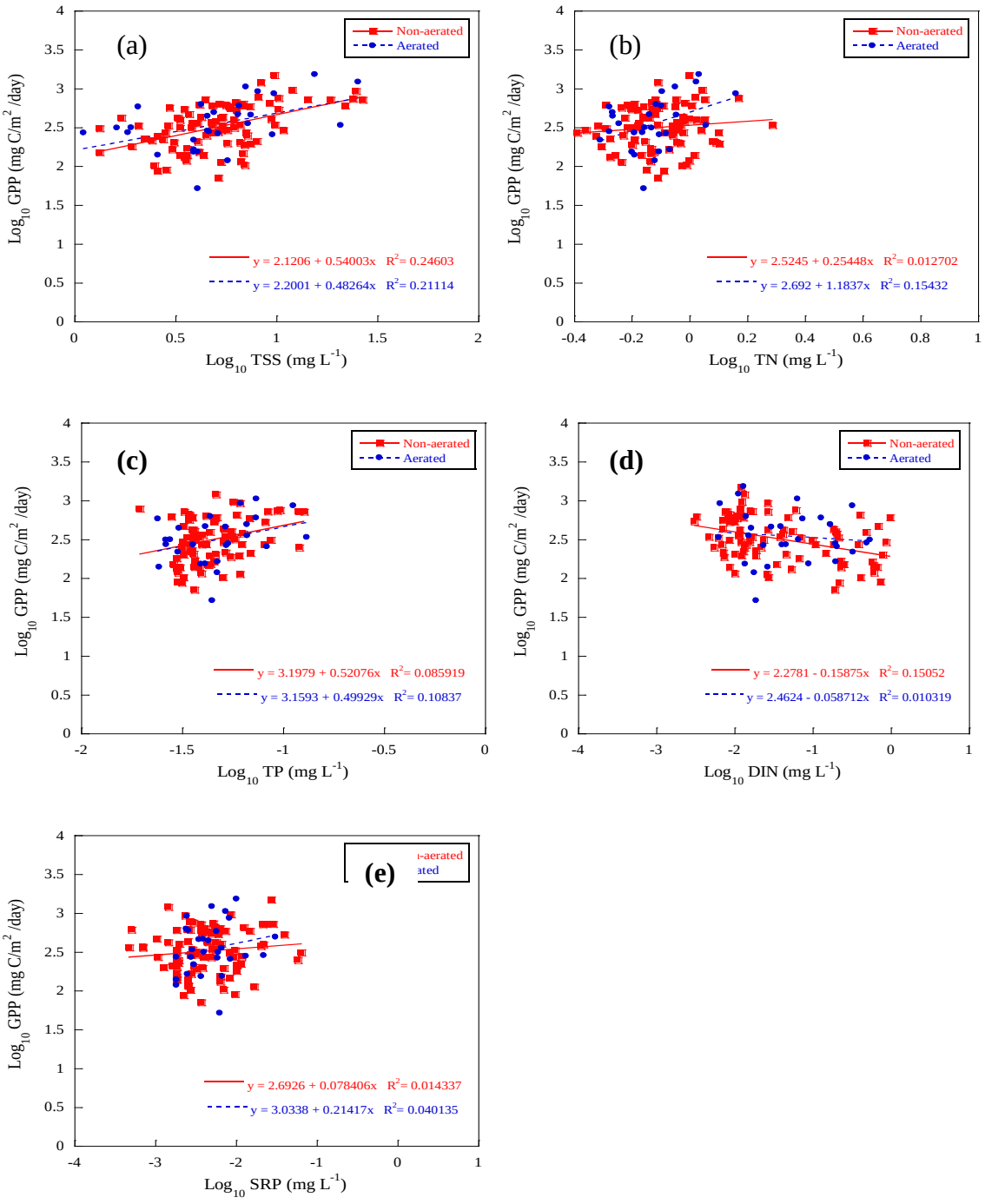

Figure 3.13. Least squares linear regression analysis (a) log GPP ($mgC/m^2/day$) vs. log TSS (mgL^{-1}), (b) log GPP ($mgC/m^2/day$)vs. log TN (mgL^{-1}), (c) log GPP ($mgC/m^2/day$)vs. log TP (mgL^{-1}), (d) log GPP ($mgC/m^2/day$)vs. log DIN (mgL^{-1}), (e) log GPP ($mgC/m^2/day$) vs. log SRP (mgL^{-1}) in each of the stormwater ponds. The red squares are non-aerated values and blue circles are aerated values.

