

Using Phytoplankton as the Focal Point for the Management of Cardiff Bay, South Wales



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SUMMARY

This study is the first to demonstrate a novel combination of hydrological and biological control of a freshwater lake system. The former through lake flushing rate and the latter through top-down control by an invasive filter-feeder; zebra mussels. The artificial freshwater lake of Cardiff Bay, South Wales was created in 2001 by the implementation of an amenity barrage across the Taff-Ely estuary. To date, management methods include an aeration system to maintain dissolved oxygen concentrations within the water column, and a reduction in phosphorus concentration from river inputs to control phytoplankton biomass. Despite this ongoing management, the state of the phytoplankton population within the lake in relation to water quality has not been analysed in depth since its formation. This study undertook an extensive assessment of the hydrological, chemical and biological variables in the lake to determine the controls on the phytoplankton community, and the implications it has for water quality management of the system. Cardiff Bay is a unique site that acts more like a river system than a lake. Its' polymictic nature prevents the lake from stratifying, and its low phytoplankton biomass is controlled by fast flushing rates and filter-feeding by zebra mussels. The lake supports a healthy phytoplankton population that is not nutrient limited and does not contribute to dissolved oxygen concentrations. This study has highlighted the unique relationships that take place within Cardiff Bay, and the need for an advanced understanding stemming from this research to ensure that management can take place to successfully maintain the clear water state of the lake, under the stressors of changes in climate.

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INTRODUCTION

1.1 Summary

Freshwater lakes are extensively relied upon by humans and organisms. They provide an important habitat for plants and animals, as well as supplying man with clean water and recreational areas. The physical, chemical and biological properties of every lake is unique, making it essential to understand the mechanisms and relationships that take place within them. This is particularly important for water bodies that have been artificially altered, to ensure that the ecosystem is healthy. Phytoplankton play a significant role in lake food-webs and their community structure has the ability to influence higher trophic levels, making phytoplankton valuable indicators of lake water quality. The hydrological, nutrient and biological controls on the phytoplankton community within Cardiff Bay need to be determined to ensure that management of the lake can be effectively implemented, particularly under the pressure of a changing climate.

1.2 The Requirement for Lake Management

Ongoing climate change and increasing anthropogenic activity is causing ecosystems to experience profound changes in structure and function (Walther et al., 2002, Spears et al., 2015, Burthe et al., 2016, Mantzouki et al., 2018). These changes have been documented within all well-studied ecosystems including freshwater lakes and have all been linked to local or regional climate change (Parmesan, 2006). As a result, lake ecosystem resilience is being constantly challenged and sudden ecological changes are occurring in response to the increased pressures and management activities that lakes find themselves exposed to (Spears et al., 2015, Mantzouki et al., 2018). Lake management must be conducted with a comprehensive understanding of the multiple factors and underlying processes that drive ecological changes

under current climatic conditions to be effective (Alpine & Cloern, 1992, Spears et al., 2015, Kumari et al., 2018).

A lake is classified as a body of water larger than 2 ha in area which has no direct exchange with an ocean (Bhateria & Jain, 2016, Freshwater Habitats Trust, 2017). Lakes have always been an important water resource for man and in more recent times have been recognised as having the ability to add to the perceived beauty of landscapes and thus the prestige and status of an area (Rees, 1997). There are 448 lakes in Wales, 90 of which have been created by man. Humans depend on these natural and man-made lakes as they provide a range of services including; drinking water, waste removal, fisheries, agricultural irrigation, transport, communication, defence, industrial activity, to provide power, for recreation, and to create a picturesque landscape (Rees, 1997, Bhateria & Jain, 2016, Welsh Government, 2017).

Lakes are also important water resources for organisms (Bhateria & Jain, 2016). They not only provide a habitat for plants and animals that require a permanent body of water but they can provide vital breeding grounds for waterfowl in the form of wetlands (Freshwater Habitats Trust, 2017). The health of a river-lake system is maintained by water and mass exchange, therefore any alterations to these interactions can have a significant impact on the water and mass balance of the ecosystem (Yang et al., 2016). The exchange of water, sediments, dissolved constituents, and energy between a river and a lake influences the geomorphology, water regime, water quality, and the aquatic and riparian ecosystems (Yang et al., 2016). At present, lakes face multiple stressors such as pollution, eutrophication and climate change, which all have the potential to negatively influence a lake ecosystem (Adrian et al., 2009a, Mantzouki et al., 2018).

When managing lake ecosystems, climate variability and anthropogenic impacts must be considered. Humans have a strong influence on almost every major aquatic ecosystem around the world and have dramatically altered the nutrient fluxes of lake systems. This often results in a negative effect on the quality of surface waters by increasing phytoplankton biomass which can result in eutrophication (Smith, 2003). In addition, all water management practices are, to an extent, a response to natural hydraulic variability and include adaptation methods to climate change. The present challenge for lake management is the ability to form practices which can cope with the future variability caused by climate change that will ultimately influence lake ecosystems (De Loe & Kreutzwiser, 2000).

1.2.1 Lake Ecosystems

A lake ecosystem is comprised of physical, chemical and biological properties. Although lakes of all types share many ecological and biogeochemical processes (Bhateria & Jain, 2016), every lake system and the interactions within them are unique. This makes understanding the complex interactions between nutrients, temperature and physical forces within a lake and the changes they can cause to the trophic structure, paramount for the implementation of effective management (Battarbee, 2000, Jeppesen et al., 2009).

Lakes normally consist of four zones; the littoral zone, the limnetic zone, the profundal zone and the benthic zone. (Figure 1.1). The shallow, nutrient rich waters near the shore which often contain rooted aquatic plants makes up the littoral zone; an abundance of aquatic life is often present here. The limnetic zone is the open-water surface layer of a lake which receives enough sunlight for photosynthesis, and contains phytoplankton, zooplankton and fish depending on the availability of plant nutrients. In lakes deep enough to contain a zone of deep water not penetrated by sunlight, the profundal zone is present. This is often inhabited by fish which are adapted to the cooler, darker waters and lower dissolved oxygen levels. The benthic zone of a lake is the deepest zone, located at the bottom of the lake and is inhabited by large numbers of bacteria, fungi and decomposers which live on dead plant debris, animal remains and animal wastes which sink through the water column (Bhateria & Jain, 2016).

Ecological indicators can be used to assess the health of a lake ecosystem as they are complex, and meteorological changes often alter internal feedback mechanisms (Xu et al., 2001, Adrian et al., 2009a). These indicators include the response of lakes to chemical stresses such as eutrophication, and structural indicators such as phytoplankton and zooplankton biomass (Xu et al., 2001). Shallow lake ecosystems are usually more responsive to sudden changes in meteorological events therefore understanding the seasonality of such effects and how these indicators respond is important to manage a lake successfully (Baulch et al., 2005, Blank et al., 2009).

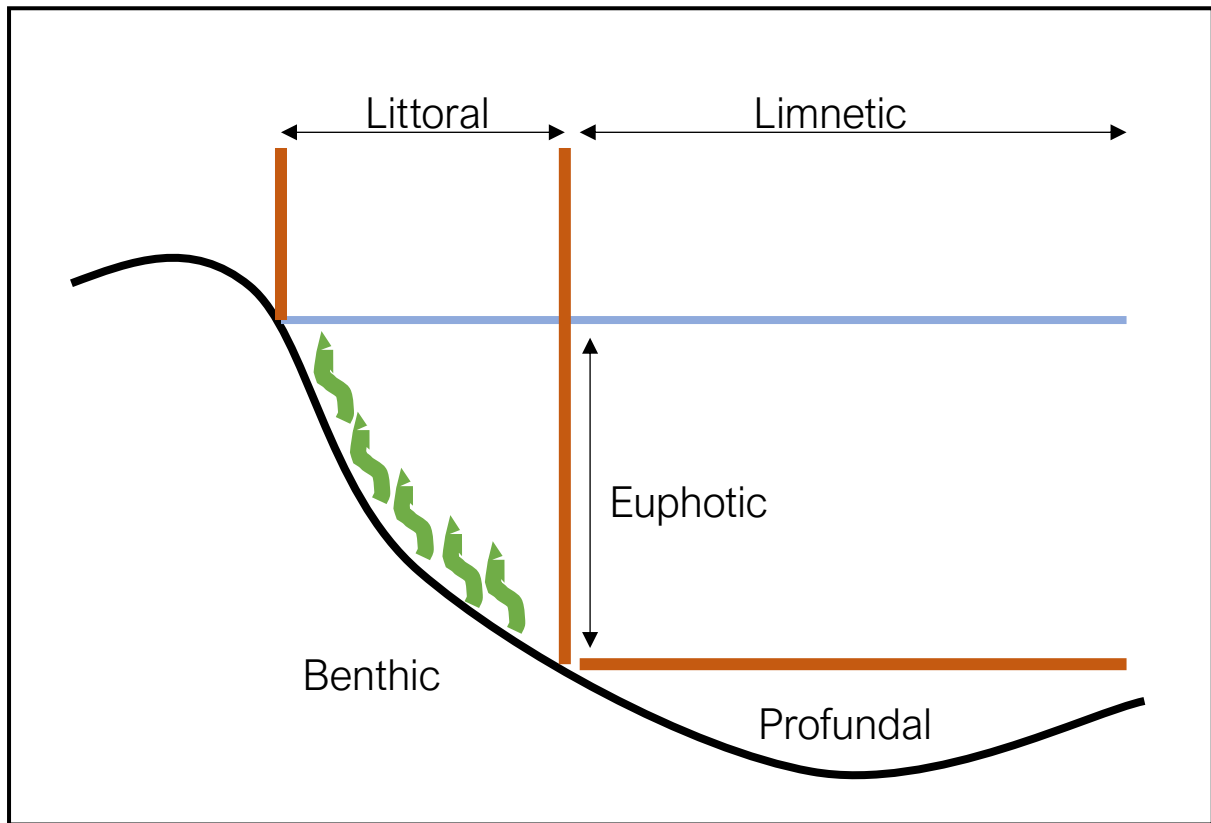


Figure 1.1: A simple schematic of characteristic lake zonation.

1.2.2 The Role of Phytoplankton in a Lake

Phytoplankton are essential for freshwater ecological and biogeochemical processes. They play a significant role in a lake as they form the basis of nearly all food-webs within aquatic ecosystems (Kruk et al., 2011). By converting solar energy to chemical energy and storing it as organic matter, phytoplankton provide energy to entire lake trophic systems (Peeters et al., 2007, Kumari et al., 2018). This transfer of energy is influenced by the composition and productivity of phytoplankton and it can determine suitable growth conditions for zooplankton and ultimately influence higher trophic organisms such as fish (Winder & Schindler, 2004, Anneville et al., 2005b, Castaneda et al., 2009, Adjou et al., 2012).

Alterations in the phytoplankton community structure of a lake can influence, as well as be caused by, nutrient cycling, invasive species colonisation and lake trophic structure, all of which have an impact on water quality. This makes phytoplankton valuable indicators, not only of water quality but also of the environmental stressors which act on aquatic ecosystems and how they vary (Anneville et al., 2005b, Adjou et al., 2012, Bhateria & Jain, 2016). Under required conditions, phytoplankton can grow excessively and cause harmful algal blooms (HABs), thus causing significant threats to local biodiversity and ecosystem functioning (Kruk et al., 2011).

There is a growing need to understand and predict the responses of phytoplankton communities to changing environmental conditions such as climate change, increasing nutrient inputs, and modifications to flow regimes as excessive phytoplankton growth causes water quality degradation (Reynolds, 1999, Kruk et al., 2011).

1.3 Environmental Influences on Lakes

Lakes are influenced by environmental parameters. The most influential of which is often the combination of water temperature and irradiance, accompanied by sufficient phosphorus availability. These parameters often regulate phytoplankton biomass and crucially for lake water quality, the presence of cyanobacteria (Millie et al., 2009).

1.3.1 Lakes and Temperature

Air temperature is one of the most important physical drivers within a lake ecosystem (Blenckner et al., 2007a); a rise in temperature not only increases evaporation but it can cause the water holding capacity of air to rise by 7 % per 1 °C warming, leading to increased water vapour in the atmosphere and potentially resulting in intense precipitation events (Trenberth, 2011). Over the past century, global mean surface air temperatures have increased by 0.74 °C, with the majority of the increase (0.55 °C) taking place over the past 30 years (Baulch et al., 2005, Jeppesen et al., 2009, Carey et al., 2012, Defra, 2012c). This has made changes in climate, specifically air temperature, one of the most severe threats to ecosystems around the world (Solomon et al., 2007, Adrian et al., 2009a). It has long been understood that there is an

linear relationship between air temperature and surface water temperatures in lakes (Livingstone & Lotter, 1998), despite this, air temperature changes and how lakes respond to them are relatively understudied (Austin & Colman, 2007). However, how lakes respond to changes in air temperature is becoming increasingly important as air temperatures have been predicted to rise in the future as a result of global warming, potentially making a change in air temperature critical for lake systems.

Water temperature changes occur within a lake as a result of weather, bank vegetation, anthropogenic discharge of water, urban storm water, and groundwater inflow (Bhateria & Jain, 2016). A rise in water temperature can strongly affect the water column of a lake, particularly during spring and summer by increasing the duration and stability of thermal stratification. This stratification has a knock-on effect on chemical and biological lake processes including nutrient cycling, oxygen concentrations and primary production (Battarbee, 2000, Winder & Schindler, 2004, Griffiths et al., 2011, Carey et al., 2012). Therefore, understanding the thermal structure of a lake is essential for the quantification of vertical exchanges within the water column. This is particularly important as increased warming of the epilimnion creates a stable water column containing underlying colder waters, therefore having the ability to alter exchanges within the water column and impacting the trophic organisation within a lake (Bonnet et al., 2000, Winder & Schindler, 2004, Adrian et al., 2009a).

1.3.2 Lakes and Precipitation

Precipitation patterns are highly variable from year to year but can be influenced by changes in climate. An increasing trend in precipitation has been observed during winter due to a rise in atmospheric warming (Solomon et al., 2007). Precipitation can alter lake discharge rates, particularly during extreme precipitation events where an increase in precipitation causes discharge to increase considerably. These events also increase the transport of terrestrial material into lakes, increasing organic input and fuelling heterotrophic processes (Sadro & Melack, 2012). Whilst it is known that precipitation can determine productivity within a lake ecosystem, the impact of changes in precipitation are relatively understudied (Galloway & Cowling, 1978, Jones & Elliott, 2007).

1.4 Phytoplankton

The phytoplankton community of a lake increases in complexity from winter to summer, in the annual plankton succession (Peeters et al., 2007). Seasonal variations in water temperature, irradiance and nutrient availability causes seasonal changes in phytoplankton biomass and assemblage to occur as a result. During spring phytoplankton growth often increases abruptly, determining the start of phytoplankton succession for the year. This is followed by a decline in phytoplankton biomass during autumn-winter as a result of a reduction in solar radiation and an increase in vertical mixing (Talling, 1971, Richardson et al., 2000, Peeters et al., 2007).

The structure of a food-web can also be a powerful factor in determining phytoplankton abundance and productivity in a lake. Phytoplankton abundance can be altered through grazing, which can cause a variation in productivity often unexplained by nutrient loading or phytoplankton standing stock and is often dependant on zooplankton grazing (Elser & Goldman, 1991). Zooplankton biomass often peaks between mid-spring and mid-summer and is positively related to phytoplankton biomass (Mccauley & Kalff, 1981, Zervoudaki et al., 2009). However, cyanobacteria-dominated phytoplankton populations can be resistant to zooplankton grazing (Elser & Goldman, 1991).

The dominance of a phytoplankton species within a lake often depends on a combination of factors. These include physical factors such as temperature and light, as well as the availability of nutrients and the composition of the zooplankton community (Huszar et al., 2003). Therefore, it is important to understand the drivers of phytoplankton community composition, not just the total biomass (Kruk et al., 2011).

1.4.1 Phytoplankton Community Composition

The phytoplankton composition in a lake is a key indicator for lake trophic status. Phytoplankton community composition is influenced by physical process such as stratification and temperature, as well as nutrient availability and grazing (Ryan et al., 2006, Fishman et al., 2010). Therefore, understanding the abundance and variation of phytoplankton species within a lake can provide a valuable insight into lake processes.

Chlorophytes are the most species-rich and morphologically diverse group of phytoplankton, allowing them to adapt to a broad range of environmental conditions and they often dominate lake systems with high total phosphorus concentrations ($> 1 \text{ mg P L}^{-1}$) (Belcher & Swale, 1976, Jensen et al., 1994, Salmaso, 2000, John et al., 2011).

Diatoms represent another diverse group of phytoplankton, they are a major component of many lake food-webs, often dominating the phytoplankton community, particularly in mesotrophic lakes (Montagnes & Franklin, 2001) (Duarte et al., 1992, Resende et al., 2005). In temperate lakes, diatoms also tend to dominate the spring phytoplankton maximum: one of the most important seasonal events (Reynolds, 1973, Neale et al., 1991), where accelerated growth is controlled by light availability and water column mixing. In contrast, a decline in diatom growth is often associated with a reduction in silica concentrations.

Cyanobacteria significantly represent a substantial component of virtually all photosynthetic assemblages, allowing them to colonise lakes over the whole trophic range (Salmaso, 2000). This phytoplankton group are of particular importance within lake systems as, unlike other phytoplankton, they have the ability to proliferate intensively under several environmental conditions and form blooms of extremely high cell density (World Health Organization, 2003). The frequency and size of these cyanobacteria blooms is increasing globally, therefore, to avoid the occurrence of, or control the impact of their presence, an understanding of the biological, chemical and physical factors that can affect their abundance is required (Suthers & Rissik, 2009, Mantzouki et al., 2018).

In many lakes cyanobacteria are favoured by warm temperatures, where total phosphorus concentrations range from 100 to 800 mg L^{-1} or total nitrogen ranges from 2.5 - 3.5 mg L^{-1} , and the average depth of the lake is less than 2 m (Chen et al., 2003). However, cyanobacteria are able to adapt to a broad range of environmental conditions and have evolved different species to colonise very different aquatic ecosystems (Salmaso, 2000); *Anabaena* or *Oscillatoria* can survive nitrogen limited conditions by fixing nitrogen (Latysheva et al., 2012), *Microcystis* is capable of intracellular storage of assimilated phosphorus during periods of supply in excess of growth requirements (Alam et al., 2001, Cottingham et al., 2015), *Anabaena* can produce gas vesicles to provide buoyancy (Cottingham et al., 2015), and *Gloeotheca* have the ability to capture light at low intensities and a range of wavelengths by producing different phycobiliproteins (Henriksen, 2010, Carey et al., 2012).

Cyanobacteria are competitive with other phytoplankton as their growth rate is often higher than other species which allows them to outcompete at optimum temperatures. They are also able to migrate vertically through the water column, unlike some other species so they can avoid sedimentation in warmer waters (Lüring et al., 2013). As a result, there will always be at least one cyanobacterial taxa which will be able to dominate under increasingly variable conditions, however they are often more abundant in summer as they are controlled by nutrient concentrations, water temperature and water column stability (Alam et al., 2001, John et al., 2011, Carey et al., 2012).

Lakes often have a cryptomonad population which is usually relatively low but consistent throughout the year. Cryptomonads are most common in cooler waters following periods of moderate turbulence, although sudden population growth is common and they can show a strong seasonal distribution within lakes with a peak in population in summer in nutrient-poor conditions (Hoek et al., 1995, Barone & Naselli-Flores, 2003, Li et al., 2003, John et al., 2011).

1.4.2 The Seasonal Succession of Phytoplankton

Phytoplankton communities are subject to seasonal periodicity, the extent of which can alter depending on the trophic status of the lake (Amblard, 1988). The phytoplankton growth season often starts when stratification begins in the spring and maximum growth rates are favoured. Once the water column begins to stabilise during summer, grazing pressure and competition for nutrients become dominant, and phytoplankton growth begins to slow, until autumn brings the return of water column mixing and light becomes limited, ultimately reducing phytoplankton growth (Sommer, 1985).

1.5 Environmental Controls on Phytoplankton

The environmental variables within a lake can be the driving factors of phytoplankton abundance and structure (Mohebbi et al., 2015), as phytoplankton community changes depend on factors such as temperature, light intensity and lake flushing rate. Temperature changes, particularly increasing temperatures, can alter the phenology of dominant phytoplankton within a lake (Elliott et al., 2006). An increase in temperature combined with low wind speeds, can cause the onset of the spring phytoplankton bloom to occur earlier in the year and cyanobacteria are particularly adept at taking advantage of rising lake temperatures (Elliott et al., 2006, Peeters et al., 2007), increasing the opportunity for harmful algal blooms (HABs) to occur.

Precipitation and the resultant river discharge entering a lake can control phytoplankton growth through dilution. Significant phytoplankton populations can only be created when their net growth rate exceeds the dilution rate. This impact of river discharge often decreases during summer as lake conditions stabilise and other factors controlling phytoplankton growth are more prominent (Descy, 1993). The rate of river discharge contributes to the flushing rate of a lake; a measure of how often the total volume of water in the lake is replaced. Lakes have low flushing rates compared to rivers and streams, which constantly replenish their water volume. The morphometric characteristic of a basin and their watersheds, combined with the volume of precipitation over the area determines the flushing rate of a lake (Ambrosetti et al., 2003).

The flushing of a lake is essential for the dilution and removal of nutrients and phytoplankton, as well as to reduce sedimentation (Interagency Freshwater Group, 2015). Therefore, the flushing rate controls the concentration and accumulative capacity of substances such as nutrients entering a lake. Calculating this rate allows the chemical and trophic state of individual lakes to be determined and management decisions made to safeguard lake communities are more successful (Ambrosetti et al., 2003), and is particularly important as a variation in lake flushing rate throughout the year can influence lake biology; reduced flushing during summer encourages conditions favouring phytoplankton blooms (Interagency Freshwater Group, 2015).

The availability of light underwater is also an important environmental variable within a lake (Luhtala & Tolvanen, 2013). Light limitation is a major control of photosynthetic activity and

therefore phytoplankton production and turnover rates (Cloern, 1987, Luhtala & Tolvanen, 2013); within shallow turbulent lakes light can strongly influence the phytoplankton species composition and biomass. Species either try to avoid the high light intensity of the surface waters, or when light conditions are poor, require the ability to float to the surface (Chen et al., 2003).

1.5.1 Lake Stratification

In most lakes, a warming of temperature experienced during spring allows heat to be transferred to deeper lake waters, creating a classic thermal structure where the lake body stratifies. An epilimnion is formed at the surface, a metalimnion intermediate layer, and a hypolimnion at the bottom of the lake water column. A thermocline is formed in the metalimnion and is the level of maximum vertical heat gradient (Ambrosetti et al., 2003). Thermal stratification of the water column such as this affects the vertical transport of nutrients and phytoplankton.

As a result, stratification has the ability to influence the dominant processes in a lake (Teixeira et al., 2013). Phytoplankton species which are non-motile, such as diatoms benefit from a mixed water column, whereas in stratified waters small motile phytoplankton such as flagellates are favoured (Teixeira et al., 2013). This causes phytoplankton blooms to form in the warmer surface waters during stratification events (Engel & Fischer, 2017). Destratification often begins in autumn when surface cooling begins to take place; the quantity of heat absorbed by the lake is less than that lost via its surface and the lake becomes more turbulent (Ambrosetti et al., 2003), which in turn causes the phytoplankton biomass to decrease as conditions for growth are less favourable.

1.6 Phytoplankton and Nutrients

One of the most crucial factors that determine lake food-webs is the supply of nutrients (Harper, 1992), as concentrations and their availability affect the health of a freshwater lake system as well as phytoplankton biomass and metabolism levels (Macintyre & Jellison, 2001, Musungu et al., 2014b, They et al., 2017). As a result, nutrient fluxes can control a lake ecosystem; nutrient-rich waters often lead to high rates of primary productivity and nutrient poor lakes

have low productivity and short growing seasons (Macintyre & Jellison, 2001, Musungu et al., 2014b, Bergström et al., 2018).

Nutrients enter a lake through four main processes; the weathering of rock, which adds nutrients in relatively small quantities; atmospheric input through precipitation; biological nitrogen fixation; immigration through organism decomposition (Perkins & Underwood, 2000, Kumar & Sekaran, 2014). This loading can occur on a local, regional and global scale (Musungu et al., 2014b).

1.6.1 The Role of Phosphorus in a Lake

Phosphorus is essential for the growth of lake organisms and is often the principle limiting nutrient for primary productivity. Changes in concentration can influence long-term changes in phytoplankton composition and cause undesirable events under the right conditions including; accelerated plant growth, phytoplankton blooms, low dissolved oxygen levels and the death of certain fish, invertebrates and other aquatic animals (Harper, 1992, Perkins & Underwood, 2000, Tallberg, 2000, Anneville et al., 2004b, Eaton et al., 2005, Chen & Taylor, 2011, Carey et al., 2012, Skwierawski, 2012, Bhateria & Jain, 2016).