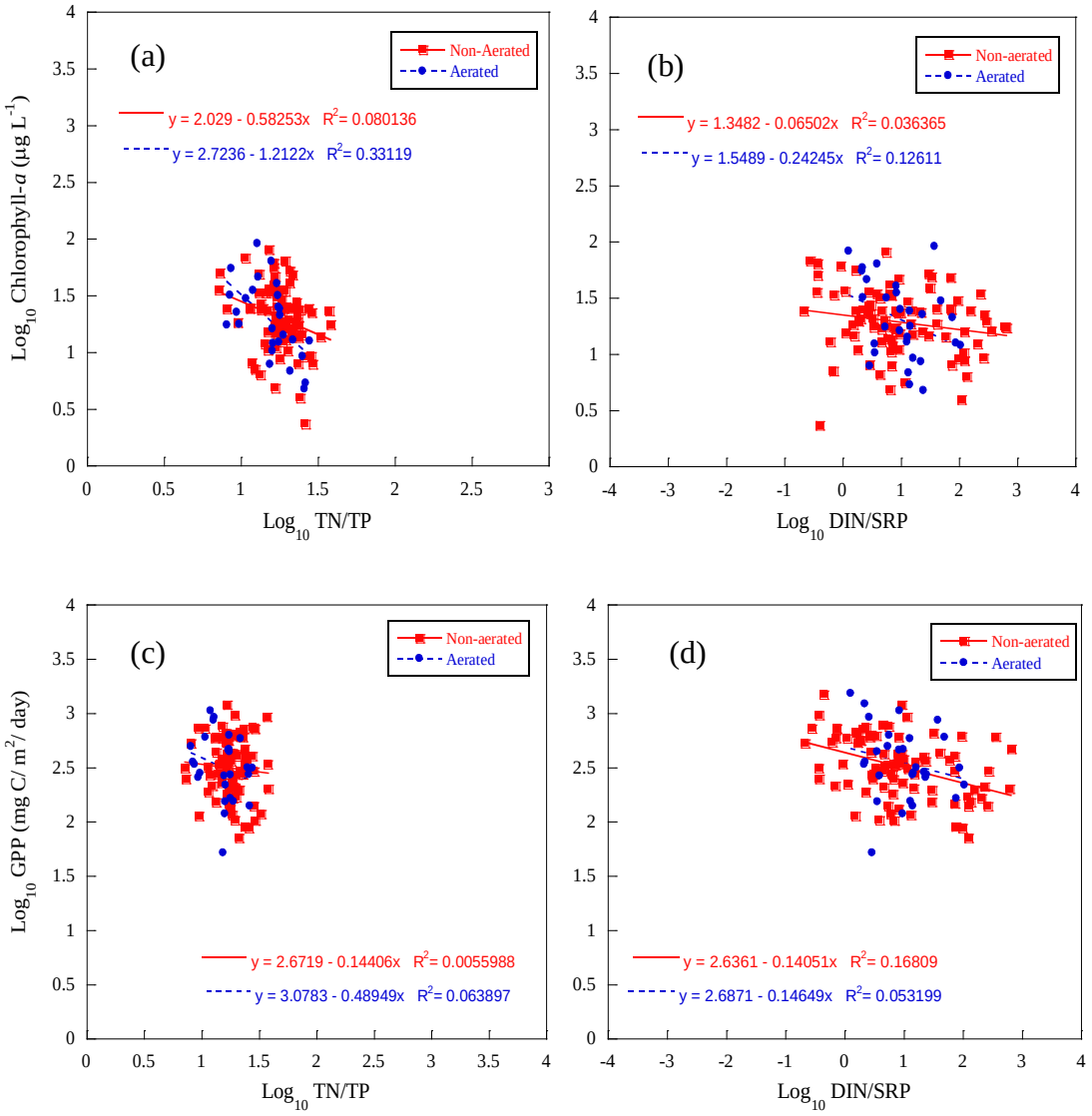

Figure 3.14. Least squares linear regression analysis (a) log chl-*a* (µgL^{-1}) vs. log TN/TP, (b) log chl-*a* (µgL^{-1}) vs. log DIN/SRP, (c) log GPP (mg C/m^2/day) vs. log TN/TP, (d) log GPP (mg C/m^2/day) vs. log DIN/SRP in each of the stormwater ponds. The red squares are non-aerated values and blue circles are aerated values.

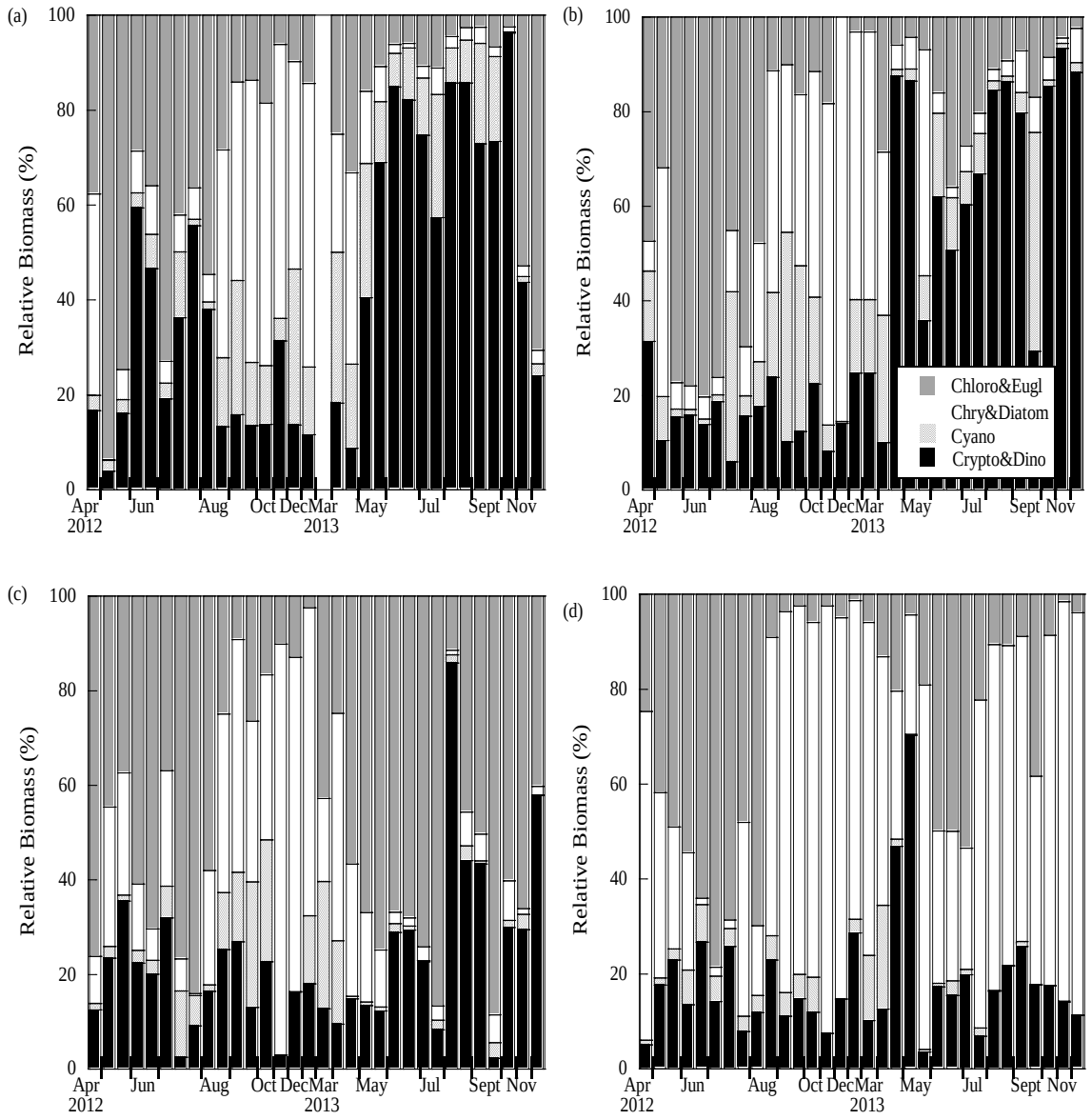

Figure 3.15. Relative abundances as % of total chlorophyll-*a* derived from HPLC and CHEMTAX based estimates in (a) SBA, (b) SBB, (c) IOA, (d) IOB. Phytoplankton community is comprised of chlorophytes & euglenophytes (grey color), chrysophytes and diatoms (white), cyanobacteria (diagonal stripes), dinoflagellates and cryptophytes (black). SBA is missing data for March 18, 2013 due to a power outage during HPLC run.

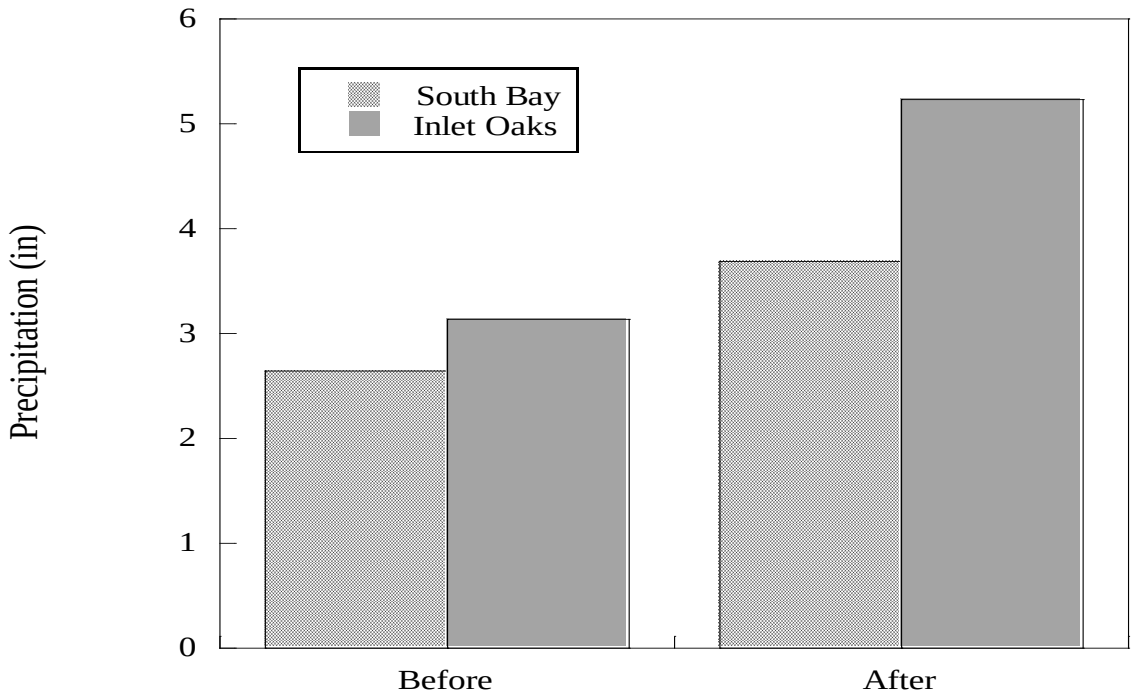

Figure 3.16. Total precipitation (in) 48 hours before sampling event (May – December) at South Bay and Inlet Oaks location before and after aerator installation. Winter precipitation data (February – April) was removed in the before period so sampling frequency was the same. Data was taken from the Community Collaborative Rain, Hail, and Snow Network (CoCoRaHS) (http://www.cocorahs.org/).