Phosphorus lacks a gaseous phase, therefore enters a lake through the weathering of rock and soil erosion, the supply of which is determined by the rate of runoff and river discharge into the system (Sterner, 2009, Kumar & Sekaran, 2014). Phosphorus that is locked up in bedrock, soils and sediments is not directly available to organisms, therefore when converted to dissolved orthophosphate, which can be directly assimilated, occurs at various stages of the phosphorus cycle (Ruttenberg, 2003).

1.6.2 The Role of Nitrogen in a Lake

Nitrogen (N) is also an essential nutrient for plants and animals within a lake system, it is commonly measured in three forms, ammonium (NH_4^+ - N), nitrate (NO_3^- - N) and nitrite (NO_2^- - N), the sum of which is referred to as total nitrogen (TN) (United States Environmental Protection Agency, 2014). Nitrate can be converted to ammonium by bacteria and

cyanobacteria through nitrogen fixation. Nitrification can then take place by bacteria converting ammonium to nitrate and nitrite. Denitrification also occurs where nitrate is reduced to dinitrogen gas (Arnell, 2014). The growth of phytoplankton within a lake is often limited by nitrogen or phosphorus, with seasonal transitions taking place between them (Rhee, 1978, Harper, 1992, Eaton et al., 2005, Carey et al., 2012).

1.6.3 The Role of Silica in a Lake

Silica is a micronutrient for most plants and animals but is often considered a non-essential trace element, as although it is present in low concentrations in freshwaters it is often well above limiting values during the seasons of optimal growth. Silica is however vital for freshwater diatom communities as it makes up the frustule of the diatom and can become a limiting nutrient under certain conditions. Limiting concentrations of silica can cause a diatom population to deteriorate faster than if it is deprived of phosphorus or nitrogen (Conley et al., 1989, Harper, 1992, Tallberg, 2000, Eaton et al., 2005).

1.6.4 Lake Stoichiometry

N:P stoichiometry affects important ecosystem characteristics including planktonic nutrient limitation, energy and nutrient turnover, and lake productivity (Bergström et al., 2018). This is often described using the Redfield Ratio; the average elemental composition of phytoplankton and it dictates the biogeochemical impacts of phytoplankton growth and decay (Tyrrell, 2019). The atomic ratios of C:N:P in aquatic biomass are approximately 106:16:1 and known as the Redfield Ratio (Perdue & Benner, 2009). When the Redfield ratio is combined with the chemical equation for photosynthesis ($\text{CO}_2 + \text{H}_2\text{O} = \text{CH}_2\text{O} (\text{s}) + \text{O}_2$) a more sophisticated chemical equation for photosynthesis in aquatic environments is obtained: $106 \text{ CO}_2 + 106 \text{ H}_2\text{O} + 16 \text{ NH}_3 + \text{H}_3\text{PO}_4 = (\text{CH}_2\text{O})_{106} (\text{NH}_3)_{16} (\text{H}_3\text{PO}_4) (\text{s}) + 106 \text{ O}_2$ which is transformed to the chemically equivalent formula: $\text{C}_{106} \text{ H}_{263} \text{ O}_{110} \text{ N}_{16} \text{ P}(\text{s})$ (Perdue & Benner, 2009). Lakes that are rich in P are often rich in N, therefore there is a positive correlation between the TN and TP concentrations of lakes (Downing & Mccauley, 1992).

Phytoplankton C:N:P ratios are influenced by species composition (e.g., N-fixing cyanobacteria, diatoms etc), temperature, the quality and quantity of organic matter inputs, light:nutrient ratio, TP, chlorophyll, particulate organic carbon (POC):chlorophyll ratio and water residence time (They et al., 2017).

1.6.5 Anthropogenic Nutrient Inputs & Eutrophication

The fluctuation in nutrient concentrations caused by seasonal and long-term meteorological changes are escalated by the addition of nutrients from human activities (Anneville et al., 2004a). Nitrate, nitrite, and phosphorus can be loaded externally into a lake from anthropogenic sources. Precipitation can wash excess nutrients from residential areas and agricultural land and sewerage treatment works often discharge effluent into water bodies (Harper, 1992, Perkins & Underwood, 2001a, Anneville et al., 2004a, Eaton & Franson, 2005, Chen & Taylor, 2011, Skwierawski, 2012). Therefore anthropogenic loading can increase the ratio of nitrogen to phosphorus, as well as silicate, with the ability to intensify phosphorus limitation (Hessen et al., 2009).

Excessive nutrient concentrations, particularly anthropogenically produced phosphorus and nitrogen, can have a major impact on lake ecosystem structure by affecting water quality and causing eutrophication (Nyenje et al., 2010b, Xu & Zhang, 2012, Abell et al., 2013b, Musungu et al., 2014a). This can have a knock-on effect on the biological aspects of a lake such as plankton growth which may directly and indirectly affect lake dissolved oxygen (DO) concentrations (Harper, 1992).

1.7 Top-Down Controls of Phytoplankton

Predators within a lake have important top-down effects that cascade down and influence lower trophic levels (Du et al., 2015). Zooplankton link phytoplankton with organisms higher up the trophic levels in a lake, therefore grazing of zooplankton influences the phytoplankton population (Du et al., 2015). Clear water periods experienced in lakes are often due to zooplankton grazing of phytoplankton (Lampert et al., 1986), as zooplankton have the ability

to control phytoplankton biomass, although the extent of this impact is often dependent on nutrient availability.

1.7.1 The Impact of Zebra Mussels on Phytoplankton

Lakes are often very vulnerable to invasive non-native species as many of these freshwater habitats are damaged by existing pressures, therefore gaps have been created that can be exploited by fast-growing non-native species, such as zebra mussels (Hatton-Ellis, 2014). Once invasive non-native species establish themselves within a habitat it is impossible to remove them, therefore any work involving lakes in the UK must take into account the risk of non-native species (Hatton-Ellis, 2014).

Zebra mussel colonisation within a lake can dramatically alter the phytoplankton community composition (Fishman et al., 2010) as they have a rapid reproduction rate which increases in warmer waters, which also makes them a threat to lake ecosystems in the future (Adrian et al., 2009a). Zebra mussels are also effective filter feeders and are able to filter between 1 and 2 litres of water per day, per mussel, giving them the potential to remove essential phytoplankton biomass within a lake (Maguire & Sykes, 2004).

1.8 Lake Water Quality

Lakes are a valuable water resource; it is essential to find ecologically sound ways to conserve and protect them. Their continuous monitoring and the assessment of a lake's water quality are key to enable their sustainable protection and management (El-Serehy et al., 2018). One of the most detrimental impact in recent years to lake water quality is the increase of anthropogenic external nutrient loading, which has caused many lakes to suffer from increasing eutrophication leading to deterioration of their water quality (Tadonlélé et al., 2009).

Water quality (WQ) describes the condition, as well as, the suitability of water for use, making it subjective and it often reflects the assessment of the user for the different uses of the water (El-Serehy et al., 2018). The assessment of lake WQ is most relevant in lakes when ecological and socio-economic balances need to be made in a water body because WQ indices characterise

and describe factors such as nutrient load, water consumption and climate change in relations to the aquatic ecosystem. This makes them key for water resource management (El-Serehy et al., 2018).

There have been different approaches to the assessment of WQ worldwide. The water quality index was developed in the United States by (Horton, 1965), and has been widely applied in European, African and Asian countries (El-Serehy et al., 2018). Carlson developed a numerical classification for the trophic status of lakes (TSI) which was modified into a trophic level index (TLI) (Liou & Lo, 2005).

1.8.1 Lake Dissolved Oxygen Concentrations

Healthy lakes have a good nutrient and DO balance (Hasan et al, 2014). The main sources of oxygen in a lake is from the air, and often most importantly, as a by-product of photosynthesis, making the role of phytoplankton within a lake an important consideration for lake management (Hasan et al., 2014). At higher temperatures, organism metabolism increases, causing a rise in lake DO utilisation, causing DO concentrations to decrease (Fang & Stefan, 2009, Jeppesen et al., 2009). Oxygen is also required within the water column of a lake for the decomposition of organic matter. When organic matter levels increase, the oxygen consumption by microbial decomposition processes needs to be balanced with sufficient oxygen production to maintain DO concentrations and prevent hypoxia, particularly at depth (Astuti et al., 2014). As a result, dissolved oxygen in a lake often varies due to thermal stratification (Wilhelm & Adrian, 2008). A lack of water movement at depth due to stratification can cause DO concentrations to fluctuate due to local distortion by faunal activity that causes oxygen fluctuations in the water above the sediment (MacPhearson, 2003).

Oxygen depletion becomes more pronounced with longer stratification times and a more stable system; simulations have shown that DO concentrations in some lake surface waters may change by $< 2 \text{ mg liter}^{-1}$ in the future but hypolimnion concentrations could decrease by as much as 8 mg L^{-1} , increasing the frequency of anoxia in bottom waters (De Stasio Jr et al., 1996). The strong influence that temperature and lake thermal structure has on DO makes lake DO concentration a valuable indicator of shifts in climate, particularly because DO

concentrations have widespread consequences for internal nutrient loading (Adrian et al., 2009a).

1.8.2 Lake Trophic Status

Lakes can be classified depending on their nutrient content. Oligotrophic lakes have low primary productivity due to low nutrient content. They have low algal production leading to clear waters with a high drinking water quality. A lake with an intermediate level of productivity, with clear waters and beds of submerged aquatic plants with medium levels of nutrients are mesotrophic. A eutrophic lake contains excessive nutrients, especially nitrogen and phosphorus. These lakes can support an abundance of aquatic plants. The water body is often dominated by aquatic plants, making the water clear, or algae which make the water darker. Eutrophication can naturally occur here as a result of respiration by algae blooms and bottom living bacteria which kills fish (Bhateria & Jain, 2016).

Hypereutrophic lakes are highly nutrient-rich lakes which suffer from severe algal blooms and low transparency. The visibility depth is less than 3 feet, they have more than 40 µg/l total chlorophyll and greater than 100 µg/l phosphorus. Excessive algal blooms in these lakes can significantly reduce oxygen levels and prevent lake from functioning at lower depths, creating dead zone below the surface (Bhateria & Jain, 2016).

1.8.3 Water Quality Directives

The restoration of freshwater ecosystems due to lowered water quality has been implemented worldwide through legislations including the European Waste Water Treatment Directive, the European Water Framework Directive, and the U.S. Federal Water Pollution Control Act (Spears et al., 2013).

The European Water Framework Directive (2000/60/EC) is one of the most significant pieces of European water legislation as of late, a key component of which requires the ecological status of waterbodies to be assessed on the condition of their biological quality elements, which for lakes includes the monitoring of phytoplankton abundance and composition (Carvalho et

al., 2006, Brierley et al., 2007, Carvalho et al., 2007), as well as the identification of environmental change (Burthe et al., 2016).

The European Water Framework Directive requires European waterbodies to be assigned an ecological class based on biological indicators. Its aim was to protect and improve the quality of all surface waters by 2015 by assessing variables including phytoplankton, macrophytes, invertebrates and fish (Søndergaard et al., 2005).

1.9 Changes in Climate & Lake Systems

Climate change is one of the more challenging issues of the last decade (Massei et al., 2007). Although large changes in the Earth's climate occur naturally over long periods of time as a result of processes including solar activity, volcanic eruptions and the cycle of the Earth's orbit around the sun, at present the Earth's climate is rapidly changing due to anthropogenic activity (Welsh Assembly Government, 2009a). Since the industrial revolution anthropogenic emissions of greenhouse gases have continuously increased on a global scale as a result of the burning of fossil fuels (Welsh Assembly Government, 2009a, Defra, 2012a). These gases which include carbon dioxide (CO₂), create a blanket in the atmosphere which traps heat from the sun and results in the unequivocal warming of the Earth's surface and lower atmosphere (Welsh Assembly Government, 2009a). There has been an increase in global average air temperature, committing the Earth to significant climate change; one of the most severe threats to ecosystems around the world (Intergovernmental Panel on Climate Change, 2007, Adrian et al., 2009b, Defra, 2012a).

It is becoming increasingly apparent that large-scale oceanic and climatic fluctuations affect interannual and interdecadal variability in ecological processes (Gerten & Adrian, 2001). Aquatic environments are often more responsive to broad-scale climatic phenomena because the thermal characteristics of most water bodies are well suited to store such climate information (Gerten & Adrian, 2001).

1.9.1 The North Atlantic Oscillation

The North Atlantic Oscillation (NAO) is a macroscale circulation pattern that varies significantly with the overlying atmosphere, making it susceptible to changes in climate. It determines the inter-annual and inter-decadal, as well as the winter climate variability of much of Europe and large areas of northern Asia (Gerten & Adrian, 2000a, Visbeck et al., 2001, Blenckner et al., 2007b). The NAO index is a measure of the difference in atmospheric pressure anomalies between the sub-tropic Azores High and the subpolar Icelandic Low, where highly positive NAO indices indicate steep surface pressure gradients and strong zonal circulation over the North Atlantic (Gerten & Adrian, 2000a, Blenckner et al., 2007b). These conditions lead to mild and wet winters in the western and northern parts of Europe, with the reverse situation causing cold winters as a result of low surface pressure gradients and weaker zonal circulation over the North Atlantic (Blenckner et al., 2007b). An increase in global warming has corresponded with an upward trend in the NAO (Visbeck et al., 2001), making the NAO an important climatic driver and the winter index a valuable tool in the investigation of climate-induced changes in both aquatic and terrestrial ecosystems, (Anneville et al., 2005a, Blenckner et al., 2007b).

The NAO has an influence on lake water temperatures, one of the major factors controlling ecological processes in lakes (Gerten & Adrian, 2001). This differs substantially among lake types of different thermal structures and mixing regimes, even under identical climatic forcing (Gerten & Adrian, 2001). A frequently circulating polymictic lake was found to be least influenced by the winter effects of the NAO, with an effect lasting only into early spring. In contrast, in a deep dimictic lake with stable summer stratification, the NAO signal persisted in the hypolimnion until the following winter. A shallow dimictic lake revealed an intermediate response, as weather conditions both in April and midsummer probably modified the strength and persistency of the NAO signal in the hypolimnion of that lake. Based on these results, it is to be expected that NAO effects on ecological processes vary significantly among lakes (Gerten & Adrian, 2001).

1.10 Managing Lakes for the Future

One of the biggest challenges facing the management of freshwater ecosystems across the world is the improvement of water quality in lakes (Spears et al., 2013). Understanding the multiple factors which drive the ecological changes in phytoplankton communities under current climatic changes is a fundamental challenge of aquatic ecology but is required for the implementation of effective lake management (Alpine & Cloern, 1992, Kumari et al., 2018). Pressures need to be reduced to ensure that ecosystems do not undergo any change (prevention of degradation) or change to a more desirable ecological state (restoration) (Spears et al., 2013).

European lakes are experiencing pressure from nutrient pollution as well as climate change. The result of these pressures, where over time lakes have been subject to a progressive increase in nutrient loading, causing phytoplankton productivity to rise (Battarbee et al., 2012). This has resulted in several countries implementing a reduction in nutrient loading to improve lake water quality (Tadonlélé et al., 2009). Figure 1.2 conceptually indicates how lakes can respond to this reduction in nutrients; some lakes experience an immediate decrease in phytoplankton production, however other lakes have a limited response despite a significant decrease in external loading. This can be a result of internal loading delaying the response of a lake. However, nutrient enrichment remains the dominant influence on the trophic status of a lake, although the influence of a changing climate has been seen to offset some lakes' recovery to a reduction in nutrient loading (Battarbee et al., 2012).

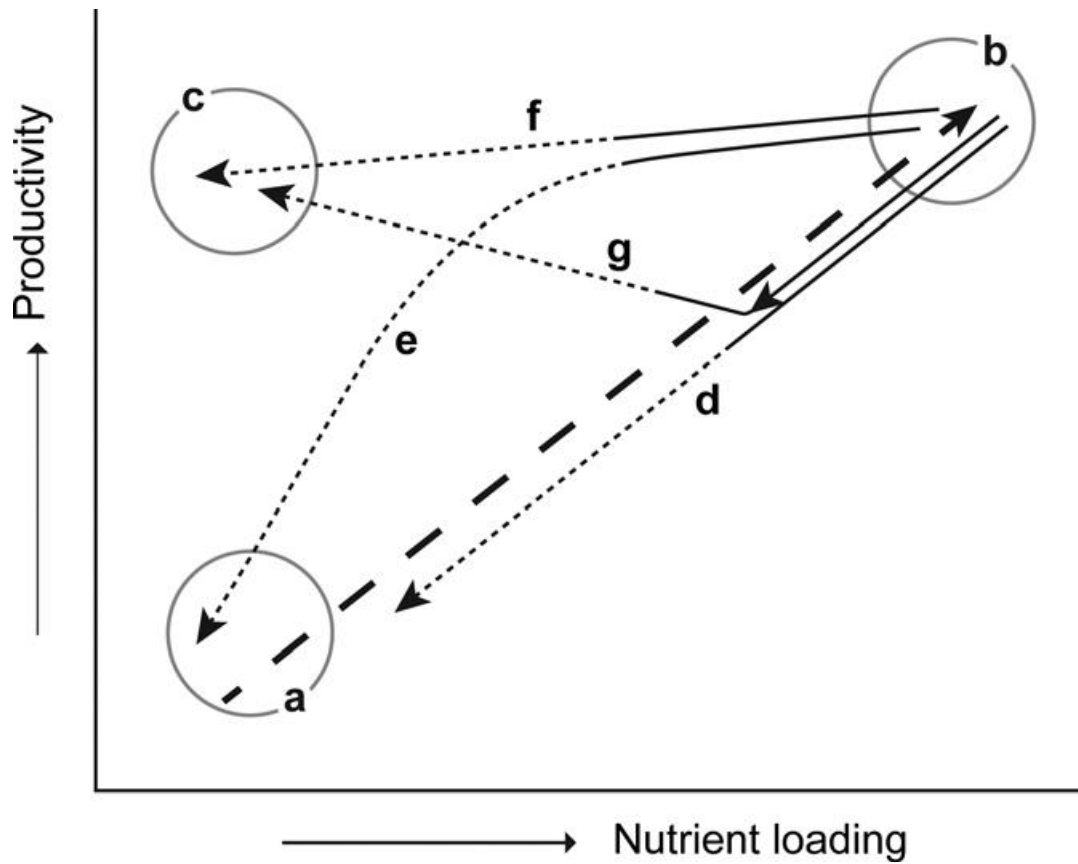


Figure 1.2: Conceptual diagram using a combination of palaeolimnological data (dashed line), contemporary long-term monitoring data (solid lines) and future conjecture (dotted lines) to show idealised changes in the past, present and future relationship between nutrient loading and productivity for European lakes recovering from eutrophication. Point (a) indicates the reference state and the target endpoint following restoration; point (b) indicates the point of intervention to reduce nutrient loading; and (c) indicates a more probable potential endpoint in cases where recovery to the past reference (a) is prevented by the enriching effects of climate change (Battarbee et al., 2012).

1.11 Cardiff Bay as a Study Site

Cardiff Bay, South Wales lies at the mouth of the Rivers Taff and Ely and outputs into the Severn Estuary, which has the second highest tidal range in the world (Cardiff Harbour Authority, unknown). During the early 1900s Cardiff Bay was the world's largest coal port (Cowell, 2003). However by 1985 the port was seen as an area in need of urban regeneration, leading to the proposed construction of a barrage across the mouth of the rivers Taff and Ely with the intention to re-generate Cardiff's docklands and re-unite the city centre with a newly formed permanent 13 km waterfront (Cowell, 2003, Cardiff Harbour Authority, unknown). Barrage construction was authorised in the early 1990s and commenced in 1994 (Andrews & Gulson, 2002, Cardiff Harbour Authority, unknown). Cardiff Bay was temporarily impounded in November 1999 when the bay was filled with seawater and occasionally emptied (Andrews & Gulson, 2002). Permanent impoundment in April 2001 signified the completion of the barrage, creating a 200 ha freshwater constant-level lake (Andrews & Gulson, 2002, Burt, 2002, Crompton, 2002, Cardiff Harbour Authority, unknown).

Both the Taff and Ely Rivers, alongside the Severn Estuary have several statutory designations. The Rivers are Sites of Nature Conservation Interest (SNCI) and Sites of Importance for Nature Conservation (SINC); designations granted before the impoundment of Cardiff Bay, but which have continued due to their importance for migratory fish, otters, wildfowl and bankside vegetation, making the Rivers Taff and Ely a major corridor for wildlife. The Taff/Ely Estuary and the Severn Estuary are Sites of Special Scientific Interest (SSSI). The Severn Estuary is also a Wetland of International Importance (Ramsar Site), Special Protection Area (SPA) European Marine Site, and a Possible Special Area of Conservation (pSAC) (Cardiff Harbour Authority, unknown).

It was recognised during the early stages of barrage design that there were high environmental risks associated with the creation of a freshwater lake at the mouth of the River Taff and River Ely. The rivers were steep, drained industrial land, and were prone to flash flooding. There were combined sewerage problems and a migratory salmonid fishery at an early stage of recovery. Therefore, post impoundment, Cardiff Bay has required careful and balanced management to cope with the occasionally conflicting requirements of water quality, fisheries,

flood defence, recreation, and business without compromising the environmental standards of the area (Andrews & Gulson, 2002).

In April 2000, under the control of Cardiff County Council (CCC) the Cardiff Harbour Authority (CHA) was formed to manage the operation of the barrage and the bay. The Environment Agency Wales (now known as Natural Resources Wales) has a regulatory role in the management of Cardiff Bay, including flood defence, fisheries and water quality (Andrews & Gulson, 2002).

The barrage was implemented to eliminate the effects of the tide, releasing the potential of Cardiff's greatest asset – its waterfront (Cowell, 2003). This required sixteen major sewerage outfalls to be diverted from the Bay to new discharge points beyond the barrage to ensure that reasonable dissolved oxygen (DO) levels could be achieved to maintain water quality and minimise siltation (Burt & Littlewood, 2002, Crompton, 2002).

The barrage also operates as a flood defence structure, it not only has sufficient storage to retain the freshwater volume accumulated during tide lock but the Bay is able to safely rise by 3 m to accommodate floodwaters and therefore deal with a 1 in 100 year fluvial flood coincident with a mean annual high tide. The five sluice gates within the barrage are able to discharge extreme river flows when necessary (Andrews & Gulson, 2002, Burt, 2002, Cardiff Harbour Authority, unknown).

In 2002 Cardiff Bay was designated as a sensitive water. As a result, phosphate removal took place at five main sewage treatment works which discharge into the Taff and Ely catchments. This was part of Dwr Cymru Welsh Water's (DCWW) management plan for 2000-2005 to reduce the scale of algal blooms in Cardiff Bay as a requirement of the Urban Waste Water Treatment Directive (Andrews & Gulson, 2002, Cardiff Harbour Authority, unknown).

The water quality of Cardiff Bay is constantly monitored. Six continuous monitoring stations can be found within the bay, River Taff and River Ely which measure dissolved oxygen (DO), water temperature, pH, conductivity and turbidity at 1 m depth in all locations, and at 1 m above the bed of the Bay at three locations. Water samples are also collected from the bay, rivers and at groundwater dewatering discharge points and analysed for pH, temperature, biochemical oxygen demand, nutrients, total suspended solids, algae and bacteria. Depth

profiling; taking measurements at a number of depths throughout the water column, is carried out within the Bay at a number of sites for DO, turbidity, conductivity, salinity, pH and temperature (Cardiff Harbour Authority, unknown).

Due to the urban catchment of the Rivers Taff and Ely it was predicted that diatom and chlorophyte algal blooms would be a regular occurrence within Cardiff Bay. It was also suggested that potentially toxic blooms of cyanobacteria may become a problem (Andrews & Gulson, 2002, Crompton, 2002). Lake water quality is likely to be sensitive to the rate of respiration by bacteria on the lake bed; if these respiration rates were less than $1 \text{ g O}_2 \text{ m}^2 \text{ day}^{-1}$ the DO level would satisfy the 5 mg L^{-1} standard at 5 percentile river flows, whereas a rise to $2 \text{ g O}_2 \text{ m}^2 \text{ day}^{-1}$ would cause lower water layers to fall below the standard DO standard. The effect of an algal crash (sudden mortality) would be of relatively short duration but would increase benthic demand for oxygen, significantly reducing oxygen levels (Burt, 2002). Therefore Cardiff Bay has an alert level framework (ALF) to monitor an increase in algal biomass, communicate its presence and the potential impacts to water users and manage its removal (Cardiff Harbour Authority, unknown).

Filamentous algae are collected by macro-algae and disposed to landfill, while surface-accumulating algae are collected by booms and weir units. Any algal scum is assumed to be toxic and is therefore removed from the Bay through disposal to the Severn Estuary via a long sea outfall at Tremorfa sewage treatment works situated 2 km upstream of the barrage on the Severn Estuary (Crompton, 2002, Cardiff Harbour Authority, unknown).

To protect and enhance the biodiversity of Cardiff Bay, CHA is committed to preventing the introduction of (and removing where applicable) invasive species of flora and fauna. Cardiff Bay was the first site in Wales to record the presence of the non-native Zebra Mussel (*Dreissena polymorpha*) which can now be found in nearly 20 locations in Wales (Gb Non-Native Species Secretariat, 2016).

Zebra mussels have the ability to become a major problem in UK freshwaters as they reproduce rapidly and changes in climate are making conditions more favourable as waters warm (Adrian et al., 2009a, Welsh Assembly Government, 2009b, Cardiff Harbour Authority, unknown). The establishment of Zebra mussels within Cardiff Bay are of concern as each individual mussel can filter between 1 and 2 litres of water per day, removing aquatic organisms such as

phytoplankton, which are necessary for the fish and bird populations dependant on the Bay (Cardiff Harbour Authority, unknown). It is therefore an offence under the Wildlife and Countryside Act, 1981 to release or allow the escape of Zebra mussels into the wild as it is not an ordinary resident or regular visitor to Great Britain in a wild state (Cardiff Harbour Authority, unknown).

1.11.1 Study Aims and Objectives

The overarching aim of this project is ‘To use phytoplankton as the focal point for the management of Cardiff Bay, by determining the factors which control them’. This will provide an insight into the drivers of the phytoplankton community within this unique artificial lake and aid Cardiff Harbour Authority (CHA) with their management of Cardiff Bay.

Ten years of historical data collected by CHA from the lake have been analysed simultaneously with in-depth present-day data collection which was carried out over a 2-year period. These data sets will enable a picture to be drawn of the mechanisms which take place within the lake using the following objectives:

1. Environmental variables will be assessed using air and water temperature data, precipitation levels and the level of light entering the system.
2. Physical lake properties will be analysed using dissolved oxygen concentrations, river discharge rates, lake flushing rate and water column stability.
3. The phytoplankton within the lake will be assessed using chlorophyll-*a* as a proxy for biomass, variable chlorophyll fluorescence as a proxy for productivity, and species identification.
4. The bottom-up control on phytoplankton biomass, productivity and community structure will be investigated using phosphate, nitrate, nitrite, ammonium and silica concentrations, alongside the potential of nutrient release from the lake sediment.
5. The abundance and diversity of zooplankton species in the Bay will be determined, alongside their relationship with the phytoplankton community.

6. The potential for zebra mussel filtration control of phytoplankton biomass will be assessed.

Once a whole-lake assessment has been carried out and the phytoplankton community understood, appropriate management requirements can be discussed.

1.11.2 Thesis Outline

The thesis is comprised of 7 chapters; following this initial introduction chapter the methods used within the study are described in Chapter 2. The content of the following chapters is described below.

1.11.2.1 The Environmental Variables & Physical Properties of Cardiff Bay

Historical data sets for Cardiff Bay were analysed between January 2003 and April 2014 to determine how temperature, precipitation, river discharge and dissolved oxygen concentrations varied with time. Chapter 3 presents the results of this data, alongside lake flushing rate and calculations. This chapter also uses data collected fortnightly between April 2012 and April 2014 to examine the relationship between precipitation, river discharge and the flushing rate of Cardiff Bay, along with changes in photic depth within the lake.

The following hypotheses were tested in Chapter 3:

Seasonal air temperature and precipitation changes will induce seasonal lake water temperature change.

The photic depth of the lake will fluctuate seasonally due to changes in water column stability. Spatial variation in photic depth will occur due to differences in sampling site characteristics (e.g. susceptibility to mixing, sediment re-suspension).

Lake dissolved oxygen concentrations will vary spatially and temporally due to the influence of water temperature and thermal stratification.

High precipitation rates will cause an increase in river discharge and therefore cause lake flushing rates to quicken. Temporal changes in flushing rate will occur as a result of seasonal changes in precipitation.

1.11.2.2 The Phytoplankton Community of Cardiff Bay

The phytoplankton community within Cardiff Bay was assessed in terms of its biomass, productivity and assemblage in chapter 4. Biomass data was collected throughout the lake by CHA between January 2003 and April 2014 and were combined with more extensive biomass, productivity and assemblage data which were collected between April 2012 and April 2014. The influence that temperature, precipitation, river discharge, and lake flushing rate had on the phytoplankton community was also analysed in chapter 4, alongside the relationship between the phytoplankton community and photic depth, and dissolved oxygen concentrations.

The following hypotheses were tested in Chapter 4:

The phytoplankton biomass, productivity and assemblage within the lake does not vary spatially or with depth within the Bay.

There will be seasonal variation in phytoplankton biomass, productivity and assemblage within the lake.

Phytoplankton biomass is controlled in part by flushing rate, such that increases in flushing rate reduces biomass due to out-flushing, and low flushing rate reduces biomass out-flushing.

1.11.2.3 The Bottom-Up Controls on Phytoplankton within Cardiff Bay

The concentration of nitrogen, phosphorus and silica found within Cardiff Bay and how it varies spatially and over time is examined in chapter 5. Data was taken between January 2003 and April 2014. The relationship of these nutrients with temperature, precipitation, river discharge and lake flushing rate were examined. The influence that phosphorus removal from wastewaters entering the lake had on nutrient limitation and therefore phytoplankton can also

be found in chapter 5. Sediment cores were taken during summer and winter and the results used to investigate the potential for nutrient release from the sediment. Ultimately, the impact that the lake concentration of nitrogen, phosphorus and silica had on the phytoplankton community was analysed in chapter 5.

The following hypotheses were tested in Chapter 5:

The concentration of phosphorus, nitrogen and silica within the lake do not vary spatially or with depth.

There is a seasonal variation in phosphorus, nitrogen and silica concentrations within the lake.

Compared to nitrogen and silica, phosphorus is the most limiting nutrient in Cardiff Bay, causing phosphorus concentrations to determine phytoplankton biomass.

Phosphorus exchange between the lake bed and the water column is relatively low due to the water column being well-mixed.

1.11.2.4 The Top-Down Controls on Phytoplankton in Cardiff Bay

Chapter 6 showcases the zooplankton population studied in Cardiff Bay between April 2012 and April 2014 and the relationship that zooplankton has with the phytoplankton community in the lake. The impact of Zebra mussel growth on the phytoplankton community within in the lake is also discussed in this chapter.

The following hypotheses were tested in Chapter 6:

The zooplankton of Cardiff Bay varied seasonally but did not alter spatially due to the lake being well-mixed.

Zooplankton control the phytoplankton biomass within the lake through grazing.

The zebra mussel population of Cardiff Bay has the potential to contribute to top down control through phytoplankton filtration.

1.11.2.5 General Discussion

To conclude the thesis, chapter 7 combines the results from chapters 3 to 6 in a final discussion. The environmental drivers of the phytoplankton community within the lake will be clarified and how the community is controlled by bottom-up and top-down interactions. Suggestions for management of the lake will be discussed accordingly with the completion of the thesis leading to a management plan for the lake of Cardiff Bay with the phytoplankton community at the forefront.

MATERIALS AND METHODS

2.1 Study Site

Located on the south coast of Wales, UK the formerly tidal Cardiff Bay is situated within a European maritime climate. The Bay became a 200 ha artificial freshwater lake with a volume of 6,734,000 m³ in 2001, following the development of the Cardiff Bay Barrage across the Taff-Ely estuary. The water level of Cardiff Bay is maintained at 4.5 m above Ordnance Datum by five sluice gates which control the outflow of water into the Severn Estuary (Figure 2.1). This relatively shallow lake has a mean depth of c4.5 m and a maximum depth of c13 m (Figure 2.2) (Andrews & Gulson, 2002, Crompton, 2002, Cowell, 2003, Vaughan et al., 2008, Alix, 2010, Jüttner et al., 2010, Cardiff Harbour Authority, unknown).



Figure 2.1: Cardiff Bay, South Wales UK (adapted from (Arcgis, 2018)).

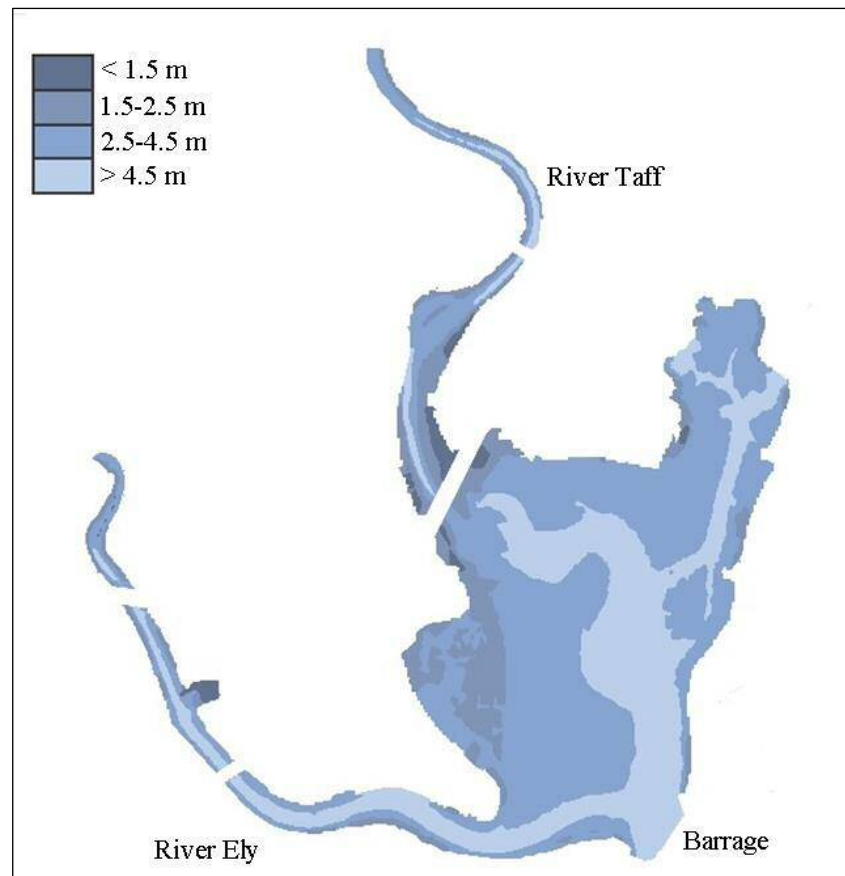


Figure 2.2: Depth level within Cardiff Bay, surveyed April 2010. Depths shown at Bay level c4.5 m above Ordnance Datum, adapted from Cardiff Harbour Authority (2011).

The catchment of Cardiff Bay is densely populated and industrial, exposing it to water pollution and high nutrient levels. As a result the lake was classed as a sensitive area (Defra, 2002) under the Urban Wastewater Treatment Directive (Council of European Communities, 1991). Post designation, Dŵr Cymru/Welsh Water (DCWW) was required to reduce the nutrient concentration of the effluent from five main sewerage treatment works discharging into the Rivers Taff and Ely to reduce algal abundance, especially that of species known to be associated with blooms and toxin production, and to increase biodiversity (Council of European Communities, 1991, Estrada et al., 2011). Phosphate removal began on 31st March 2005 at

Duffryn Isaf on the River Ely and at subsequent sites; Rhiwsaeson and Coslech on the River Ely; Cynon and Cilfynydd on the River Taff, on 31st March 2006 (Hall D 2012, pers. comm.).

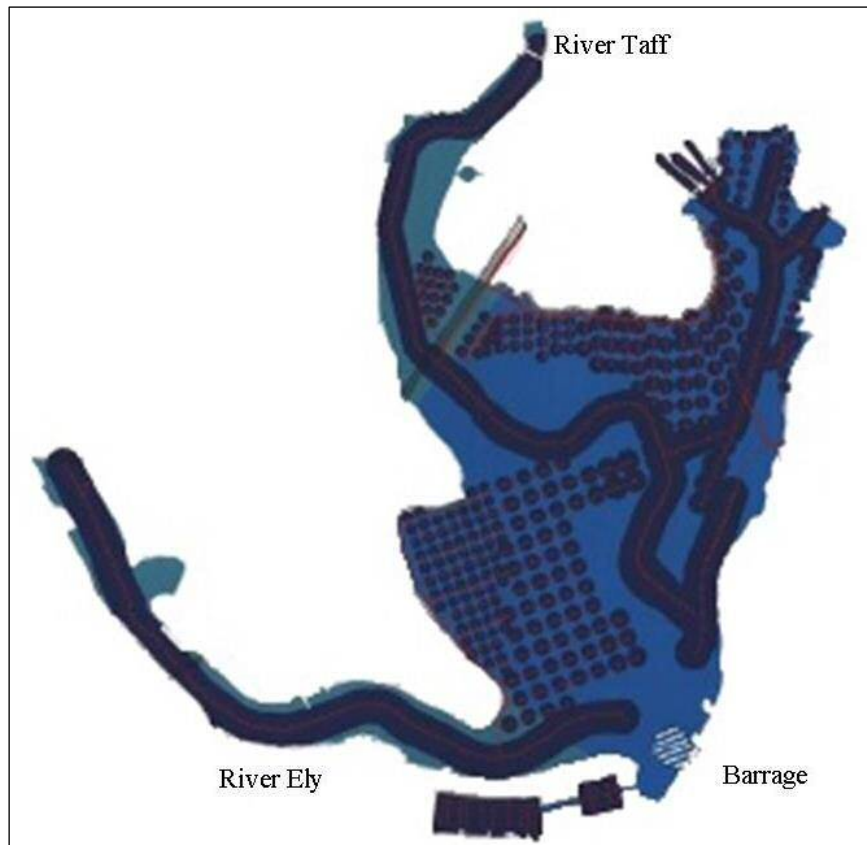


Figure 2.3: Aeration system within Cardiff Bay. Dark blue areas indicate pipes and diffusers, adapted from Cardiff Harbour Authority (2011).

Cardiff Bay is managed by the government organisation Cardiff Harbour Authority (CHA) which is responsible for the upkeep of the Bay. One of CHA's responsibilities is to ensure that the water of Cardiff Bay has a dissolved oxygen (DO) level of at least 5 mg L^{-1} in all places at all times; a specification required under the Cardiff Bay Barrage Act (1993). Upon assessment after barrage implementation, it was realised that this level of DO was unlikely to occur naturally within the lake, particularly during the summer months. As a result, 5 compressor stations, a network of around 38 km of flexible pipe and approximately 750 diffusers were installed on the bed of the Bay to carry compressed air to all areas within the lake (Figure 2.3).

This forced aeration system is designed to mix the oxygen concentrated surface water with the lower DO carrying water at the bed of the Bay; potentially allowing surface reaeration and a reduction in stratification, whilst achieving a DO content of at least 5 mg L⁻¹ throughout the water column (Cardiff Harbour Authority, unknown). However, if DO concentrations fall below 5 mg L⁻¹ an Alert Level Framework (ALF) will deploy an oxygenation barge. This barge comprises a pump, generator and cryogenic tank containing liquid oxygen and can inject oxygen into areas with low dissolved oxygen concentrations, however it has not been required to date (Jüttner et al., 2010, Cardiff Harbour Authority, unknown).

2.2 Data Collection

This study uses data that was collected by CHA for the period encompassing January 2003 and April 2014. Data was recorded at sites 4, 5, 6, 9, 10 and 17 (Figure 2.4) by CHA using a YSI multiprobe. In addition, fortnightly sampling of the lake was conducted from April 2012 to April 2014 inclusive at sites 4, 5, 9, 10, A, B, C and D (Figure 2.4) for this study. Sites 4, 5, 9 and 10 were chosen to correspond with the data collected by CHA. The remaining four sites (sites A, B, C and D), were chosen to provide comprehensive coverage of the rest of the lake, and were selected to cross known water temperature, dissolved oxygen, chlorophyll-*a* and turbidity gradients within the lake as identified by (Alix, 2010), (Figure 2.5).

2.2.1 Environmental Variables and Physical Properties Data

For the period encompassing January 2003 to April 2014 air temperature and precipitation data were collected by CHA continuously at 15 min intervals from the Davis Weather Station (319252E 173493N at 10 m Ordnance Datum Newlyn and 3 m above ground) which is situated on the banks of Cardiff Bay (Figure 2.4). Air temperature and precipitation data was collected using a Davis Vantage Pro anemometer model no. 6410. These were analysed by CHA.

River discharge was recorded continuously by CHA every 15 minutes at Pontypridd on the River Taff (c15 miles from Cardiff Bay) and at St Fagans on the River Ely (c6 miles from Cardiff Bay).

Water temperature, dissolved oxygen (DO), conductivity, turbidity and pH of Cardiff Bay were recorded continuously at 15 minute intervals by CHA using fixed YSI 6600 sondes with temperature 6560, optical DO 6150 ROX, conductivity 6560, turbidity 6136 and pH 6589 probes. Data were collected from six YSI multiprobe sites within Cardiff Bay (Figure 2.4) between January 2003 and April 2014. Water temperature and DO were recorded at c1 m depth at all six sites and additionally at c7 m at site 5 and c6 m at site 10.

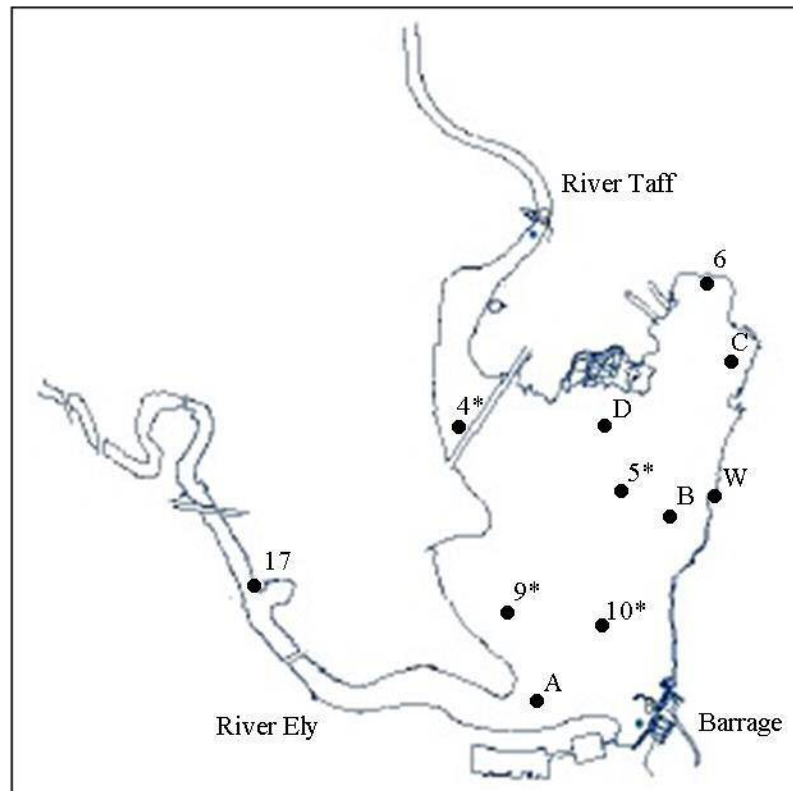


Figure 2.4: Monitoring sites within Cardiff Bay. Sites 4, 5, 6, 9, 10 & 17 are CHA YSI multiprobe buoys. Sites with an asterisk (4, 5, 9 & 10) and sites A, B, C & D were used for additional sampling. The Davis Weather Station is labelled as W.

From April 2012 until April 2014, water temperature and DO were taken in Cardiff Bay at c1 m intervals throughout the water column at all eight sites using a YSI 6600 EDS V2 Water Quality Monitoring Sonde and YSI 650MDS Data Display/Logger. Additionally, a YSI 6560 Temperature/Conductivity probe was used to collect water temperature data.

At each site a Secchi disk was used to determine the transparency of the water column by lowering the disk into the water until it was no longer visible. This data was then multiplied by 2.7 to determine the photic depth (m); the part of the water column sufficiently illuminated to allow photosynthesis to take place.

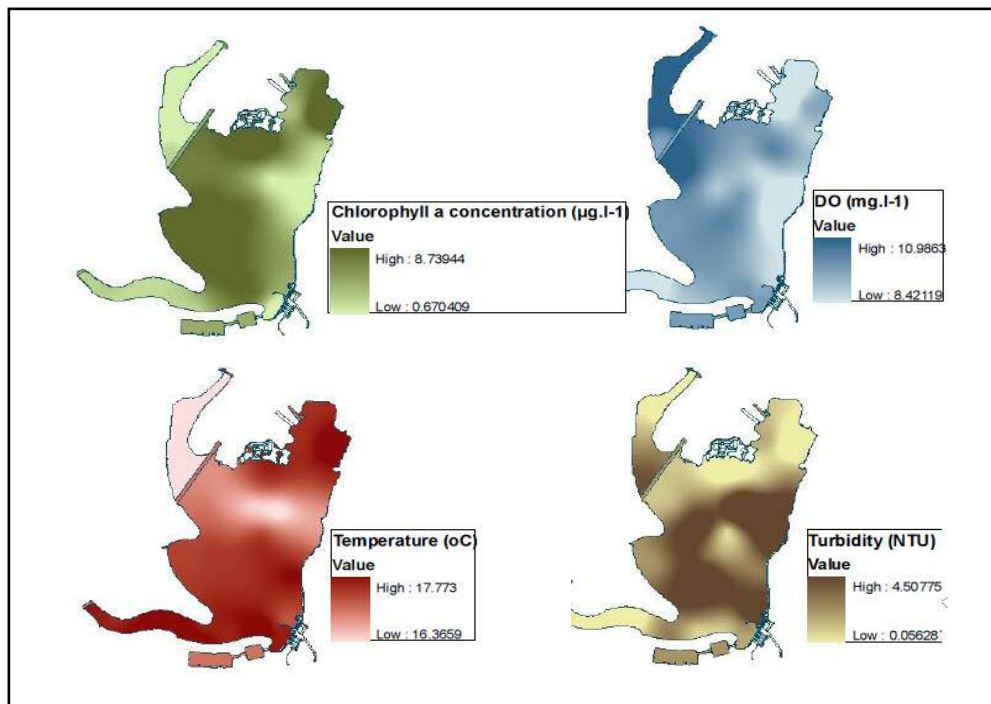


Figure 2.5: Chlorophyll-*a*, dissolved oxygen, water temperature & turbidity in Cardiff Bay during September 2007, interpolated from data recorded at 30 sites.
Sourced from Alix (2010).

2.2.2 Phytoplankton Biomass

Samples were collected at the six YSI multiprobe sites within Cardiff Bay (Figure 2.4) by Cardiff Harbour Authority in accordance to a monitoring programme agreed with Natural Resources Wales. A spectrofluorometer consisting of bbe software and a FluoroProbe was used to determine the abundance of green algae, blue-green algae, diatoms, cryptophytes and total algal biomass within samples. The FluoroProbe was able to discriminate between algal classes through mathematical algorithms created using the spectral properties of each class. Chlorophyll-*a* is mainly responsible for red light emission, and accessory pigments close to chlorophyll-*a* affect fluorescence; creating a different excitation spectrum for each algal class.

Turbidity within a sample is automatically corrected by the bbe software. The software is also responsible for FluoroProbe functions, measurement initiation and results accumulation (Bbe-Moldaenke, 2015). Samples were taken at c1 m water depth on a weekly basis from January 2003 to April 2014.

The trichromatic method Mackereth et al. (1978) was used to determine chlorophyll-a, b and c spectrophotometrically in each of the 24 samples. The 500 ml samples were passed through glass fibre GF/F 0.45 µm filters before 5 mL 90% aqueous acetone and 0.2 mL magnesium carbonate was added to the filter paper to extract the pigments from the plankton concentrate. Once left to steep for 24 hours in the cold (0-5 °C) and dark the samples were analysed at 750 nm (to correct for turbidity), 664, 647 and 630 nm using a Cary 50 Probe UV-Visible Spectrophotometer. The results were then inserted into calculation 1 (Mackereth et al., 1978), see below, to determine the concentration of chlorophyll-a, b and c, and the amount of pigment per volume.