CHAPTER 4: DISCUSSION

Aeration has been reported to improve DO concentrations in the bottom waters and thereby create oxic conditions at the sediment water interface resulting in sequestration of nutrients which lowers total water column nutrients, decreasing phytoplankton biomass, productivity and altering community composition from mixing. Aeration did increase bottom water DO concentrations and did have an effect on TSS, OSS, DOP, k_d, and altered community composition. However, only community composition was altered in both sites. The observed aeration effect on nutrients was likely a product of site specific factors (e.g. external nutrient loading) rather than from the aeration.

Stormwater ponds of the coastal southeast are characterized by stratified water columns, high concentrations of nutrients, and warm water temperatures (DeLorenzo and Fulton, 2009). Bottom water DO concentrations reached points of hypoxia and even anoxia during the first year of this study in all of the ponds and were below the guideline concentration of < 4 mgL^{-1} indicative of conditions potentially harmful to aquatic organisms (SCDHEC, 2012). Stratification in the ponds was often intensified by high temperatures (> 25 $^{\circ}C$). Aeration caused a breakdown of thermal stratification and increased bottom DO concentrations, while non-aerated ponds still had depletion of bottom water DO concentrations often still reaching critical levels.

Concentrations of oxygen in the surface waters were high when primary productivity and chl-*a* concentrations were highest in surface waters as a result of oxygen

production during photosynthesis. However, even at the surface of the ponds DO concentrations dropped during the summer time due to outgassing and the resulting reduction in solubility (e.g. Boyd 1998). As chl-*a* concentrations and GPP rates were high this corresponded with large DOC concentrations. The DOC pools can stimulate heterotrophic bacteria activity and lower DO concentrations further in bottom waters (e.g. Prairie *et al.,* 2002). Symptoms of anoxia and hypoxia are of growing concern to estuarine areas globally (Diaz and Rosenberg, 2008) thus stormwater ponds can exacerbate DO impairment by discharging nutrient and DOC enriched waters into the coastal zone. This, in turn, will stimulate bacterial activity (e.g. Smith and Kemp, 2003). Aeration increased DO concentrations while DOC concentrations remained similar during both years of study.

Phosphorus proved to be the better indicator of phytoplankton biomass within these ponds. Aeration had no significant effect on TP in either of the ponds. Similarly, the effect of aeration on TP concentrations has contradicting results in the literature. Gronchowska and Gawrońska (2004) showed that phosphorus compounds decrease in water column with introduction of aeration however other studies conclude that TP did not change (e.g. Jungo *et al.,* 2001), or even showed year to year variation. Cowell *et al.* (1977) demonstrated that TP decreased significantly with aeration in year two of study but then TP showed unexpected increases in year three. Similarly, in Lakes Waccabuc and Oscaleta (New York) showed that TP decreased with aeration in the first year but in the second year showed significant increases in TP (Garrell *et al.,* 1977). Gächter and Wehrli (1998) demonstrated from long-term monitoring on several Swiss lakes that the phosphorus concentrations were a result of external loading and increased oxygenation

had no net sedimentation effect. Similarly, sediment accumulation and organic content did not decline several lakes in Minnesota with aeration and the only reduction occurred due to decrease in external loading of nutrients (Engstrom and Wright, 2002).

Sediment exchange of phosphorus is complex and controlled by many different chemical and biological processes and thus should no longer be viewed as a simple paradigm that oxygen alone controls the release of TP from sediments (Hupfer and Lewandowski, 2008). Summer 2013 had high levels of precipitation especially in Inlet Oaks sites. However, this should be considered carefully since precipitation data was not collected directly and at the end of 2013 (August – December) sampling often was biased to occur during 72 hours of dry weather when the pond was not flushing. However, still high TP concentrations could have been entirely driven by external loading into the ponds rather than internal phosphorus loading (e.g. from the sediments). Therefore this could have resulted in the high chl-*a* concentrations matched by high TP concentrations seen in Inlet Oaks in 2013. Artificial aeration shown by Heo and Kim (2004) was not effective in preventing seasonal high TP and chl-*a* concentrations which was a product of monsoon-related inputs. However, even with the rain events not all nutrients, like TN, increased. Despite the ponds being ideal candidates for a BACI design as exemplified in South Bay that featured the same arrangement of infrastructure and impervious surface surrounding the area there was still environmental variability within the control and the impacted sites. Nutrient sources within freshwater sources can vary and can result in diverse N:P ratios with urban stormwater drainage and runoff having lower N:P ratios (Downing and McCauley, 1992).

The Inlet Oak sites were deeper than South Bay sites and therefore the phytoplankton could have faced greater light limitation. TSS increases seen in the aeration site were matched by increases in GPP and higher chl-*a* values than the previous year when k_d had highest values. By enhancing the water column mixing with aeration, IOA could have promoted an increase in phytoplankton biomass production with depth because of an increase in light availability (e.g. Huisman *et al.,* 2004). The effects of aeration have been shown to be variable on chl-*a* biomass and primary productivity with aeration. Chlorophyll-*a* biomass integrated over depth has seen to increase with aeration in other studies (e.g. Heo and Kim, 2004) and Fast *et al.* (1973) demonstrated almost a 3-fold increase in primary production values due to an increased in availability of nutrients from mixing. Primary production could have been effected in the aerated pond of Inlet Oaks due to the high input of TP which could have stimulated the greater response in biomass and production. In South Bay, primary production remained similar to the previous year and aeration had no significant effect on chl-*a* or primary production. In SBB a high chl-*a* concentrations occurred that were not matched by high primary production values when k_d values were at the highest. Photoacclimation to low light can result in an increase in cellular chl-*a* content (e.g. Falkowski, 1980) and can appear to be an increase in biomass. However if only light availability was the factor in regulating primary production we should see an increase in SBA sites and there would not be high GPP values in IOB. Therefore, the control of nutrients rather than light most likely would play more of a role in sequestering or decreasing total biomass and primary production in these ponds.