Calculation 1: Chlorophyll

To find the concentration of pigment, the corrected (after subtraction of 750 n values) optical densities were inserted into the following formula:

$$\text{Chlorophyll-}a \text{ (mg L}^{-1}\text{)} = 11.85 * (\text{OD}_{664}) - 1.54 * (\text{OD}_{647}) - 0.08 * (\text{OD}_{630})$$

$$\text{Chlorophyll-}b \text{ (mg L}^{-1}\text{)} = 21.03 * (\text{OD}_{647}) - 5.43 * (\text{OD}_{664}) - 2.66 * (\text{OD}_{630})$$

$$\text{Chlorophyll-}c \text{ (mg L}^{-1}\text{)} = 24.52 * (\text{OD}_{630}) - 7.60 * (\text{OD}_{647}) - 1.67 * (\text{OD}_{664})$$

Following the calculation of pigment concentration, the values were inserted into the following equation to calculate the amount of pigment per unit volume:

$$\text{Chlorophyll } x, \text{ mg/m}^3 = (\text{Chl } x * \text{ extract volume, L}) / \text{ volume of sample, m}^3$$

2.2.2.1 Phytoplankton Photophysiology

A Walz Water (Walz GmbH Germany) Pulse Amplitude Modulated (PAM) fluorometer was used in the field to determine the photosynthetic activity of the phytoplankton population at

each site in Cardiff Bay, at surface, middle and bottom water depths, using rapid light curves (RLC) (Perkins et al., 2006). Samples were exposed to eight 30 second incremental light steps at the end of which a saturating light pulse of approximately $7,600 \mu\text{mol photons m}^{-2} \text{sec}^{-1}$ was applied for 600 ms to measure the variable fluorescence rise to the maximum fluorescence yield, F_m' . Photosystem II quantum efficiency (Oxborough et al., 2000, Lawson et al., 2002, Perkins et al., 2002, 2010) was then determined from the change in variable fluorescence as a fraction of maximum fluorescence $(F_m' - F')/F_m'$ (where F' is the operational fluorescence yield at each light step) and this was used within Equation 2 (see below), to determine relative electron transport rate (rETR) and used as a proxy for photosynthetic productivity (Sakshaug et al., 1997, Perkins et al., 2001, Perkins et al., 2002, Perkins et al., 2006, Perkins et al., 2010).

Calculation 2: Relative electron transport rate (rETR)

Equation 1: $\text{rETR} = ((F_m' - F')/F_m') * (\text{PAR}/2)$

Rapid light curves (RLCs) of rETR against light intensity (PAR) were constructed using the model of Eilers and Peeters (1988), estimating the maximum electron transport rate (rETR_{max}), the maximum light use efficiency (α) and the light saturation coefficient (E_k) calculated as $(\text{rETR}_{\text{max}} / \alpha)$. Curve fitting was achieved using custom written R scripts (R Core Team, 2013) with light curve parameters a, b and c tested for significance at $p < 0.01$. Only RLCs that reached saturation and had significant fits using the Eilers and Peeters (1988) model were reported.

2.2.2.2 Phytoplankton Identification

Phytoplankton samples were collected in 500 ml polyethylene bottles at each of the present study water sampling sites (Figure 2.4) using a pump system on a fortnightly basis. A weighted hose was lowered into the lake to the required depth and a suction pump allowed water to be pumped to the surface. This enabled homogeneous samples to be collected from the surface, mid-depth and bottom of the water column at each of the eight sites (totalling 24 samples per fortnight). Samples were stored with sufficient air space and preserved with 1.5 ml Lugol's

solution (containing potassium iodide, iodine crystals, distilled water and glacial acetic acid) and kept in a cool, dark environment (Tomas, 1997) until identification.

Following collection, sedimentation was used to concentrate the phytoplankton samples as it is a nonselective and non-destructive method. A minimum of seven days was allowed for settling to take place before siphoning to create a concentrated sample for identification. As cell numbers were low, surface, mid-depth and bottom samples were combined and concentrated before identification. Phytoplankton was identified using a Sedgewick rafter chamber with a volume of 1 mL using a Zeiss Axiovert 35 inverted light microscope with a magnification of 200X. Identification to genus was carried out and 300 individuals counted per sample.

2.2.3 Bottom-Up Control Data

Water samples were collected by CHA as part of routine sampling at the six numbered YSI multiprobe sites within Cardiff Bay (Figure 2.4) in accordance to a monitoring programme agreed with Natural Resources Wales (NRW), on a monthly basis between January 2003 and April 2014 at a water depth of c1 m. These water samples were analysed for ammonium, nitrate, nitrite, orthophosphate, total phosphorus (TP) and silica by the National Laboratory Services (NLS) (Cardiff Harbour Authority, unknown). Ammonium concentrations were determined by NLS by using salicylate and dichloroisocyanurate in the presence of sodium nitroprusside to form a blue colour, measured photometrically at 660nm. Nitrate was calculated by subtracting nitrite from total oxidised nitrogen calculations determined spectrophotometrically. Nitrite was reacted with sulphanilamide and N-(1-naphthyl)-ethylenediamine dihydrochloride, in the presence of acid to produce an azo dye which was measured spectrophotometrically at 540 nm. NLS determined orthophosphate concentrations using molybdate and antimony potassium tartrate under acid conditions to form a complex which, when reduced with ascorbic acid, produces an intense blue colour, the absorbance of which was measured photometrically at 880 nm (National Laboratory Service, 2016a). TP was analysed by NLS through the oxidation of samples by persulphate, followed by hydrolysis by boiling under acidic conditions to convert them to orthophosphate. Subsequently ammonium molybdate was used under acidic conditions to form a 12-molybdophosphoric acid complex. Antimony (III) was added as a catalyst and the

complex reduced ascorbic acid to form a heteropoly compound (phosphomolybdenum blue), the absorbance of which was analysed using an AquaKem Discrete analyser (National Laboratory Service, 2016b). Silica was determined by reacting silicates in solution with molybdate under acidic conditions to form a silicomolybdate complex. Reduction with ascorbic acid to silicomolybdate blue was measured spectrophotometrically at 760 nm (National Laboratory Service, 2016a).

NRW calculated the concentrations of TP and nitrate entering Cardiff Bay from the River Taff and River Ely by sample collection on a monthly basis at NRW site 17001 (ST16935 78275) on the River Taff and NRW site 16039 (ST14542 76829) on the River Ely. These sites were the closest NRW TP and nitrate monitoring sites to Cardiff Bay. TP data was collected from January 2003 until December 2008 inclusive and nitrate data was collected from January 2003 until June 2012 inclusive.

At each of the eight present data sites in Cardiff Bay (Figure 2.4) 500 ml water samples were collected in acid-washed polyethylene bottles (washed in 10% sulphuric acid and rinsed in UHP water 6 times), kept in the dark and immediately refrigerated at 4°C. Samples were collected using a pump system at all eight sites at c1 m (surface sample), c1 m above the lake bed (bottom sample) and in the middle of the water column (mid-depth sample). The bottom and mid-depth sample depths varied between all sites due to alterations in the overall water column depth (Figure 2.2).

2.2.3.1 Total Phosphorus (TP)

Total phosphorus (TP) concentration of the 24 water samples (three samples taken at each of the eight sites) collected once a fortnight was determined using the persulphate digestion method adapted from (Mackereth et al., 1978, Perkins & Underwood, 2000, Eaton et al., 2005). The organic and inorganic matter within the unfiltered 5 ml samples was digested using persulphate. Samples of 5 mL were added to 5 mL deionised water before being hydrolysed using 0.2 mL sulphuric acid and 0.08 g ammonium persulphate and placed in a water bath at 100 °C for an hour. 1N sodium hydroxide was added to neutralise the samples, using phenolphthalein as an indicator. Subsequently colourmetric analysis was carried out on the samples using the ascorbic acid method. A cuvette containing 0.8 mL of sample, 0.1 mL of

reagent 1 (5N sulphuric acid and ammonium molybdate to form molybdophosphoric acid, and potassium antimonyl tartrate to form phosphomolybdic acid) and 0.1 mL of reagent 2 (ascorbic acid to reduce the sample to molybdenum blue) was inserted into a spectrophotometer. The intensity and therefore the concentration of which was determined spectrophotometrically at 880nm using a Cary 50 Probe UV-Visible Spectrophotometer. Standards were calculated at 1, 5, 10, 15 and 20 μM P- PO_4 using sodium dihydrogen phosphate monohydrate.

2.2.3.2 Soluble Reactive Phosphorus (SRP)

Subsamples from each of the 24 water samples taken fortnightly were filtered through a glass fibre GF/F 0.45 μm filter before the ascorbic acid and colourmetric methods adapted from (Mackereth et al., 1978, Perkins & Underwood, 2000, Eaton et al., 2005) were used to analyse soluble reactive phosphorus (SRP) concentrations. Reagent 1 (0.1 mL) was made up of 5N sulphuric acid and ammonium molybdate to form molybdophosphoric acid, with the addition of potassium antimonyl tartrate to form phosphomolybdic acid and added to the 0.8 mL sample. Ascorbic acid (0.1 mL) was then added to the sample to reduce it to molybdenum blue, the intensity and therefore the concentration of which was determined spectrophotometrically at 880nm using a Cary 50 Probe UV-Visible spectrophotometer. Standards were produced covering the concentrations of 1, 5, 10, 15 and 20 μM P- PO_4 using sodium dihydrogen phosphate monohydrate.

Ion chromatography was also used to calculate filtered SRP. The samples were filtered using a glass fibre GF/F 0.45 μm filter and 10 ml samples processed through a Dionex DX-80 Ion Analyser (DX-80). See section 2.2.3.3 for detailed method.

2.2.3.3 Nitrate and Nitrite

Nitrate-N and nitrite-N concentrations were analysed using ion chromatography; 10 ml water samples were filtered through a glass fibre GF/F 0.45 μm filter and processed through a Dionex DX-80 Ion Analyser. The DX-80 contains a liquid eluent, high-pressure pump, sample injector, separator column, chemical suppressor, and a conductivity cell. This performs isocratic ion analyses using compressed conductivity detection. Once the ion analyser has been calibrated

using a standard solution, data contained in a sample can be compared to that of the standard allowing sample ions to be identified and quantified. A chromatogram is created using chromatography software and each peak is automatically converted into a sample concentration (Dionex Corporation, 2002).

2.2.3.4 Silica

Subsequent to filtering the 24 water samples through a glass fibre GF/F 0.45 μm filter, reactive silica was measured using the molybdate method adapted from Perkins & Underwood (2000) and Eaton et al. (2005). 0.4 mL of molybdate reagent (para-molybdate/ammonium heptamolybdate, deionised water and 12N hydrochloric acid) was added to 1 mL of sample, forming silicomolybdate, phosphomolybdate and arsenomolybdate complexes. A reducing reagent consisting of metol sulphite, oxalic acid and 50% sulphuric acid was added (0.6 mL) to decompose the phosphomolybdate and arsenomolybdate and reduce the silicomolybdate complex, resulting in a blue colour. Absorbance was then measured spectrophotometrically at 810nm (APHA, 1995) after 3 hours using a Cary 50 Probe UV-Visible Spectrophotometer. Standards were produced covering the concentrations of 0, 20, 60, 100, 120 and 150 μM Si- $\text{Na}_2\text{O}_3\text{Si}$.

2.2.3.5 Ammonium

The phenate method adapted from Eaton et al. (2005) and Mackereth et al. (1978) was used to detect ammonium in each of the 24 glass fibre GF/F 0.45 μm filtered 5 mL samples. In the presence of hypochlorite, ammonium reacts with phenol (0.2 mL made from $\geq 89\%$ phenol and 95% v/v ethyl alcohol) and an alkaline base (0.2 mL oxidising solution consisting of alkaline citrate solution and sodium hypochlorite) to form an indophenol blue. The latter was catalysed by sodium nitroprusside (0.2 mL) and the absorbance measured spectrophotometrically at 640nm using a Cary 50 Probe UV-Visible Spectrophotometer. Standards were produced covering the concentrations 1, 3, 5, 7, and 10 μM N- NH_4Cl .

2.2.3.6 Sediment Coring

Three sediment cores were collected at each of the eight sampling sites (Figure 2.4); once during winter and once during summer to assess nutrient exchange across the water-sediment interface.

To assess the potential phosphorus release from the lake sediment within Cardiff Bay and the availability of phosphorus to the phytoplankton community, twenty-four sediment cores were taken from Cardiff Bay during March 2014 for a winter representative sample and July 2014 for a summer representative sample. Three cores were taken at each of the eight sampling sites (Figure 2.4) using a multi-corer (Duncan and Associates, Cumbria, UK), operated from Cardiff University's research vessel "Guiding Light" (Figure 2.6).

Each core was cut into 3 cm sections, up to a maximum of 15cm depth and sediment phosphorus fractionation, adapted from (Perkins & Underwood, 2001b) was carried out. Each section was homogenised before 1 g was weighed and placed in an oven for 24 hours at 85°C to calculate sediment water content (loss of weight). Samples were then incinerated at 550°C for 1 hour to calculate ash-free weight. Iron/aluminium (Fe/Al) bound phosphorus calculations required 17 hours steep with 0.1 M sodium hydroxide addition to wet samples, calcium (Ca) bound phosphorus required the addition of 0.5 M hydrochloric acid to wet samples for 24 hours and labile phosphorus calculations required the addition of 1 M ammonium chloride to wet samples which was neutralised with hydrochloric acid for 2 hours, before repeating. Following these additions and subsequent steep times, all samples were centrifuged at 4500 rpm for 10 minutes before soluble reactive phosphorus analysis was carried out on neutralised (using sodium hydroxide) samples, as in section 2.3.1.2. The labile phosphorus data were then subtracted from the Ca and Fe/Al bound fractions.



Figure 2.6: Multi-corer on the Guiding Light Vessel.

2.2.3.7 In-situ Nutrient Addition Mesocosms

Mesocosms were set up in Cardiff Bay at site C (Figure 2.4) from 20th – 27th May 2015 to determine if the phytoplankton community was phosphorus or nitrogen limited. A total of fifteen 20 L polyethylene bags, each filled with 15 L of lake water to allow enough head space for gas exchange were deployed into the Bay just below the water surface (Figure 2.7). Five bags acted as controls, with no nutrient addition; five bags contained 1 mg P-PO₄ l⁻¹ by adding sodium dihydrogen phosphate (NaH₂PO₄) for phosphorus addition; and the remaining five contained 2 mg N-NH₄ l⁻¹ by adding ammonium chloride (NH₄Cl) for nitrogen addition.

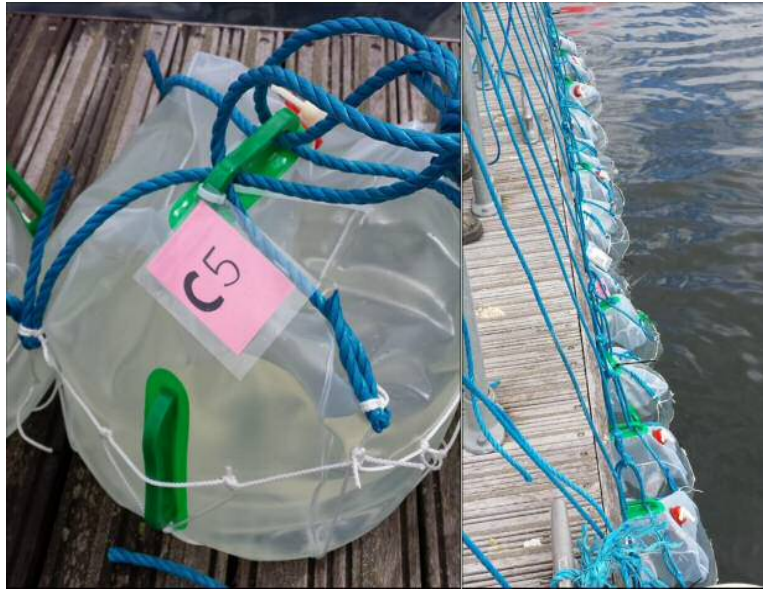


Figure 2.7: Mesocosm bags deployed in Cardiff Bay.

At the time of deployment and extraction the Bay water temperature and DO concentrations were recorded at 1 m depth using YSI 6600 EDS V2 Water Quality Monitoring Sonde and YSI 650MDS Data Display/Logger. Additionally, a YSI 6560 Temperature /Conductivity probe was used to collect water temperature data and a YSI 6150 ROX Optical DO probe was used to collect DO data. A Secchi disk reading was also taken.

One 500 ml water sample was taken from each mesocosm bag during deployment (after nutrient addition) and extraction after 7 days and stored at 0-5 °C in 500 ml polyethylene bottles until processed later the same day. The sample was used for nutrient and chlorophyll analysis.

2.2.4 Top-down Control Data

Zooplankton samples were taken once a fortnight at each of the eight present data sites (Figure 2.4) using a fine mesh (60 μm) conical zooplankton net. The net was weighted and surrounded by a cylindrical tube to enable vertical samples to be taken through the water column without the net dragging through the lake bed as it reached the bottom of the water column (Figure 2.8).

The net was rinsed, and the samples fixed with 70% methylated ethanol after each trawl. The volume of water that passed through the net for each trawl was calculated as the product of the net aperture area and the length of trawling (depth net released to).



Figure 2.8: Zooplankton net with conical tubing.

Samples were preserved with 0.04% rose bengal stain and kept cool and dark until identification. Individuals were identified and counted to genus level using a XTC-3A Dual Magnification GX microscope at 30X and a circular rotating zooplankton counting tray. Subsampling was not necessary as zooplankton numbers were low (<200 per 500 ml).

A study undertaken by Alix (2010) mapped the zebra mussel (*Dreissena polymorpha*) population of Cardiff Bay using side-scan sonar (SSS) during November 2007, coupled with grab samples to ground-truth any areas of apparent zebra mussel formation. To determine if there had been any population changes since 2007, this survey was repeated in July 2014.

Side scan sonar; C-Max CM2 towfish linked to a Coda Octopus 760 geophysical acquisition unit and a Coda Octopus F180 positioning and attitude system, was used to map the bed of Cardiff Bay, and the presence of zebra mussels within it using a series of 25 transects (Figure 2.9) with a sonar range setting of 25 m and a frequency of 350. Data were processed using Hypack side scan sonar, targeting and mosaicking software.

During July 2012 three grabs were taken at each of the 20 sites previously surveyed by Alix (2010) in 2007 (Figure 2.10) using a manual grab with the capacity of 2 L and a sample area of 27 cm². The grab was manually lowered to the lake bed with the jaws locked open, once it hit the lake bed and was pulled sharply upwards the grab collected the underlying substrate. Any mussels found within each sample were bagged, labelled and frozen prior to counting and measurement. In the laboratory the mussels were identified and sorted into age categories by size (Figure 2.11); <5 mm individuals were less than 6 months old, individuals 6–12 months had a shell length of 5-8 mm, individuals 1+ years had a shell length of 9-16 mm, 2+ years had a shell length of 17-24 mm and 3+ years had a shell length of >25 mm and counted.

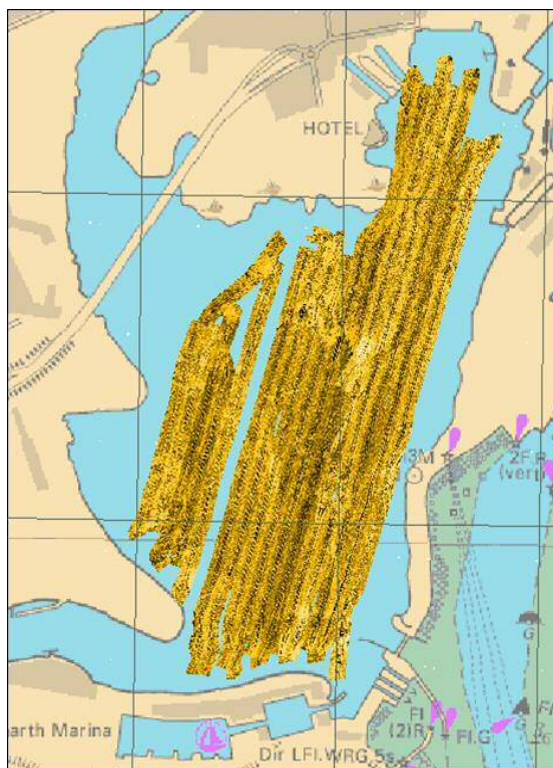


Figure 2.9: Side scan sonar transects undertaken in Cardiff Bay

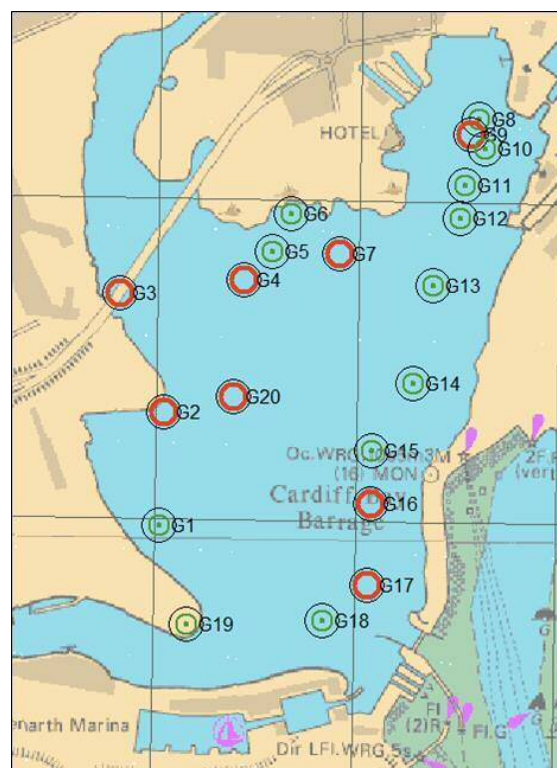


Figure 2.10: Location of grab sites within Cardiff Bay. Green sites indicate grabs where zebra mussels were present. Red sites indicate grabs where zebra mussels were absent



Figure 2.11: Zebra mussel count from grab 18B in Cardiff Bay

THE ENVIRONMENTAL VARIABLES & PHYSICAL PROPERTIES OF CARDIFF BAY

3.1 Summary

The environmental variables and physical properties of the artificial lake of Cardiff Bay, South Wales were analysed between January 2003 and April 2014, to understand how and why the phytoplankton community structure and abundance within the lake alters with time. The lake was found to be polymictic as a result of fast flushing rates and the presence of an aeration system preventing the water column from stratifying. The lake was subject to natural seasonal changes and is likely to be affected by climatic changes in the future, particularly during summer where temperatures are predicted to increase, and precipitation extremes will become more frequent. These conditions will require the aeration system to be used more frequently to maintain lake dissolved oxygen concentrations. The impact of the aeration system and the fast lake flushing rate is pivotal to the processes that take place within the lake and the resulting management required to maintain a healthy lake system.

3.2 Introduction

The environmental factors of a lake such as temperature, precipitation and light levels are associated with harmful algal blooms (HABs). This has a relationship with the characteristics of the lake such as dissolved oxygen concentrations and stratification (Jacoby et al., 2000). Therefore, understanding the environmental variables and physical properties of the lake of Cardiff Bay is paramount for successful management of the system in the future. The potential for eutrophication can be established and the pathways of nutrient acquisition determined (Anderson et al., 2002), both of which are key to the management of lake water quality.

3.2.1 Lake Thermal Structure

The annual cycle of a lake typically involves the development and reduction of vertical thermal structures. These structures differ in size and depth between lakes, depending on their geographical, climatic and morphological conditions and often cause one or more periods of vertical thermal stratification to occur within a 12-month period (Ambrosetti et al., 2003). Understanding the timing and extent of stratification enables the changes that take place within a phytoplankton population to be understood. Phytoplankton blooms often coincide with stratification events due to the reduction in mixing of the surface layer (Cloern, 1987). Therefore, the onset of phytoplankton growth is predominantly determined by mixing conditions as they affect the exposure of phytoplankton to light and the availability of nutrients (Cloern, 1987, Peeters et al., 2007).

Lake surface water temperature is often highly correlated with air temperature and alterations in temperature affect water column stability and stratification (Adrian et al., 2009a). As a result, these fluctuations in water temperature the resultant mixing of the water column causes variations in nutrient availability, all of which ultimately influence the lake phytoplankton biomass (Kumari et al., 2018).

3.2.2 River Inputs and Lake Turbidity

High turbidity environments are often created by river inputs containing large amounts of suspended particulate matter and/or the resuspension of bottom sediments. The extent to which the water column is mixed and therefore the amount of suspended sediment present in the water column plays an important role in light attenuation; a higher sediment load causes light to be rapidly attenuated, reducing the photic zone (Cloern, 1987, Blom et al., 1994).

River discharge also affects lake nutrient supplies, making it a major integrating factor related to important lake processes that affect phytoplankton communities (Descy, 1993). Precipitation not only has the ability to mix the water column of the lake, reducing lake stability and increasing DO concentrations (Kumari et al., 2018), during reduced precipitation during summer months it also causes river discharge to decrease (Acharyya et al., 2012, Defra, 2012c).

Understanding the mean flushing rate of a lake has long been a problem of fundamental importance for limnology and lake management as it is required to identify the proportions and dynamics of the chemical substances dissolved in water, or the rate at which the processes of concentration, dilution and permanence of substances within the lake occur, with resulting implications for the water quality (Ambrosetti et al., 2003). Before the flushing rate of a lake can be calculated, the internal physical processes of the water body must first be assessed (Ambrosetti et al., 2003). The measurement of inputs and outputs of a lake enables the movement of nutrients in and out of a lake to be tracked and the flushing rate of these nutrients is usually determined by river discharge (Jones & Elliott, 2007).

3.2.3 Lake Dissolved Oxygen Concentrations

The distribution of dissolved oxygen (DO) and its concentration within the water column is one of the key ways to ensure the health of a lake (Hasan et al., 2014). The main sources of oxygen in a water body are the air, and often most importantly, as a by-product of photosynthesis. Concentrations are also affected by lake aeration and vertical mixing (Bella, 1970, Hasan et al., 2014).

Changes in air temperature affect the concentrations of DO in lakes (Adrian et al., 2009a, Fang & Stefan, 2009, Murphy et al., 2009), where an increase in lake temperature has the potential to indirectly effect lake DO concentrations, which may decrease as the duration of summer stratification increases (Livingstone & Lotter, 1998).

3.2.4 Climatic Pressures on Lake Ecosystems

The North Atlantic Oscillation influences lake water temperatures in the Northern Hemisphere, having an indirect control on lake ecological processes (Gerten & Adrian, 2001). Despite the difficulty in achieving good predictive models of the NAO (Massei et al., 2007), the NAO can be used as a proxy for forces that regulate aquatic ecosystems (Ottersen et al., 2001). Ecological responses to changes in the NAO include the variation in the timings of reproduction, population dynamics, abundance, spatial distribution, and interspecific relationships such as competition and predator-prey relationships (Ottersen et al., 2001), making the presence of the

NAO an important factor to consider when assessing the environmental variables and physical processes that take place within Cardiff Bay.

There are several pressures facing lakes in the future as a result of changes in climate. The physical properties within lakes such as the amount and temporal variability of river discharge is affected (Ormerod, 2009), which can cause lake levels to decline and alter nutrient concentrations within a lake (Mortsch et al., 2000). Rising temperature rises have been found to cause an increase in water column stability and lengthen stratification events. All of these pressures have the ability to reduce lake primary productivity, and ultimately have an impact on higher trophic organisms such as fish (O'reilly et al., 2003).

3.2.5 Managing Lakes in Response to Physical Changes

Using a multi-proxy, whole-lake approach to monitor and assess the environmental and physical variables of Cardiff Bay, including the variations in air and water temperature, and precipitation events, is paramount to the understanding and analysis of the physiology, development and dispersal of phytoplankton species within the lake. One of the fundamental requirements for the successful management of this lake is the ability to reduce the pressure on the lake from changes in climate, to ensure that rather than degrading in health, the lake can be restored if necessary (Spears et al., 2015) and the impacts on the lake food-web and water quality minimised or prevented.

There are many uses of an aeration system within a lake. They are very effective at mixing a water column to control stratification, phytoplankton growth can be limited to prevent eutrophication, and DO concentrations can be increased to allow fish propagation and prevent gas escaping from benthal deposits (Hasan et al., 2014).

There are increasing concerns that hydrological extremes and the nature of changes occurring as a result will put new pressures on lake systems (Jones et al., 2009). Therefore, the effective management of a lake system not only requires us to adapt our behaviour to accommodate these changes, but the physical lake environment must be able to mediate them. One of the most important pressures resulting from the global changes in climate is the variation in water temperature (Livingstone & Lotter, 1998, Jones et al., 2009).

3.3 Hypotheses

To determine the environmental variables and physical properties of Cardiff Bay the following thesis objectives will be addressed within this chapter:

1. Environmental variables will be assessed using air and water temperature data, precipitation levels and the level of light entering the system.
2. Physical lake properties will be analysed using dissolved oxygen concentrations, river discharge rates, lake flushing rate and water column stability.

Using data collected between January 2003 and April 2014 in Cardiff Bay, the following hypotheses have been tested:

Seasonal air temperature and precipitation changes will induce seasonal lake water temperature change.

The photic depth of the lake will fluctuate seasonally due to changes in water column stability. Spatial variation in photic depth will occur due to differences in sampling site characteristics (e.g. susceptibility to mixing, sediment re-suspension).

Lake dissolved oxygen concentrations will vary spatially and temporally due to the influence of water temperature and thermal stratification.

High precipitation rates will cause an increase in river discharge and therefore cause lake flushing rates to quicken. Temporal changes in flushing rate will occur as a result of seasonal changes in precipitation.

3.4 Materials and Methods

The 200 ha artificial lake of Cardiff Bay has two river inputs, the River Taff and River Ely. The water level of the lake is maintained at 4.5 m above ordnance datum by five sluice gates. It is a relatively shallow lake with a mean depth of c4.5 m and a maximum depth of c13 m. Within the lake there is a forced aeration system used to mix the water column to ensure that

the lake is sufficiently oxygenated with a minimum concentration of 5 mg L⁻¹ throughout (Cardiff Harbour Authority, unknown).

3.4.1 Data Processing

Data were collected from January 2003 until April 2014. Air temperature ($n = 692,188$) and precipitation ($n = 481,710$) were collected at the Davis Weather Station on the banks of Cardiff Bay continuously at 15-minute intervals. Mean air temperature and precipitation were calculated on an annual and seasonal basis.

By determining the hottest 10% and coldest 10% of temperatures experienced from January 2003 until April 2014, the number of ‘hot’ days and nights (those experiencing the warmest 10% of temperatures) and ‘cold’ days and nights (those experiencing the coldest 10% of temperatures) were calculated (Solomon et al., 2007).

Surface (c1 m depth) water temperature was recorded at 15-minute intervals between January 2003 and April 2014 (totalling $n = 948,838$) at site 4 ($n = 152,014$), site 5 ($n = 151,999$), site 6 ($n = 149,989$), site 9 ($n = 164,652$), site 10 ($n = 165,092$) and site 17 ($n = 165,092$). Mean site-specific water temperature was calculated on a seasonal and monthly basis. Mean whole-Bay surface water temperature values were calculated using all surface water temperature data from each site within the Bay as no spatial variation was found between sites and calculated seasonally and monthly.

Water temperature was also recorded at c1 m intervals throughout the water column at site 4 up to c1.5 m depth ($n = 154$), site 5 up to c7.5 m depth ($n = 344$), site 9 up to c2.5 m depth ($n = 196$), site 10 up to c7.5 m depth ($n = 405$), site A up to c6.0 m depth ($n = 367$), site B up to c4.0 m depth ($n = 242$), site C up to c3.0 m depth ($n = 204$) and site D up to c3.0 m ($n = 216$), between April 2012 and April 2014 once every fortnight (totalling $n = 2,128$). This data was used to calculate mean monthly water temperature values throughout the water column at each site.

Mean annual, seasonal and monthly whole-Bay water temperature and dissolved oxygen (DO) values were calculated using all data collected at 1 m depth from sites 4, 5, 6, 9, 10 and 17 (see

Chapter 2 for site locations). Individual site means for water temperature and DO were calculated for sites 4, 5, 6, 9, 10 and 17 using all data collected at 1 m on an annual, seasonal and monthly basis.

Photic depth was calculated (see Chapter 2, Section 2.2.1) once a fortnight between April 2012 and April 2014 (totalling $n = 416$) at sites 4, 5, 9, 10, A, B, C and D. Mean monthly site-specific values were calculated as well as mean monthly whole-Bay photic depth values using all photic data from each site within the Bay.

River discharge was recorded every 15 minutes at Pontypridd on the River Taff ($n = 402,024$) and at St Fagans on the River Ely ($n = 402,024$). Mean seasonal and monthly River Taff and River Ely discharge values were calculated (see below) as well as mean seasonal and monthly combined river discharge values which were calculated using the sum of the River Taff and River Ely discharge values ($n = 804,048$).

Lake flushing rate was calculated for Cardiff Bay (see below) using mean monthly River Taff and River Ely discharge data ($n = 252$ data points). No other significant inflow sources are known for the lake and the only outflow is through Cardiff Bay Barrage. The level of the lake is maintained at c4.5 m OD therefore the volume of the lake is consistent at 6,734,000 m³. Due to this level of management the same amount of water must flow out of the Bay as flows in. Lake flushing rate was calculated using the following equation:

$$\text{Flushing Rate (day}^{-1}\text{)} = (\text{River Taff discharge (m}^3\text{ day}^{-1}\text{)} + \text{River Ely discharge (m}^3\text{ day}^{-1}\text{)}) / \text{Volume of Cardiff Bay (m}^3\text{)}$$

Seasonal and monthly lake flushing rate values were then calculated.

Seasons were calculated from winter 2003 until winter 2014 and categorised as follows: Winter refers to the January and February of the named year and the December of the previous year; spring refers to the months of March, April and May; summer encompasses June, July and August; autumn incorporates September, October and November.

At various times throughout the 11-year study period some water temperature and DO data points were missing due to technical errors. Due to the extensive data set, gaps in data were not filled in. All graphs contain standard error bars.

3.4.2 Statistical Analysis

Mean monthly values for all data sets were used individually to conduct the decomposition of additive time series in ‘Relevance’ (R) (R Core Team, 2013), script:

```
xTimeSeries<-ts(x, frequency=12, start=c(2003,1)).  
plot.ts(xTimeSeries).  
library("TTR").  
xTimeSeriesSMA25<-SMA(xTimeSeries,n=25).  
plot.ts(xTimeSeriesSMA25).  
xTimeSeries<-decompose(xTimeSeries).  
xTimeSeries$seasonal.  
xTimeSeries$trend. xTimeSeries$random.  
plot(xTimeSeries).
```

Pearson’s correlation was undertaken on monthly and seasonal data for all parameters sets using R.

Nested Analysis of Variance (ANOVA) was carried out using MINITAB statistical software to determine if there was any variation in data on a seasonal basis. Single-factor ANOVA was used on data sets shown to be significant in nested ANOVA to identify more specifically where differences lie using *post hoc* Tukey test ($\alpha = 0.05$) in MINITAB.

Monthly data were used for linear regression analysis using R to determine if there was a “cause and effect” relationship between the meteorological and hydrological variables in Cardiff Bay. Following significant linear regression results, multiple regressions using R were undertaken using backward elimination to refine the “cause and effect” relationships.

3.5 Results

3.5.1 Temporal Variation in the Air Temperature of Cardiff Bay

The mean annual air temperature of Cardiff Bay fluctuated between 2003 and 2014, with warmer mean annual temperatures being experienced before 2007 (Figure 3.1). Between 2003 and 2014, 11 ‘hot’ days and nights were established (Table 3.1), defined by the IPCC as being within the hottest 10% of the temperatures experienced (Solomon et al., 2007), therefore having temperatures of 28.9 °C and warmer. The year with the most ‘hot’ days and nights was 2006, with seven occurring during July, all ranging from 29.6 to 32.8°C. Three ‘hot’ days and nights were experienced in 2003 (the warmest of which was on 14th July at 31°C) and one ‘hot’ day and night was experienced on 18th July 2013 with a temperature of 29.2°C. There were 23 ‘cold’ days and nights (Table 3.1), defined by the IPCC as being within the coldest 10% of the temperatures experienced (Solomon et al., 2007), therefore reaching -2.7°C and colder, between 2003 and 2014. Thirteen ‘cold’ days and nights were experienced in 2010, the coldest of which occurred on 7th December at -6.6°C.

A clear seasonal cycle in air temperature was apparent between January 2003 and April 2014 (Figure 3.2, seasonal panel), and air temperatures were significantly different between seasons ($F_{3,44} = 209.67$, $P < 0.001$). Mean seasonal air temperatures were at their lowest during winter (mean 6.3 ± 1.27 °C) and highest during summer (mean 16.6 ± 0.89 °C), with autumn being the second warmest season (mean 12.3 ± 0.87 °C), followed by spring (mean 10.01 ± 0.94 °C) (Figure 3.3).

There were statistically significant differences between mean monthly air temperatures in Cardiff Bay ($F_{11,135} = 121.03$, $P < 0.001$). January and February experienced statistically significant lower mean monthly air temperatures than April, May, June, July, August, September, October and November. March experienced statistically significant lower mean monthly air temperatures than April, May, June, July, August, September and October. April experienced statistically significant lower mean monthly air temperatures than May, June, July, August, September and October. May experienced statistically significant lower mean monthly air temperatures than June, July, August and September, but statistically significant higher

mean monthly air temperatures than November and December. June, July, August and September experienced statistically significant higher mean monthly air temperatures than October, November and December (Figure 3.4).

The winter North Atlantic Oscillation (NAO) Index (Figure 3.5) decreased from 2003 to 2006, showed a positive index in 2007, 2008, before sharply declining to the lowest winter NAO Index since records began in 2010 (Obsorn, 2015). The NAO Index showed a weak positive correlation with mean annual air temperature in Cardiff Bay (Appendix Figure 3.1). The NAO Index displayed a significant positive correlation during every season with air temperature, however these relationships were only strong during winter and autumn (Figure 3.6).

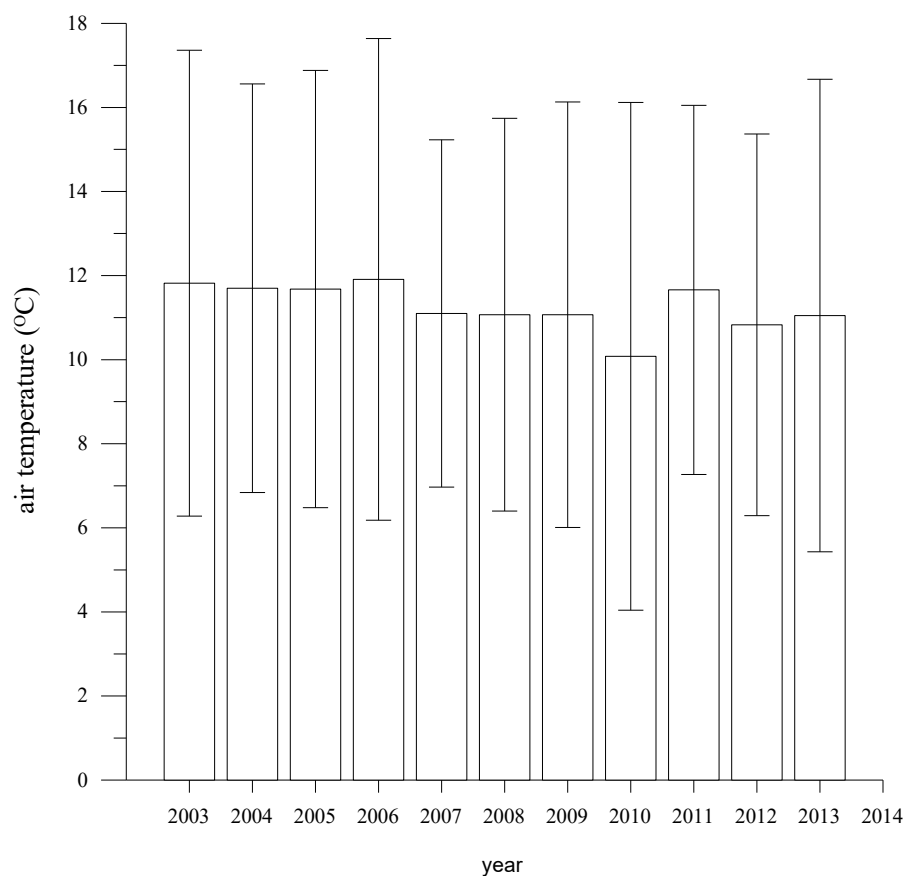


Figure 3.1: Mean annual air temperature of Cardiff Bay from 2003 until 2013 (standard error bars).

Table 3.1: ‘Hot’ and ‘Cold’ days experienced in Cardiff Bay between January 2003 and April 2014.

Date	‘hot’ day temperature (°C)	‘cold’ day temperature (°C)
08/01/2003		-2.8
12/01/2003		-3.0
14/07/2003	31.0	
15/07/2003	30.8	
09/08/2003	30.5	
29/12/2005		-3.4
02/07/2006	29.6	
03/07/2006	30.4	
04/07/2006	30.2	
17/07/2006	30.5	
18/07/2006	31.3	
19/07/2006	32.8	
25/07/2006	30.3	
06/01/2009		-3.1
07/01/2009		-3.6
04/01/2010		-3.2
08/01/2010		-3.2
28/11/2010		-4.8
29/11/2010		-3.2
03/12/2010		-3.4
06/12/2010		-4.8
07/12/2010		-6.6
08/12/2010		-3.2
18/12/2010		-3.8
19/12/2010		-4.2
20/12/2010		-4.7
25/10/2010		-5.4
26/12/2010		-5.4
21/01/2011		-2.9
31/01/2011		-3.3
03/02/2012		-3.2
04/02/2012		-3.7
08/02/2012		-3.2
11/02/2012		-2.9
18/07/2013	29.2	

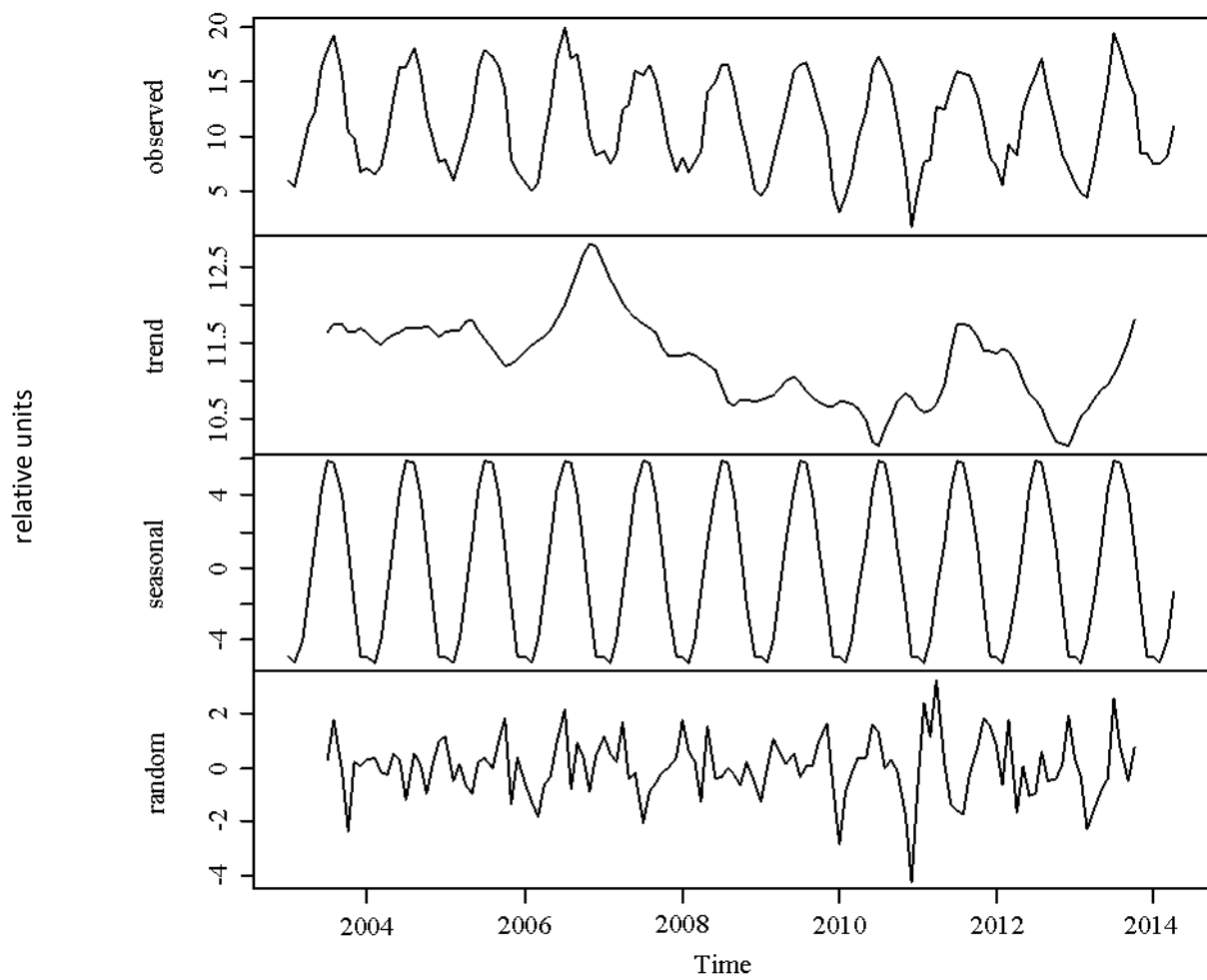


Figure 3.2: Time series analysis of air temperature in Cardiff Bay, using data from January 2003 until April 2014.

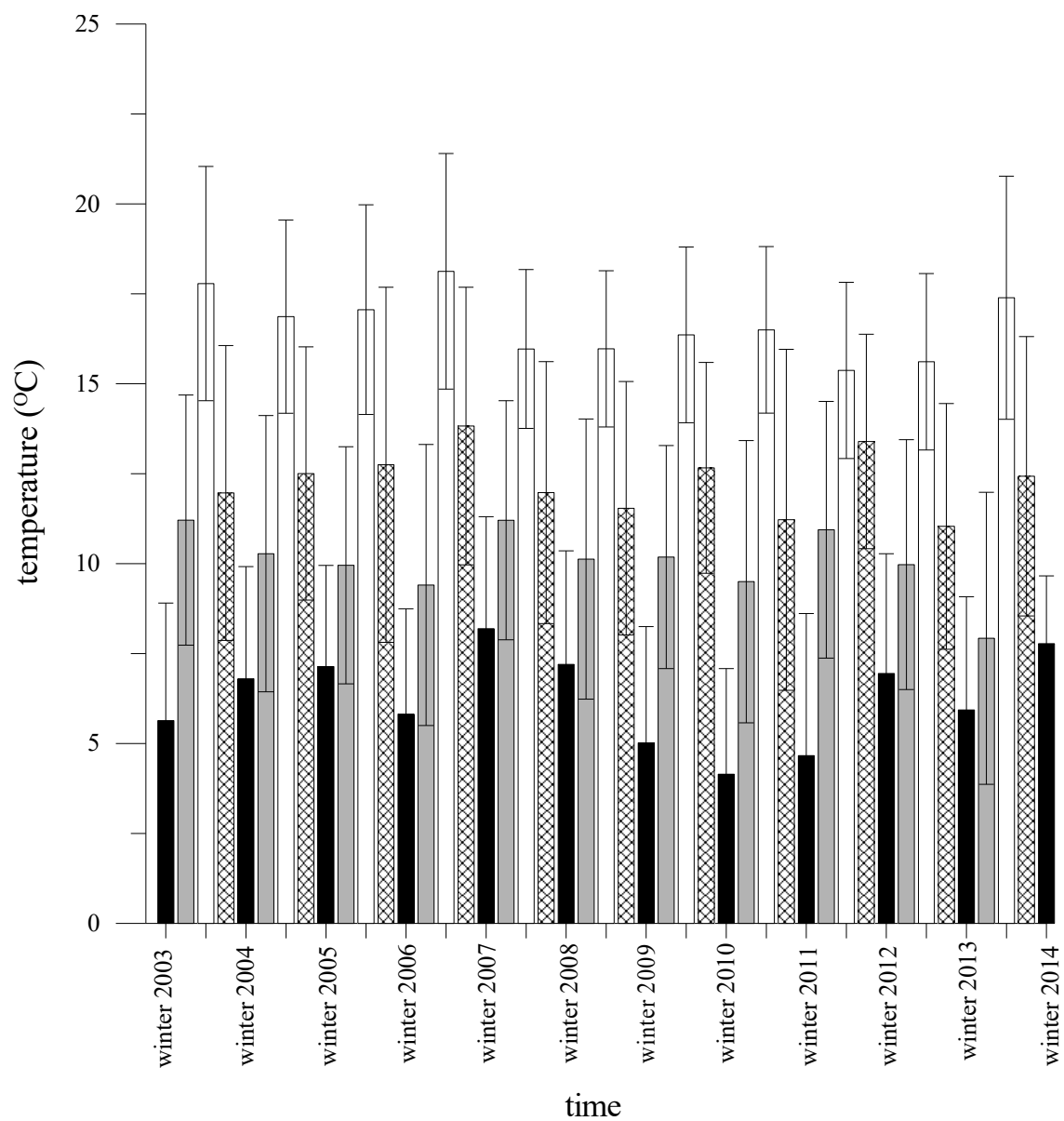


Figure 3.3: Mean seasonal air temperature in Cardiff Bay from winter 2003 until winter 2014. Winter: black bars. Spring: grey bars. Summer: white bars. Autumn: hatched bars (standard error bars).