Phytoplankton community composition varied seasonally and spatially within the sites. Chlorophytes and euglenophytes dominated summer 2012 with late summer cyanobacteria blooms then diatoms and chrysophytes blooming during the winter months. However, in 2013, the South Bay sites developed cyanobacteria and cryptophytes and dinoflagellates, but IOA was dominated by chlorophytes and euglenophytes and IOB typically featured large amounts of chrysophytes and diatoms. Cyanobacteria dominance is typically a sign of advanced freshwater eutrophication (Paerl and Ustach 1982). They can pose a major human health concern with production of cyanotoxins leading to poisonings of wild/ domestic animals and humans (Carmichael, 2001) and thus have become the focus of many freshwater aeration studies in eutrophic lakes and reservoirs (e.g. Downing *et al.,* 2001). Cowell *et al.* (1987) showed that aeration increased the frequency of occurrence in cryptophytes and chrysophytes bloom while diatoms, dinoflagellates and euglenophytes showed no increase, however with a significant decrease in water pH there was a reduction in cyanobacteria. Cyanobacteria typically do well under stratified and hot (> 25°C) conditions (Paerl and Huisman, 2008) where aeration, from physical and chemical changes, stimulates other groups such as diatoms or chlorophytes (Steinberg, 1983).

However in our study mixing in the ponds did not promote the decrease in cyanobacteria but did stimulate the increase of dinoflagellates and cryptophytes at the expense of diatoms and chrysophytes which typically were not abundant in summer. The effect of aeration on other cyanobacteria populations have varied in other research as well where *Microcystis sp.* populations can drastically decline with deep mixing (Jungo and Visser, 2001). However, in shallow strong mixing conditions concentrations can increase

(Lindenschmidt and Chorus, 1997) while remaining more tolerant to mixing then competitor *Ceratium sp*. (Lindenschmidt and Chorus, 1998) but have also been shown to be suppressed with pulsed aeration (Lilndenschmidt, 1999). Mixing alone from aeration has proven to be insufficient to suppress blooms of *Microcystis sp.* in other areas (e.g. Oberholster *et al.,* 2006) and similarly prevailed in South Bay with aeration.

Light and external nutrient loading into the ponds played more of a role in the community composition rather than the effect of aeration. Antenucci *et al.* (2005) showed that with aeration increased diatoms, cyanobacteria, and chlorophytes in a dam where the cyanobacteria, *Cylindrospermopsis raciborskii,* was able to outcompete competitors in light as mixing enhanced prolonged light exposure. Shallow ponds (<15 m) with continuous aeration often enhance cyanobacteria production since there is still enough light to not cause limitation and they are able to utilize phosphorus throughout the water column (Lilndenschmidt, 1999). Cyanobacteria can also replace diatom communities with aeration when concentrations of Si are depleted (Hawkins and Griffiths, 1992) which diatoms need for frustules. NH_4^+ have been shown to preferentially stimulate cyanobacteria growth in South Carolina (Siegel *et al.*, 2011). Concentrations of NH_4^+ were high in SBA during August 2102 and October 2012 in IOA which corresponded to the onset of late summer blooms of cyanobacteria. Similarly, NH_4^+ was higher in SBA until March and higher cyanobacteria contributions were seen higher until then.

Nitrogen to phosphorus ratios have also been shown to be good indicators of phytoplankton community composition in freshwater (e.g. Vrede *et al.*, 2009). Smith (1983) determined that a TN:TP mass ratio < 29:1 will typically result in N-fixing cyanobacteria dominance within a lake. As phosphorous loading increases, there is a shift

towards becoming more nitrogen limited within a system. Low TN:TP ratios occurred during late summer in all of the ponds where blooms of N-fixing cyanobacteria (*Anabaena sp.*) occurred. However, during 2013, TN:TP ratios were low in the Inlet Oaks sites and should have favored more N-fixing cyanobacteria. Calm stratified conditions have been seen to favor cyanobacteria growth (Havens *et al.*, 2003) However, since the ponds had more rain events during 2013 and IOA had enhanced mixing this could have played a role in preventing *Anabaena sp.* blooms from occurring while *Microcystis sp.* made greater contributions.

Functional groupings proposed by Reynolds *et al.* (2002) classified *Dinobryon* and *Synura* (in group E) as able thrive under lower nutrient conditions in heterotrophic ponds. *Dinobryon sp.* and *Synura sp*. was found in very high numbers in IOB especially during 2013 corresponding to high inputs of SRP, TP, and DOC but DIN concentrations were often near minimum detection limits. *Dinobryon* growth is hindered with low concentrations of phosphate and the ability of phagotrophy allows for additional supply of nutrients when inorganic nutrients are low as demonstrated by Jones (2000). However, during this time in the aerated pond (IOA) *Eudorina sp., Pediastrum sp., and Scenedesmus sp.* were dominant and classified as groups G and J (Reynolds *et al.*, 2002) which thrive under nutrient and organic rich water columns. The OSS, TN, and TP concentrations were all much higher in this pond and mixing reduced sensitivities of settling into low light.