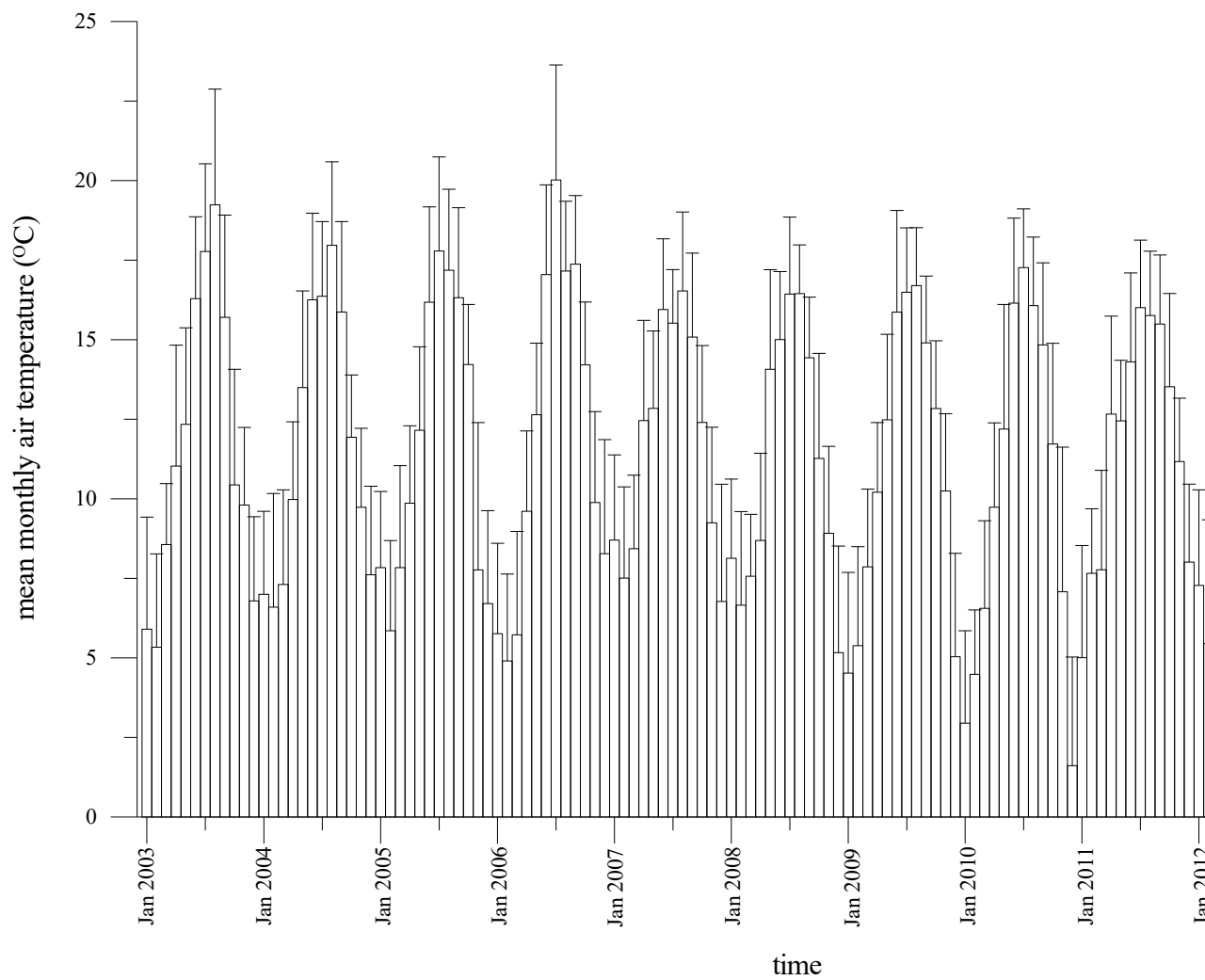


Figure 3.4: Mean monthly air temperature in Cardiff Bay from January 2003 until April 2014 (stan

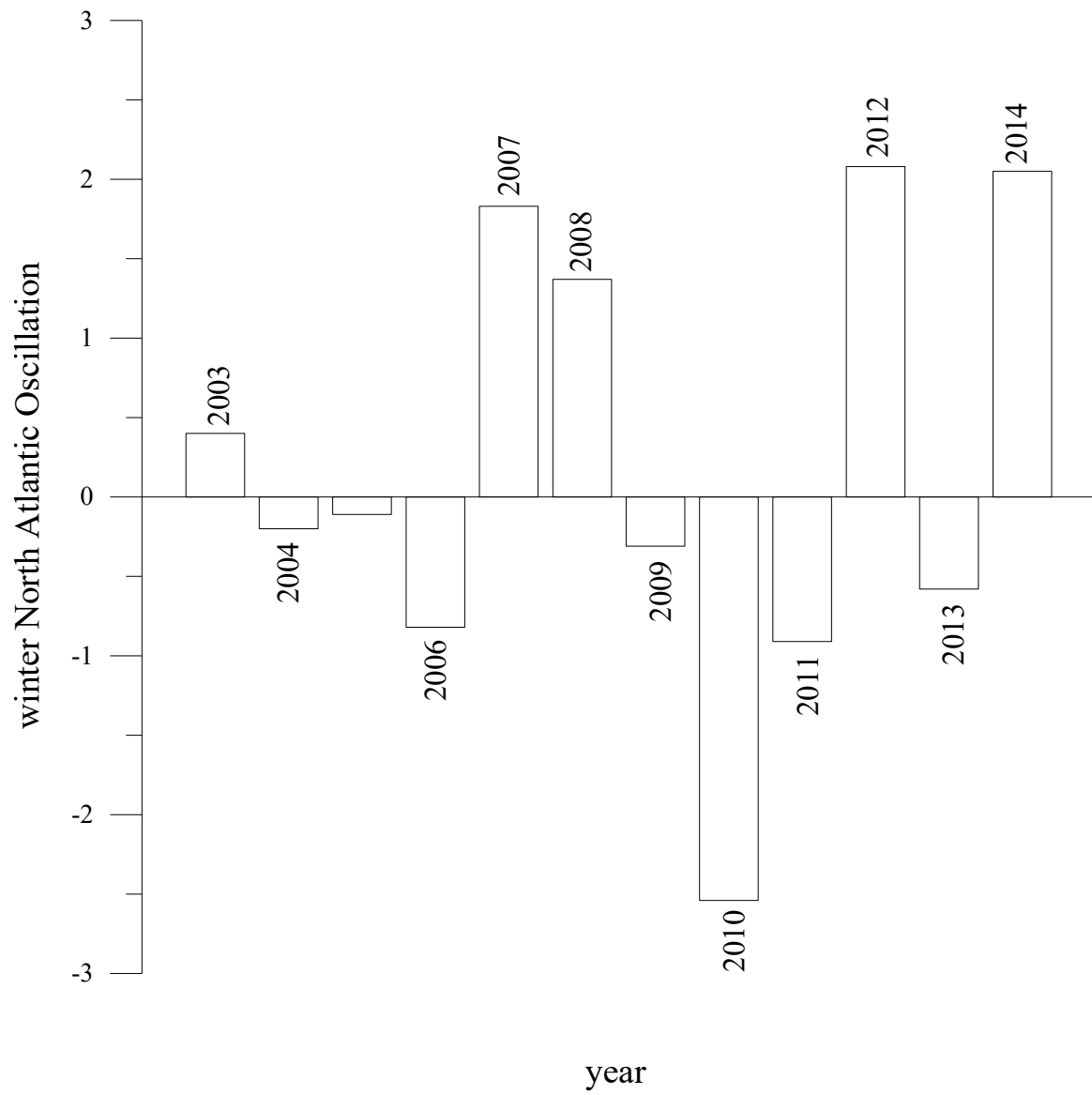


Figure 3.5: Winter North Atlantic Oscillation index 2003 – 2014 (Data from Osborne (2015)).

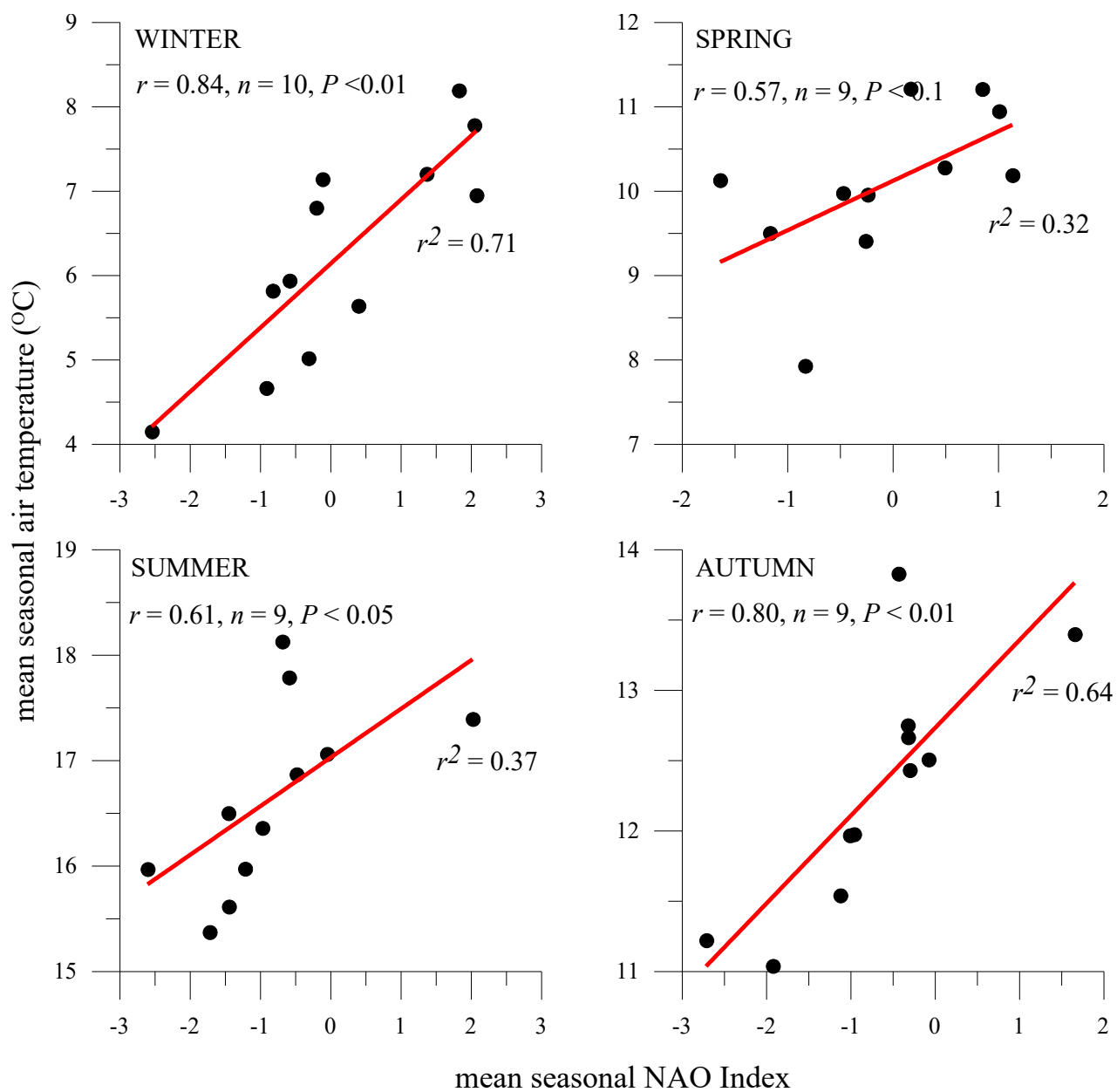


Figure 3.6: Mean seasonal North Atlantic Oscillation Index and mean seasonal air temperature of Cardiff Bay between winter 2003 and winter 2014.

3.5.2 The Relationship between Air & Surface Water Temperature of Cardiff Bay

Whole-Bay surface water temperature followed the same annual trend as air temperature, with the exception of 2010 (Figure 3.7). There was a strong significant positive correlation between mean monthly air temperature and mean whole-Bay monthly surface water (Figure 3.8).

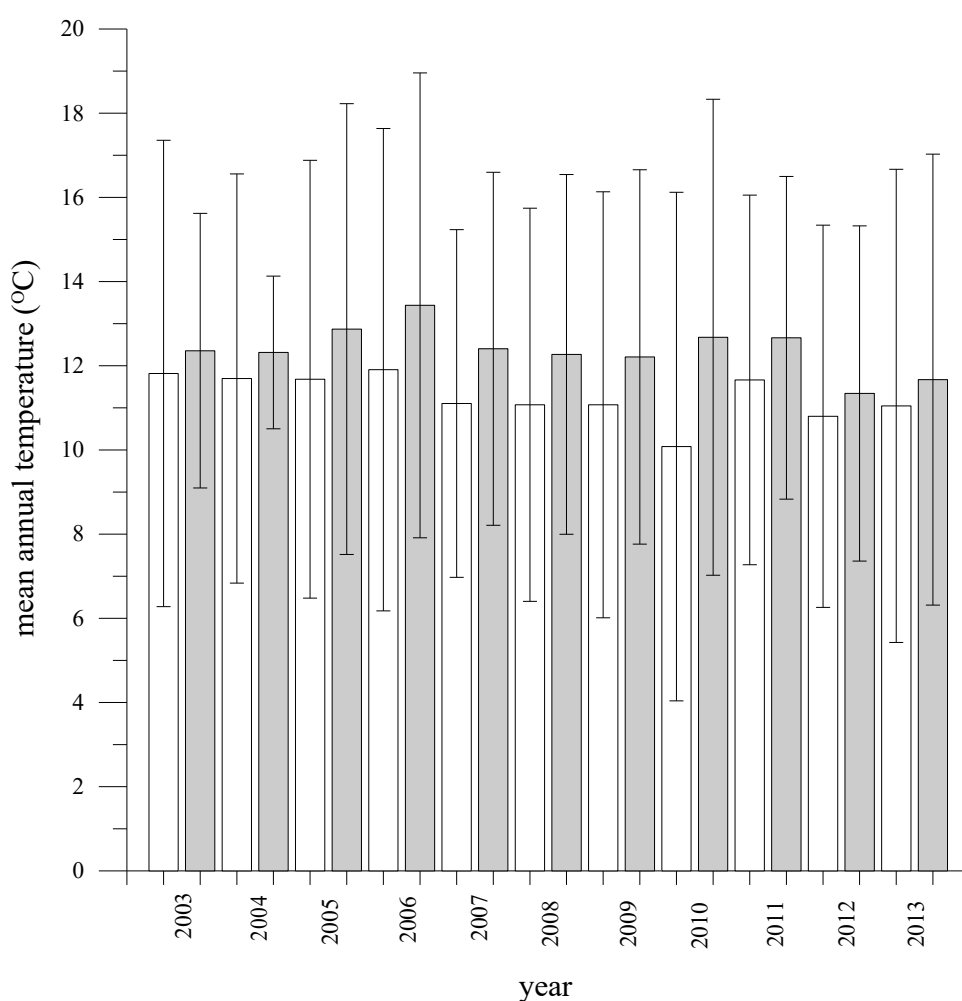


Figure 3.7: Mean annual air temperature (white bars) and mean annual whole-Bay surface water temperature (grey bars) in Cardiff Bay from 2003 until 2013 (standard error bars).

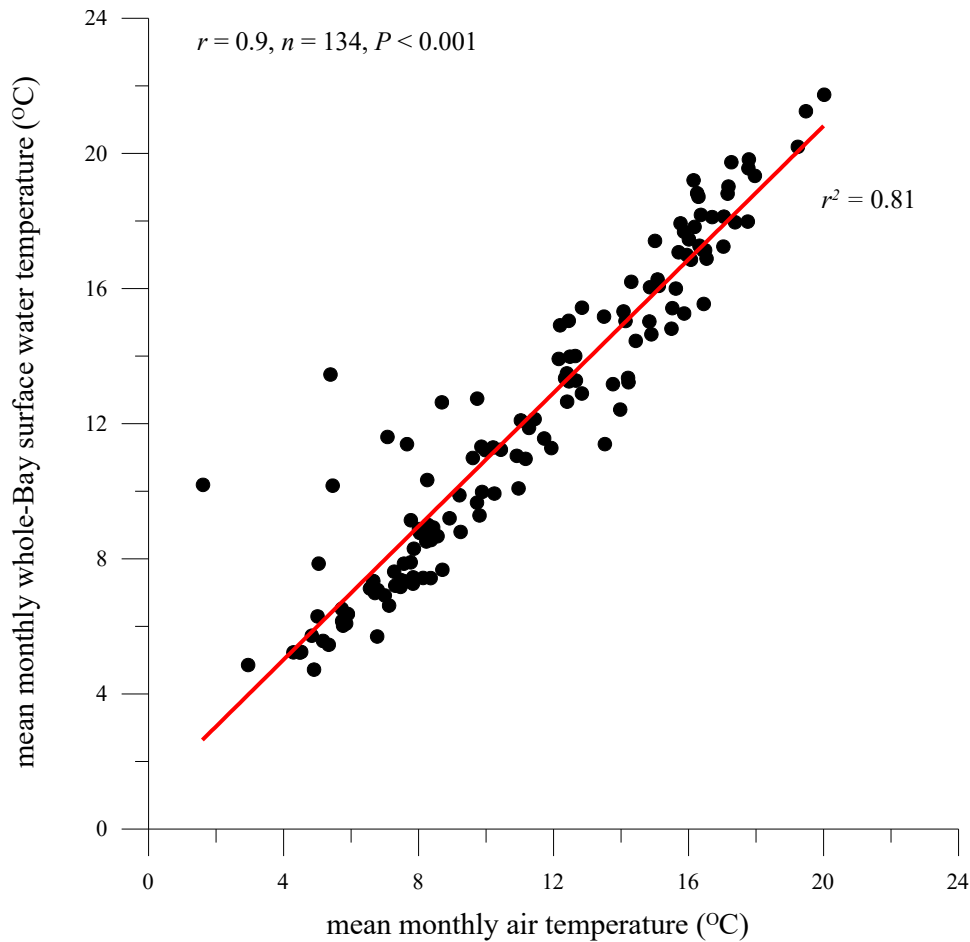


Figure 3.8: Mean monthly air temperature and mean monthly whole-Bay surface water temperature data taken in Cardiff Bay between January 2003 and April 2014.

Mean whole-Bay surface water temperature showed a strong seasonal pattern (Figure 3.9, seasonal), where seasons were significantly different ($F_{33,131} = 9.08$, $P < 0.01$). Surface water temperatures were at their lowest during winter; the lowest seasonal mean whole-Bay surface water temperature of 5.5 ± 0.22 °C was experienced during winter 2009. Summer was the warmest season with the highest seasonal mean whole-Bay surface water temperature experienced during summer 2003 of 19.5 ± 0.13 °C. Autumn was the next warmest season with whole-Bay surface water temperature seasonal means ranging from 11.1 ± 0.4 °C in 2005 to 15.1 ± 0.29 °C during 2006. Spring experienced whole-Bay surface water temperature seasonal means ranging from 9.0 ± 0.26 °C in 2013 to 12.4 ± 0.43 °C in 2007, making it the second coolest season. However, seasonal mean whole-Bay surface water temperature only showed a

strong significant correlation with seasonal mean air temperature during winter and summer (Figure 3.10).

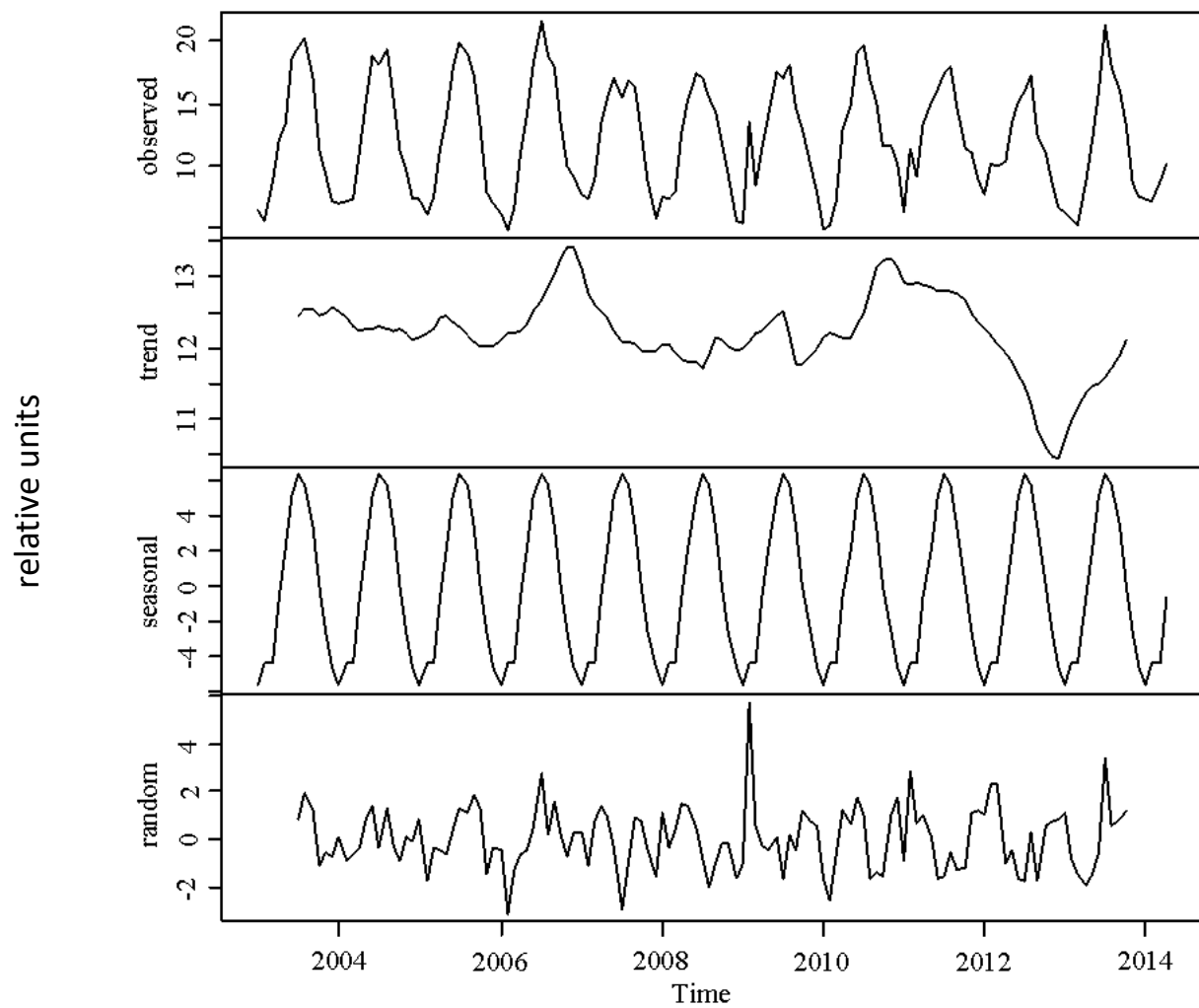


Figure 3.9: Time series analysis of water temperature in Cardiff Bay, using data from January 2003 until April 2014.

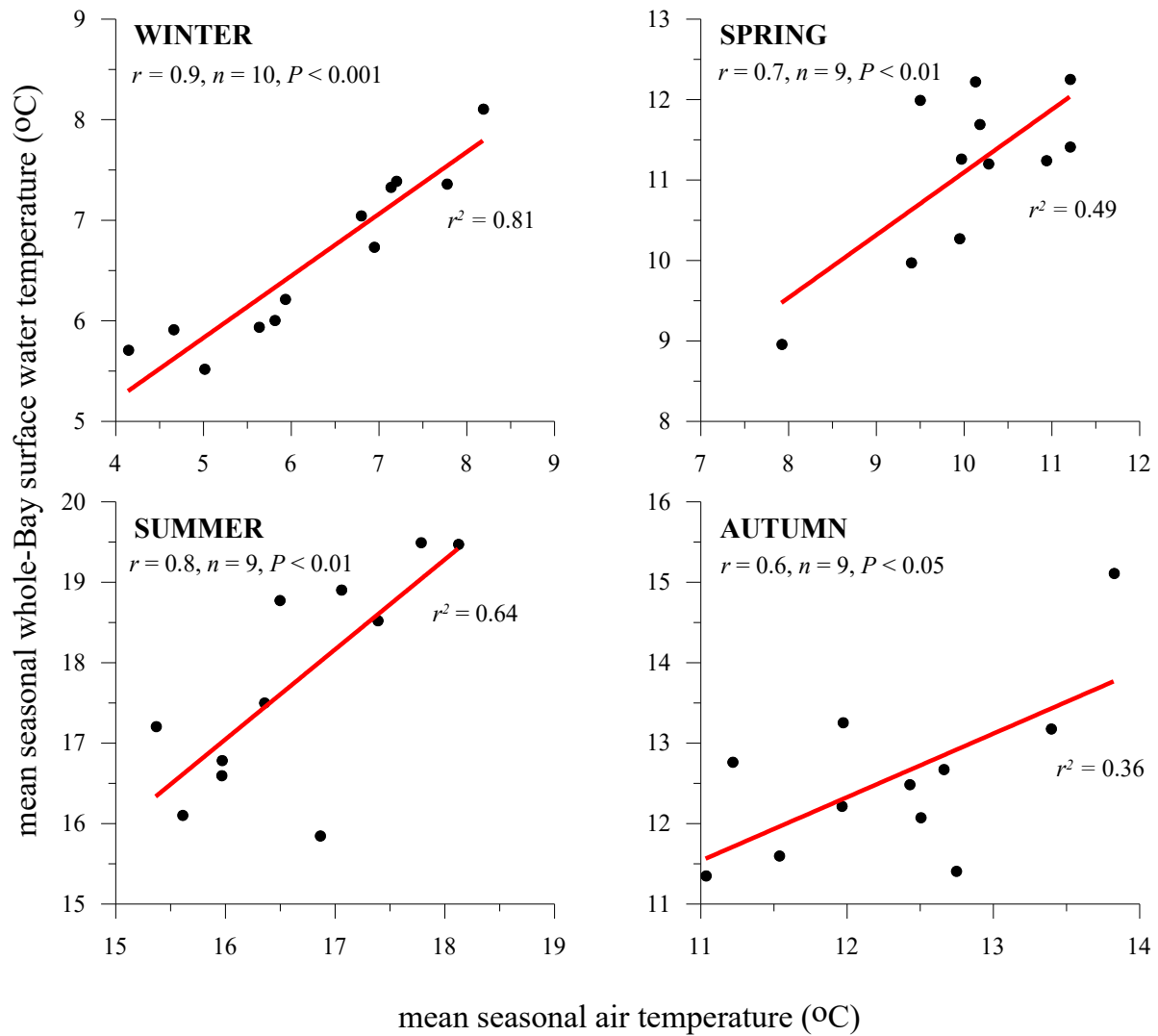


Figure 3.10: Mean seasonal air temperatures and mean seasonal whole-Bay surface water temperatures taken between winter 2003 and winter 2014 in Cardiff Bay.