CHAPTER 5: CONCLUSION

Overall, aeration did improve bottom water DO concentrations by breaking down thermal stratification but it did not lead to an overall reduction in nutrient concentrations, rates of primary productivity, and phytoplankton biomass or community composition within the ponds. Changes in biomass within one of the aerated sites and to community composition were probably the product of external nutrient loading and land management use surrounding the ponds rather than aeration itself. Mixing associated with aeration led to a prolonged exposure to light, uniform nutrients within the water column, and did not suppress nuisance blooms from still occurring. The use of aeration as an alternative to algaecide within South Carolina stormwater ponds is ultimately ineffective and could perhaps exacerbate biomass accumulation. Many South Carolina coastal stormwater ponds are facing eutrophic symptoms and are the sources of biological and chemical pollutants draining directly or indirectly into the coastal zone and have major implications for coastal water quality. Therefore, there is a need to improving land use management practices in order to reduce external nutrient loading before making it into the ponds.

REFERENCES

Anderson, D. M., Glibert, P. M., and Burkholder, J. M. (2002). Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. *Estuaries*, 25, 704-726.

Antenucci, J. P., Ghadouani, A., Burford, M. A., and Romero, J. R. (2005). The long-term effect of artificial destratification on phytoplankton species composition in a subtropical reservoir. *Freshwater Biology*, 50, 1081-1093.

Benner, R., and Strom, M. (1993). A critical evaluation of the analytical blank associated with DOC measurements by high-temperature catalytic oxidation. *Marine Chemistry*, 41, 153-160.

Beutel, M. W., and Horne, A. J. (1999). A review of the effects of hypolimnetic oxygenation on lake and reservoir water quality. *Lake and Reservoir Management*, 15, 285-297.

Boyd, C. E. (1998). Pond water aeration systems. *Aquacultural Engineering*, 18, 9-40.

Boström, B., Anderson, J.M., Fleischer, S., and Jansson, M. (1988) Exchange of phosphorus across the sediment-water interface. *Hydrobiologia*, 179, 229-244.

Carmichael, W.W. (2001) Health effects of toxin-producing cyanobacteria: "the cyanoHABs. *Human and Ecological Risk Assessment* 7, 1393-1407.

Cooke, G.D., and Kennedy, R.H. (2001) Managing drinking water supplies. *Lake and Reservoir Management*, 17, 157-174.

Cowell, B. C., Dawes, C. J., Gardiner, W. E., and Scheda, S. M. (1987). The influence of whole lake aeration on the limnology of a hypereutrophic lake in central Florida. *Hydrobiologia*, *148*, 3-24.

Crawford, K. D., Weinstein, J. E., Hemingway, R. E., and Garner, T. R., and Globensky, G. (2010). A survey of metal and pesticide levels in stormwater retention pond sediments in coastal South Carolina. *Archives of Environmental Contamination and Toxicology*, *58*, 9-23.

Drescher S., Messersmith, M., Davis, B., and Sanger, D. (2007) State of knowledge: stormwater ponds in the coastal zone. *South Carolina Department of Health and Environmental Control – Ocean and Coastal Resource Management* http://www.gcrc.uga.edu/PDFs/StormwaterPondsSOK.pdf

DeLorenzo, M. E., Thompson, B., Cooper, E., Moore, J., and Fulton, M. H. (2012). A long-term monitoring study of chlorophyll, microbial contaminants, and pesticides in a coastal residential stormwater pond and its adjacent tidal creek. *Environmental Monitoring and Assessment*, *184*, 343-359.

DeLorenzo M.E., and Fulton, M.H. (2009) Water quality and harmful algae in southeastern coastal stormwater ponds. NOAA Technical Memorandum NOS NCCOS 93, 27 pp. http://aquaticcommons.org/2147/1/NCCOS_TM_93.pdf

Diaz, R.J., and Rosenberg, R. (2008) Spreading dead zones and consequences for marine ecosystems. *Science*, 321, 926-929.

Downing, J.A., and McCauley, E. (1992). The nitrogen: phosphorus relationship in lakes. *Limnology and Oceanography*, 37, 936-945.

Downing, J.A., Watson, S.B., and McCauley, E. (2001). Predicting cyanobacteria dominance in lakes. *Canadian Journal of Fisheries and Aquatic Sciences*, 58, 1905-1908.

Engstrom, D.R., and Wright, D.I. (2002). Sedimentological effects of aeration-induced lake circulation. *Lake and Reservoir Management*, 18, 201-214.

Falkowski, J.P. (1980) Light-shade adaptation in marine phytoplankton. In: Primary Productivity in the Sea (Ed. Falkowski, J.P.). Plenum Press, New York.

Fast, A.W., Moss, B., and Wetzel, R.G. (1973) Effects of artificial aeration on the chemistry and algae of two Michigan lakes. *Water Resources Research*, *9*, 624-647.

Field, K.M., and Prepas, E.E. (1997) Increased abundance and depth distribution of pelagic crustacean zooplankton during hypolimnetic oxygenation in a deep, eutrophic Alberta lake. *Canadian Journal of Fisheries and Aquatic Sciences*, *54*, 2146-2156.

Garrell, M. H., Confer, J. C., Kirschner, D., and Fast, A. W. (1977) Effects of hypolimnetic aeration on nitrogen and phosphorus in a eutrophic lake. *Water Resources Research*, *13*, 343-347.