Mean monthly surface water temperature closely followed the trend in mean monthly air temperature between January 2003 and April 2014 (Figure 3.11). Mean monthly whole-Bay surface water temperature in January, March, May, June, July, August, September and October all had a significant strong correlation with mean monthly air temperature (Figure 3.12). The three months that experienced the warmest water temperatures were July 2006 (21.74 ± 0.12 °C), July 2013 (21.25 ± 0.19 °C) and August 2003 (20.19 ± 0.15 °C). Air temperature explained 89% of the monthly variation in water temperature ($F_{1,134} = 1123, P < 0.01$); water temperature = $0.2651 + 0.9047 \cdot \text{air temperature}$; $r^2 = 0.89$.

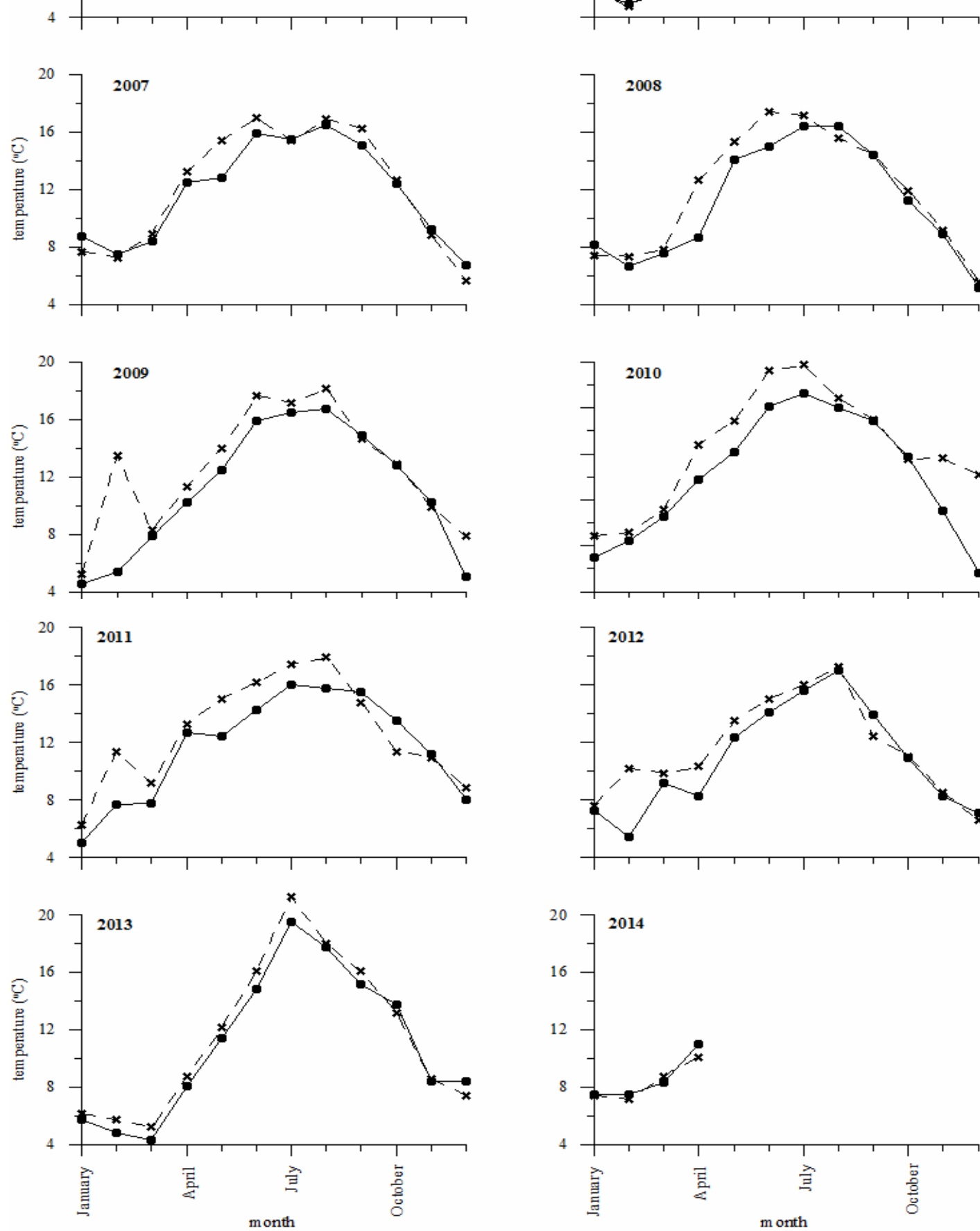


Figure 3.11: Mean monthly air temperature (solid line and black dots) and mean monthly whole-Bay surface water temperature (dashed line with crosses).

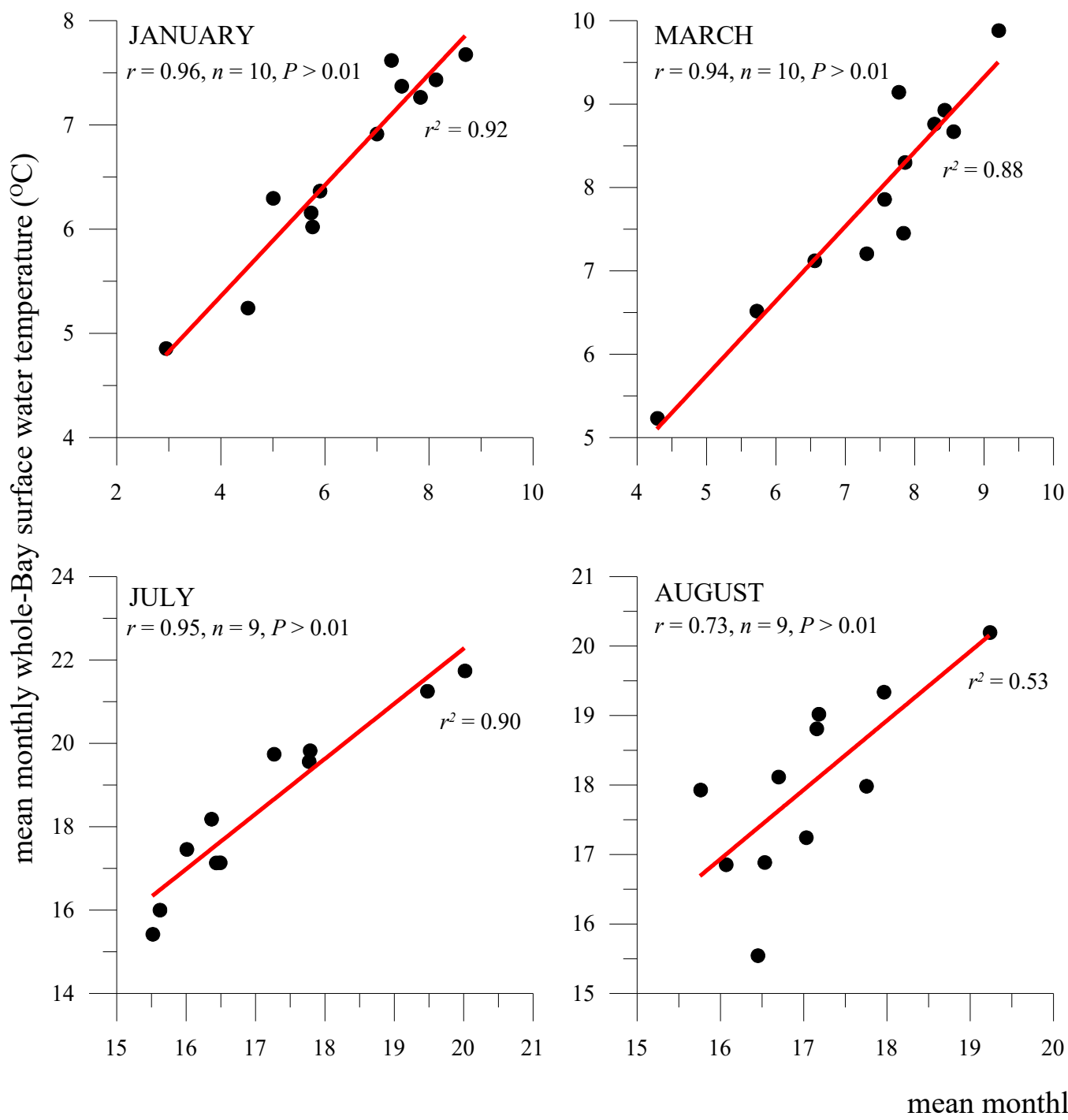


Figure 3.12: Mean monthly air temperature and mean monthly whole-Bay surface water temperature of Cardiff Bay

3.5.3 Temporal Variation of Precipitation in Cardiff Bay

There was no overall temporal trend observed in the magnitude of precipitation in Cardiff Bay between 2002 and 2014 (Figure 3.13, trend panel). However, the summer of 2009 experienced extremely high precipitation levels in comparison to any other season studied with a total precipitation of 1,651 mm, the majority of which fell during July (1,043 mm). There was a significant difference between seasonal precipitation ($F_{33,131} = 2.45$, $P < 0.01$), displayed in Figure 3.13 (seasonal panel). The highest levels of precipitation between January 2003 and April 2014 occurred during the seasons of autumn and winter, with seasonal means of 239 ± 34 mm and 204 ± 24 mm respectively. Spring had the lowest seasonal mean precipitation level of 150 ± 14 mm with summer being the second driest season with a seasonal precipitation mean of 196 ± 14 mm (excluding the extreme precipitation event during the summer of 2009), (Figure 3.14). There was no annual or seasonal statistical relationship between precipitation and mean surface water temperature or winter NAO index.

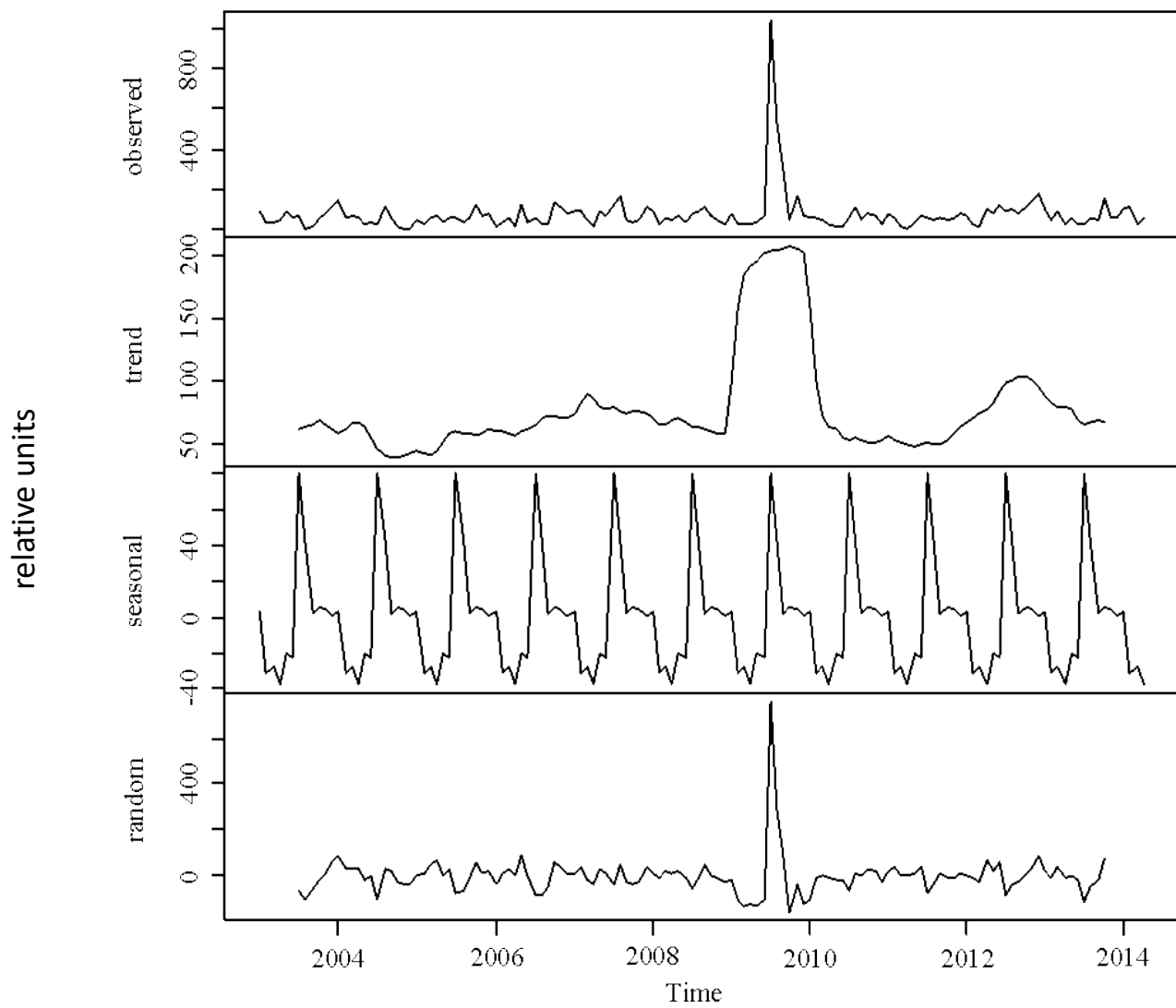


Figure 3.13: Time series analysis of precipitation in Cardiff Bay, using data from January 2003 until April 2014.

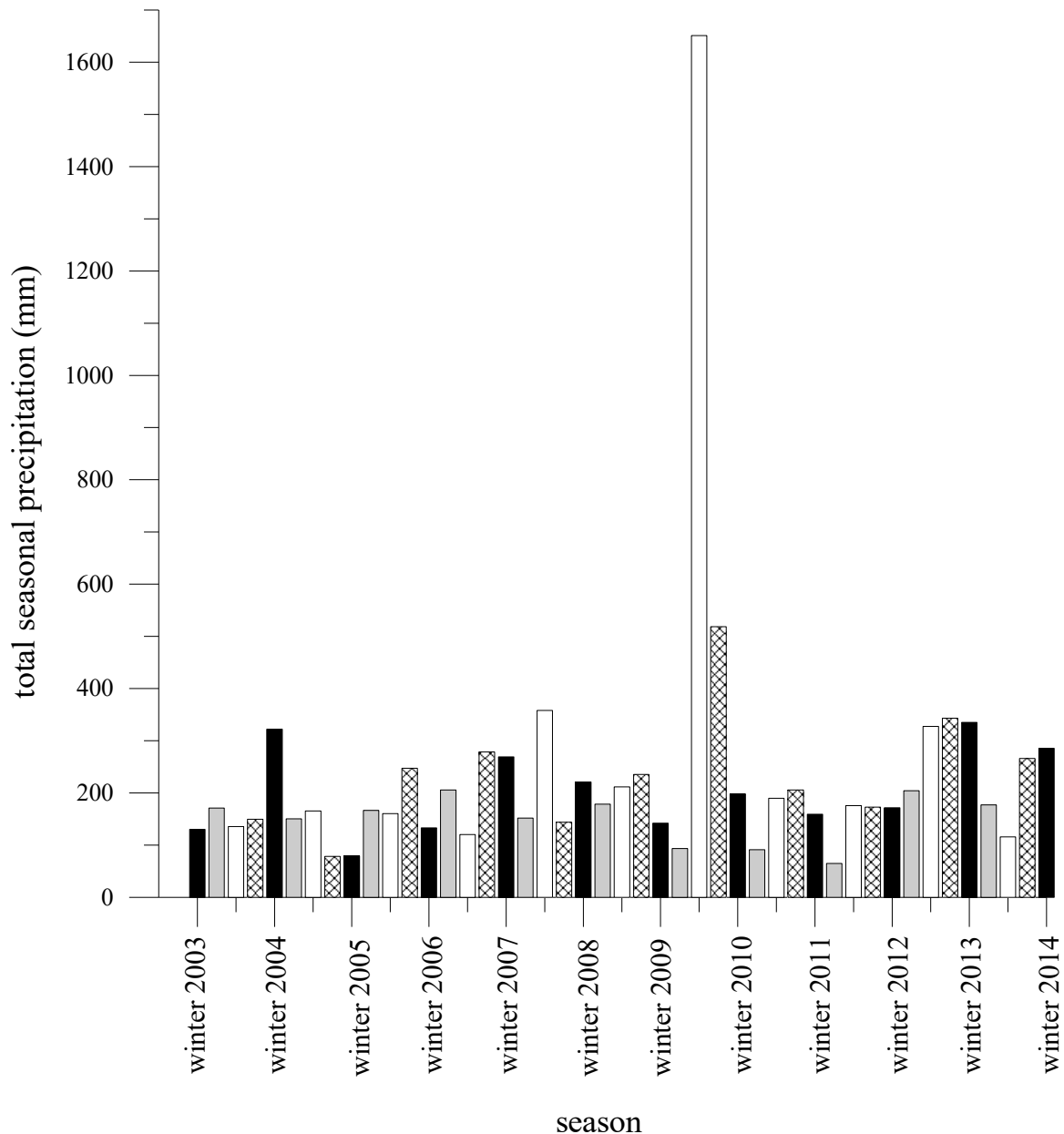


Figure 3.14: Seasonal precipitation totals in Cardiff Bay from winter 2003 until winter 2014. Winter: black bars. Spring: grey bars. Summer: white bars. Autumn: hatched bars.

3.5.4 The Impact of Precipitation on River Taff and River Ely Discharge

The River Taff contributes to between 77.2% (spring 2011) and 85.6% (winter 2007) of the river discharge entering Cardiff Bay, and the River Ely contributes between 14.4% (winter 2007) and 22.8% (spring 2011) of the river discharge entering the lake (Figure 3.15). This

resulted in the mean monthly combined river discharge rate (Figure 3.16) closely following the mean monthly River Taff discharge rate.

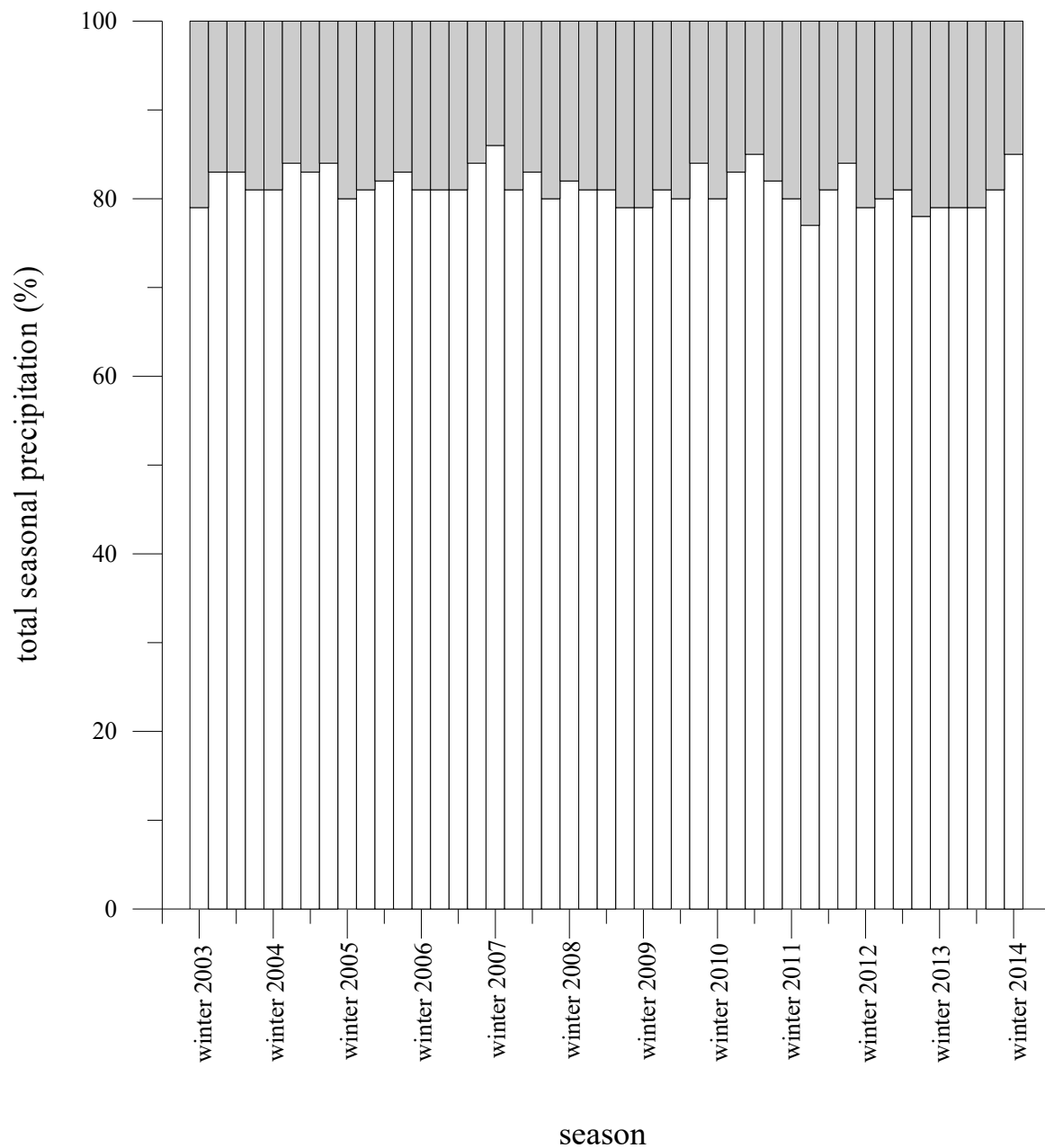


Figure 3.15: Percentage contribution of river discharge from the River Taff and River Ely, from winter 2003 until winter 2014.