Gätcher, R., and Wehrli, B. (1998) Ten years of artificial mixing and oxygenation: no effect on the internal phosphorus loading of two eutrophic lakes. *Environmental Science and Technology*, 32, 3659-3665.

Gibson, C.E. (1972) The algicidal effect of Copper on a green and a blue-green alga and some ecological implications. *Journal of Applied Ecology*, 9, 513-518.

Glibert, P.M., and Loder, T.C. (1977) Automated analysis of nutrients in seawater: A manual of techniques. Woods Hole Oceanographic Institution Technical Report 77, pp. 47

Gilbert, P.M, Magnien, R., Lomas, M.W, Alexander, J., Fan, C., Haramoto, E., Trice, M., and Kana, T.M. (2001) Harmful algal blooms in the Chesapeake and coastal bays of Maryland USA: comparison of 1991, 1998, and 1999 events. *Estuaries,* 24, 875-883.

Grochowska, J., and Gawrońska, H. (2004) Restoration effectiveness of a degraded lake using multi-year artificial aeration. *Polish Journal of Environmental Studies,* 13, 671-681.

Havens, K. E., James, R.T., East, T.L., and Smith, V.H. (2003). N: P ratios, light limitation, and cyanobacterial dominance in a subtropical lake impacted by non-point source nutrient pollution. *Environmental Pollution, 122,* 379-390.

Hawkins, P.R., and Griffiths, D.J. (1992) Artificial destratification of a small tropical reservoir: effects upon the phytoplankton. *Hydrobiologia,* 254, 169-181.

Hedges, J.I, and Stern, J.H (1984) Carbon and nitrogen determinations of carbonate-containing solids. *Limnology and Oceanography,* 29, 657-663.

Heo, W.M, and Kim, B. (2004) The effect of artificial destratification on phytoplankton in a reservoir. *Hydrobiologia,* 524, 229-239.

Hupfer, M., and Lewandowski, J. (2008) Oxygen controls the phosphorus release from lake sediments a long lasting paradigm in limnology. *International Review Hydrobiologia,* 93, 415-432.

Huisman, J., Sharples, J., Stroom, J.M, Visser, P.M, Karardinaal, W.E.A, Verspagen, J.M, and Sommeijer, B. (2004) Changes in turbulent mixing shift competition for light between phytoplankton species. *Ecology,* 85, 2960-2970.

Jones, R.I. (2000) Mixotrophy in planktonic protists: an overview. *Freshwater Biology,* 45, 219-226.

Jungo, E., Visser, P. M., Stroom, J., and Mur, L. R. (2001). Artificial mixing to reduce growth of the blue-green alga Microcystis in Lake Nieuwe Meer, Amsterdam: an evaluation of 7 years of experience. *Water Science and Technology: Water Supply*, 1, 17-23.

Knap, A. H., Michaels, A., Close, A. R., Ducklow, H., and Dickson, A. G. (1996). Protocols for the joint global ocean flux study (JGOFS) core measurements. *JGOFS, Report No 19. Reprint of the IOC Manuals and Guides No. 29, UNESCO 1994*.

Le Jeune, A. H., Charpin, M., Deluchat, V., Briand, J. F., Lenain, J. F., Baudu, M., and Amblard, C. (2006). Effect of copper sulphate treatment on natural phytoplanktonic communities. *Aquatic Toxicology*, *80*, 267-280.

Lewis, M.R, and Smith, J.C. (1983) "A small volume, short-incubation-time method for measurement of photosynthesis as a function of incident irradiance." *Marine Ecology Progress Series,* 13, 99-102.

Lewitus, A.J., Schmidt, L.B., Mason, L.J., Kempton, J. W., Wilde, S. B., Wolny, J. L., Williams, B.J., Hayes, K.C., Hymel, S.N., Keppler, C.J., and Ringwood, A. H. (2003). Harmful algal blooms in South Carolina residential and golf course ponds. *Population and Environment*, 24, 387-413.

Lewitus, A. J., Brock, L. M., Burke, M. K., DeMattio, K.A., and Wilde, S.B. (2008). Lagoonal stormwater detention ponds as promoters of harmful algal blooms and eutrophication along the South Carolina coast. *Harmful Algae, 8,* 60-65.

Lilndenschmidt, K.E. (1999) Controling the growth of Microcystis using surged artificial aeration. *International Review Hydrobiologia*, 84, 243-254.

Lindenschmidt, K.E, and Chorus, I. (1997). The effect of aeration on stratification and phytoplankton populations in Lake Tegel, Berlin. *Archiv für Hydrobiologie, 139,* 317-346.

Lindenschmidt, K., and Chorus, I. (1998) The effect of water column mixing on phytoplankton succession, diversity, and similarity. *Journal of Plankton Research,* 20, 1927-1957.

Mackey, M.D., Mackey, D.J., Higgins, H.W., and Wright, S.W. (1996) CHEMTAX- a program for estimation class abundances from chemical markers: application to HPLC measurements of phytoplankton. *Marine Ecology Progress Series,* 144, 265-283.

Mallin, M.A, and Wheeler, T.L. (2000) Nutrient and fecal coliform discharge from coastal North Carolina golf courses. *Journal of Environmental Quality,* 29, 979-986.

Müller, R., and Stadelmann, P. (2004) Fish habitat requirements as the basis for rehabilitation of eutrophic lakes by oxygenation. *Fisheries Management and Ecology,* 11, 251-260.

Oberholster, P.J., Botha, A.M., and Cloete,bT.E. (2006) Toxic cyanobacterial blooms in a shallow, artificially mixed urban lake in Colorado, USA. *Lakes & Reservoirs: Research & Management*, 11, 111-123.

Padovesi-Fonseca, C., and Philomeno, M. G. (2004) Effects of algicide (Copper Sulfate) application on short-term fluctuations of phytoplankton in Lake Paranoá, Central Brazil. *Brazilian Journal of Biology,* 64, 819-826.

Paerl, H.W, and Ustach, J. (1982) Blue-green algal scums: an explanation for their occurrence during freshwater blooms. *Limnology and Oceanography,* 27, 212-217.

Paerl, H.W. (1988) Nuisance phytoplankton blooms in coastal, estuarine, and inland waters. *Limnology and Oceanography*, 33, 823-847.

Paerl, H.W, and Huisman, J. (2008) Blooms like it hot. *Science*, 320, 57-58.

Pinckney, J.L., Millie, D.F., Howe, K.E., Paerl, H.W., Hurley, and J.P. (1996) Flow scintillation counting of 14C-labeled microalgal photosynthetic pigments. *Journal of Plankton Research*, 18, 1867-1880.

Pinckney, J. L., Paerl, H. W., Tester, P., and Richardson, T. L. (2001) The role of nutrient loading and eutrophication in estuarine ecology. *Environmental Health Perspectives*, 109, Supplement 5, 699.

Platt, T., and Gallegos, C.L. (1980) Modelling primary production. In *Primary Productivity in the Sea* (ed. P. Falkowski), pp. 339-362. New York: Plenum Press.

Prairie, Y.T., Bird, D.F., and Cole, J.J. (2002) The summer metabolic balance in the epilimnion of southeastern Quebec lakes. *Limnology and Oceanography*, 47, 316-321.

Reynolds, C.S. (1984) The ecology of freshwater phytoplankton. Cambridge University Press, Cambridge.

Reynolds, C.S., Huszar, V., Kruk, C., Naselli-Flores, L., and Melo, S. (2002) Towards a functional classification of the freshwater plankton. *Journal of Plankton Research*, 24, 417-428.

Rysgaard, S., Risgaard-Petersen, N., Sloth, N.P., Jensen, K., Nielsen, L.P. (1994) Oxygen regulation of nitrification and denitrification in sediments. *Limnology and Oceanography*, 39, 1643-1652.

SCDHEC (2012) Water Classifications and Standards (Regulation 61-68) and Classified Waters (Regulation 61-69) for the State of South Carolina. Office of Environmental Quality Control, Columbia, SC

Serrano, L., and DeLorenzo, M. E. (2008) Water quality and restoration in a coastal subdivision stormwater pond. *Journal of environmental management*, *88*, 43-52.

Schlüter, L., Lauridsen, T.L., Krogh, G., and Jørgensen, T. (2006) Identification and quantification of phytoplankton groups in lakes using new pigment ratios- a comparison between pigment analysis by HPLC and microscopy. *Freshwater Biology*, 51, 1474-1485.

Siegel, A., Cotti-Rausch, B., Greenfield, D.I., and Pinckney, J.L. (2011) Nutrient controls of planktonic cyanobacteria biomass in coastal stormwater detention ponds. *Marine Ecology Progress Series,* 434, 15-27.

Smith, E.P (2002) BACI design. In: Encyclopedia of Environmetrics (eds. El-Shaarawi AH, Piegorsch, WW), pp 141–148. Wiley, Chichester.

Smith, E.M., and Kemp, W.M. (2003) Planktonic and bacterial respiration along an estuarine gradient: responses to carbon and nutrient enrichment. *Aquatic Microbial Ecology*, 30, 251-261.

Smith, V.H. (1983). Low nitrogen to phosphorus ratios favor dominance by blue-green algae in lake phytoplankton. *Science*, 221, 669–671.

Smith, V. H. (2003). Eutrophication of freshwater and coastal marine ecosystems a global problem. *Environmental Science and Pollution Research*, 10, 126-139.

Steinberg, C. (1983) Effects of artificial destratification on the phytoplankton populations in a small lake. *Journal of Plankton Research,* 5, 855-864.

Van Dolah, R. F., Riekerk, G. H., Bergquist, D. C., Felber, J., Chestnut, D. E., and Holland, A. F. (2008). Estuarine habitat quality reflects urbanization at large spatial scales in South Carolina's coastal zone. *Science of the Total Environment, 390*, 142-154.

Vrede, T., Ballantyne, A., Mille-Lindblom, C., Algesten, G., Gudasz, C., Lindahl, S., and Brunberg, A.K. (2009). Effects of N: P loading ratios on phytoplankton community composition, primary production and N fixation in a eutrophic lake. *Freshwater Biology, 54*, 331-344.

Welschmyer, N. (1994) Fluorometric analysis of chlorophyll-α in the presence of chlorophyll-β and pheopigments. *Limnology and Oceanography,* 38, 1985-1992.

Wetzel, R.G. (2001). Lake and river ecosystems. Academic Press, San Diego, CA.

Zhu, W.X., Dillard, N.D., and Grimm, N.B. (2004) Urban nitrogen biogeochemistry: status and processes in green retention basins. *Biogeochemistry,* 71, 177-196.