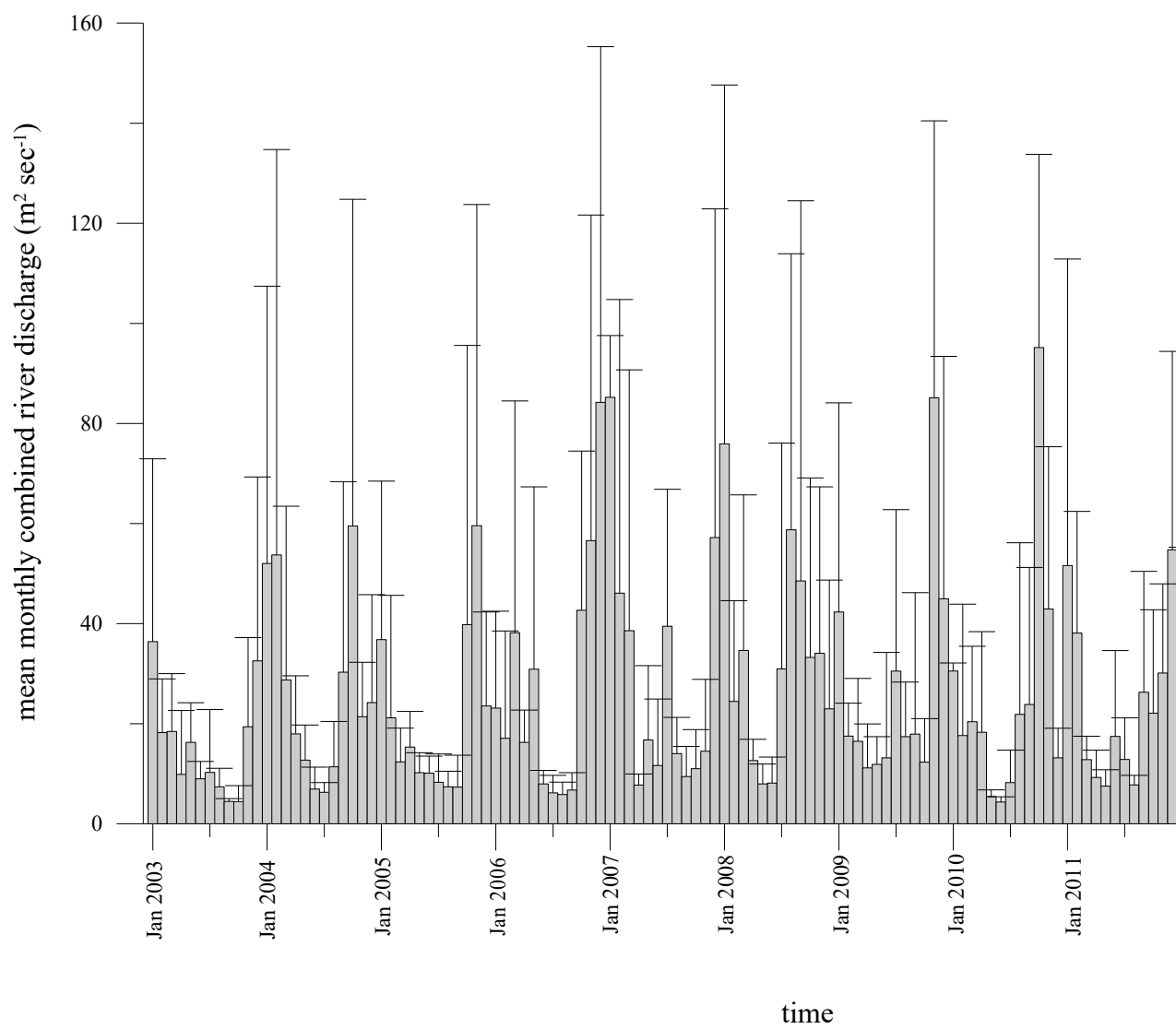


Figure 3.16: Mean monthly combined river discharge rates from River Taff and River Ely, data from January 2003 un

Although an increase in combined river discharge is visible in Figure 3.17 (trend panel), statistically there was no difference in combined river discharge between years. However, there was a seasonal trend in combined river discharge rates (Figure 3.17, seasonal panel) with a significant difference between seasons (Taff: $F_{3,135} = 17.1$, $P < 0.01$. Ely: $F_{3,135} = 22$, $P < 0.01$. Combined: $F_{3,135} = 18.3$, $P < 0.01$). Combined River discharge rates (Figure 3.18) from the River Taff (Figure 3.19) and River Ely (Figure 3.20) were lower during the spring (combined mean of $24.3 \pm 25 \text{ m}^3 \text{ sec}^{-1}$) and summer (combined mean of $22 \pm 21 \text{ m}^3 \text{ sec}^{-1}$) than during autumn (combined mean of $30 \pm 10.3 \text{ m}^3 \text{ sec}^{-1}$) and winter (combined mean of $45.8 \pm 17.5 \text{ m}^3 \text{ sec}^{-1}$). The mean river discharge rates were significantly different between months for the River Taff ($F_{11,135} = 6.85$, $P < 0.01$), River Ely ($F_{11,135} = 9.25$, $P < 0.01$) and combined river discharge ($F_{11,135} = 7.40$, $P < 0.01$). The months of November and December experienced significantly higher river discharge rates than April, May, June, July, August and September (Figures 3.21, 3.22 and 3.23).

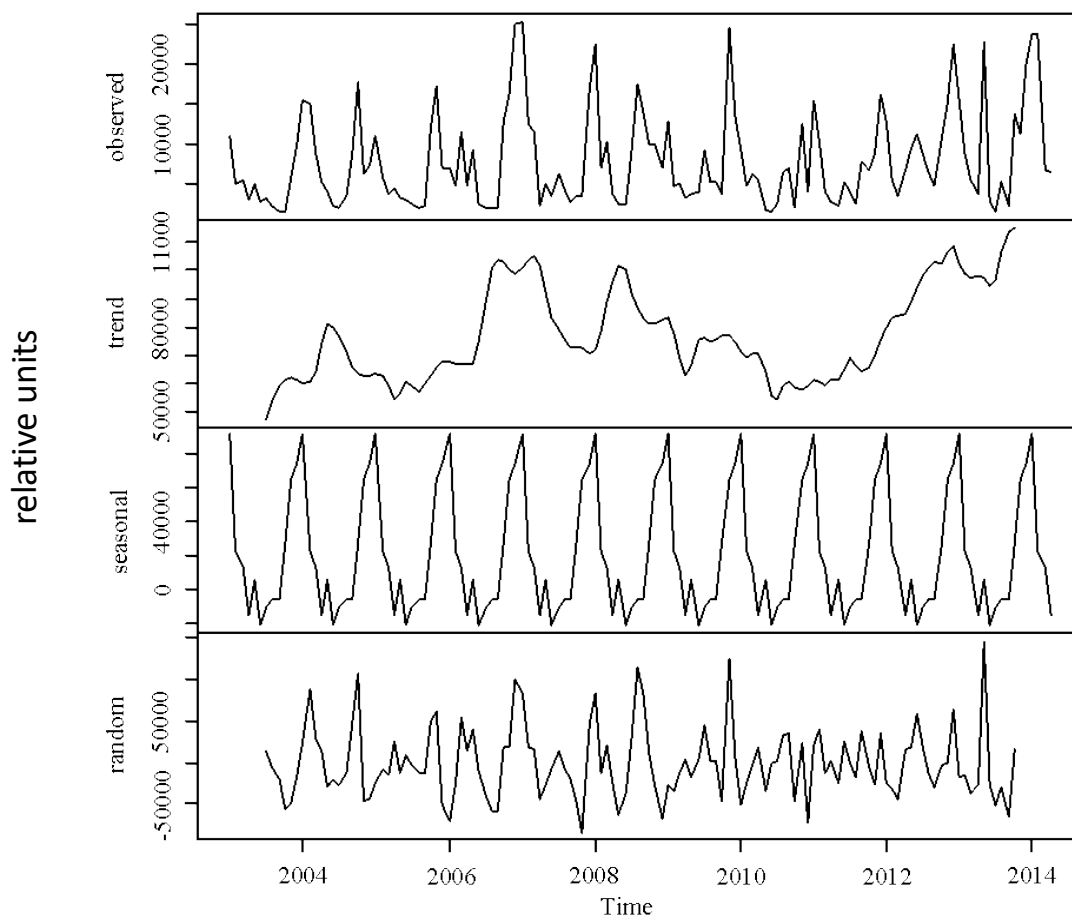
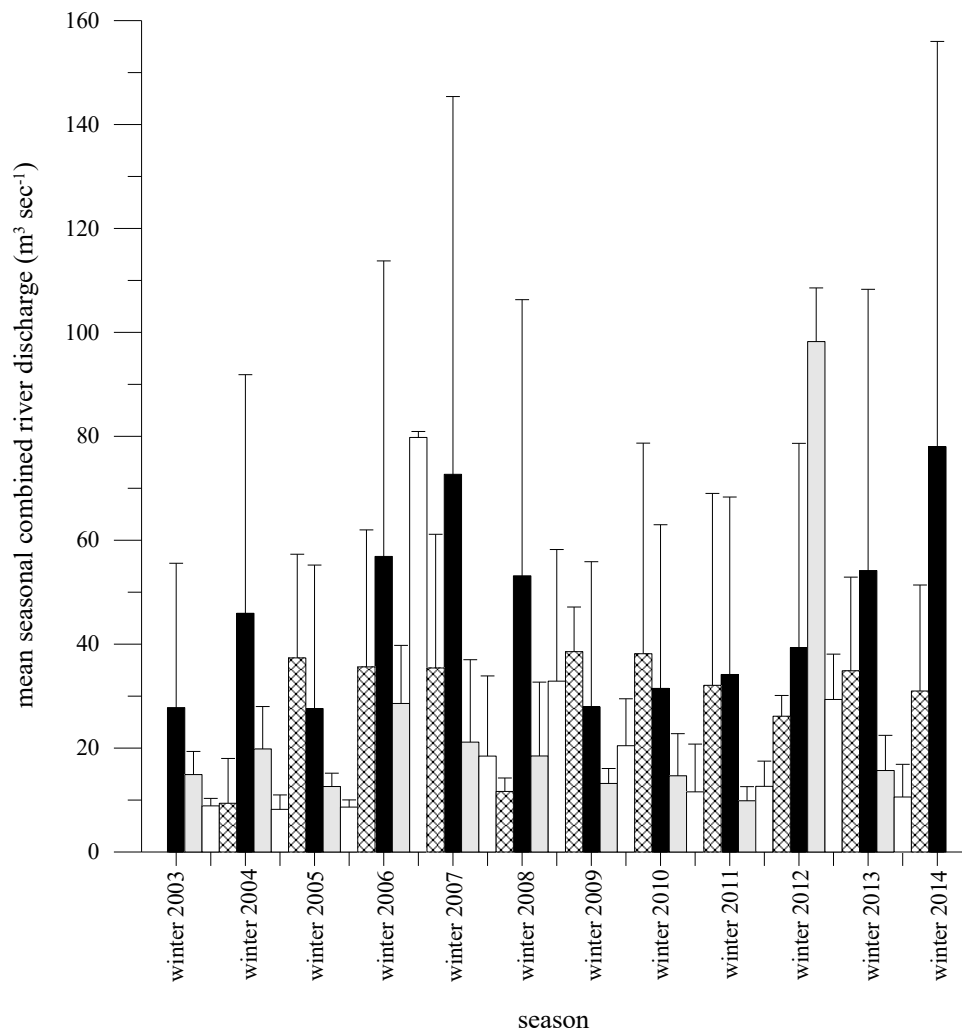


Figure 3.17: Time series analysis of combined river discharge from the River Taff and River Ely, which supply Cardiff Bay. Data recorded from January 2003 until April 2014.



3.18: Mean seasonal combined River Taff and River Ely discharge rates from January 2003 until April 2014. winter: black bars, spring: grey bars, summer: white bars, autumn: hatched bars (standard error bars).

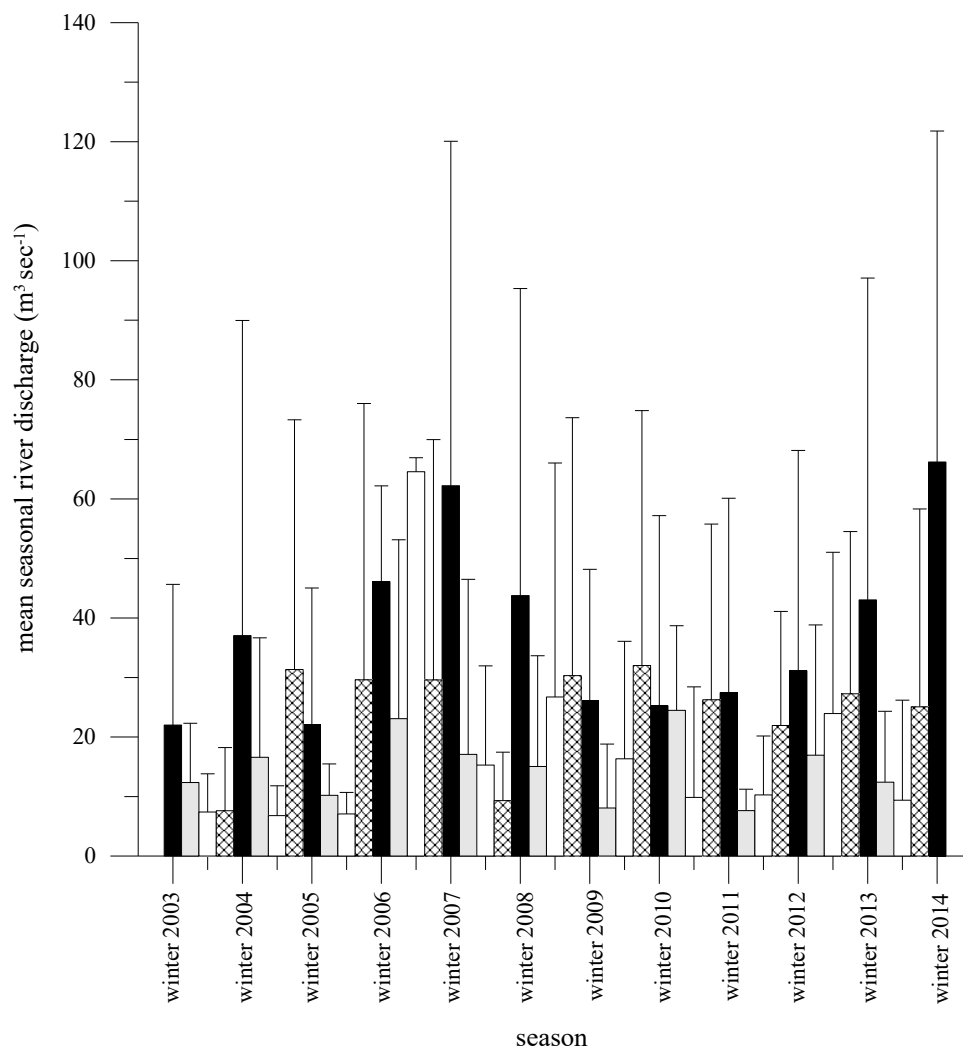


Figure 3.19: Mean seasonal River Taff discharge rates from January 2003 until April 2014. winter: black bars, spring: grey bars, summer: white bars, autumn: hatched bars (standard error bars).

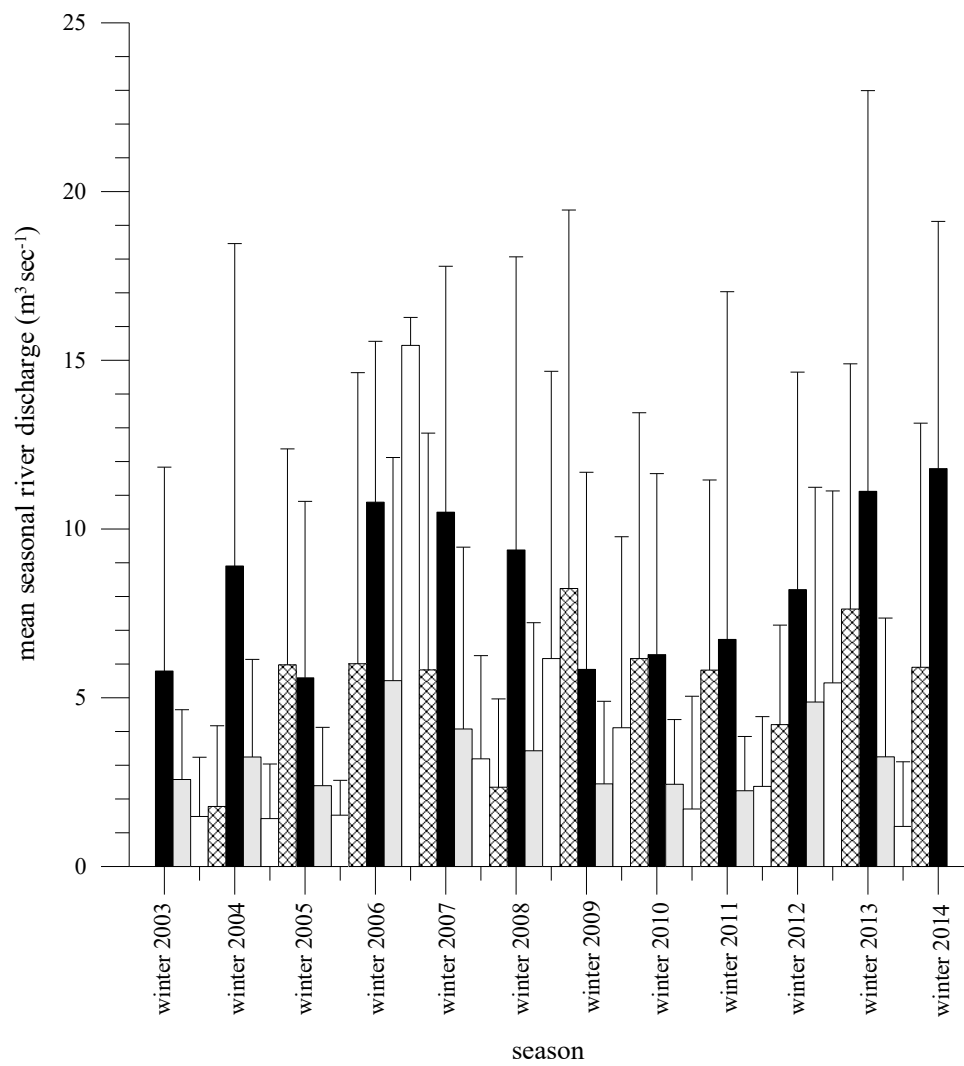


Figure 3.20: Mean seasonal River Ely discharge rates from January 2003 until April 2014. winter: black bars, spring: grey bars, summer: white bars, autumn: hatched bars (standard error bars).

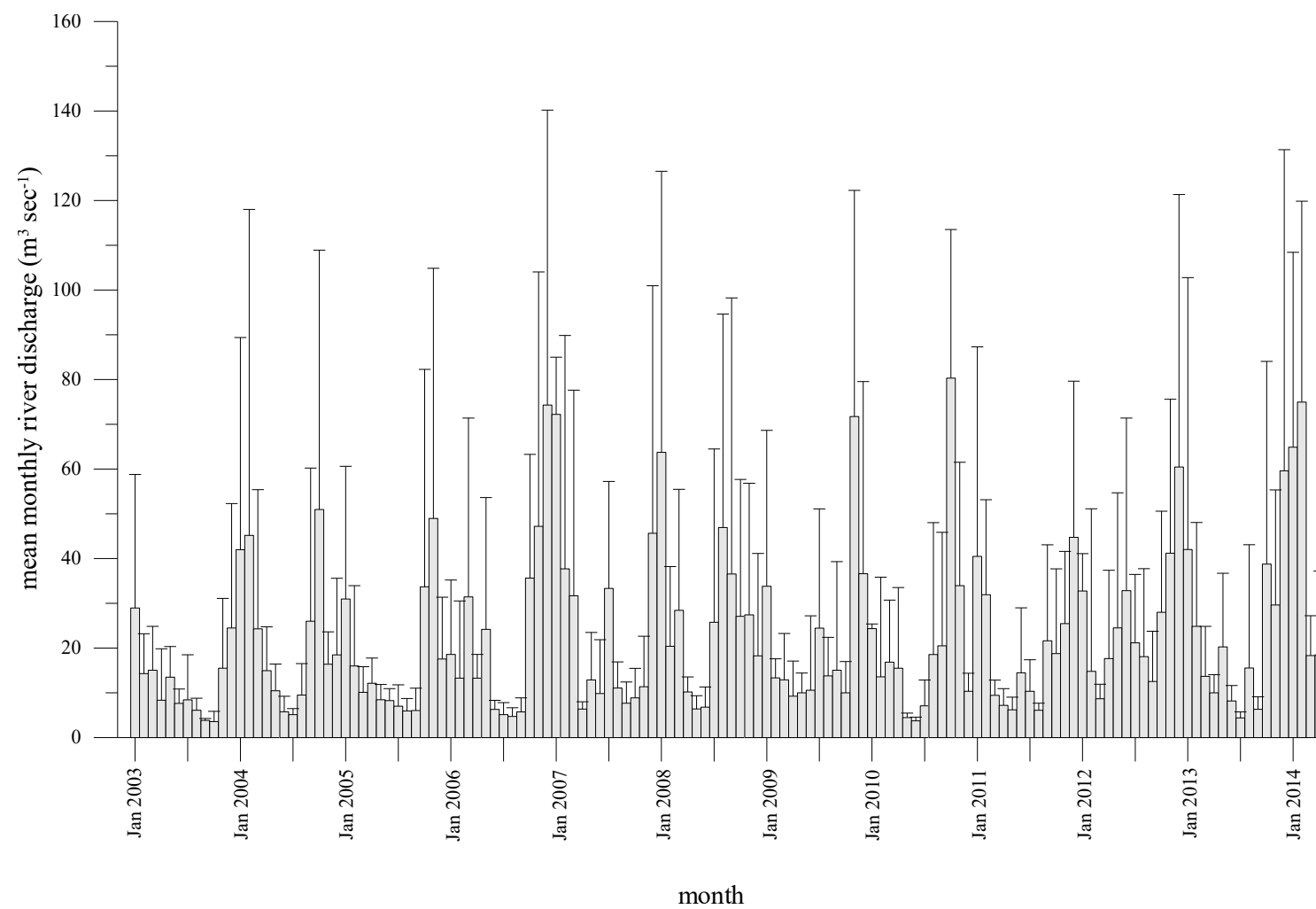
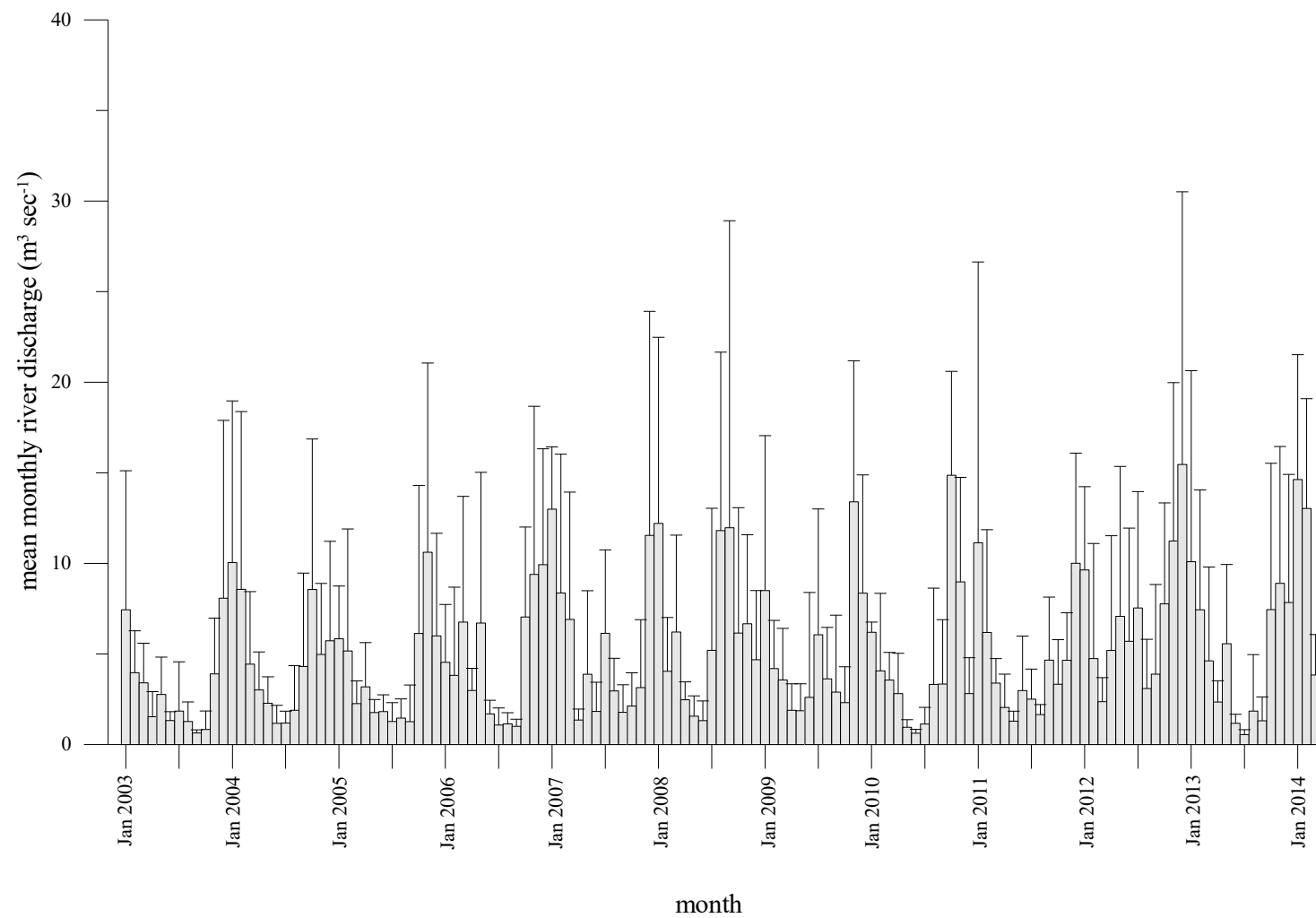


Figure 3.21: Mean monthly River Taff discharge from January 2003 until April 2014 (standard error bars).



3.22: Mean monthly River Ely discharge from January 2003 until April 2014 (standard error bars).

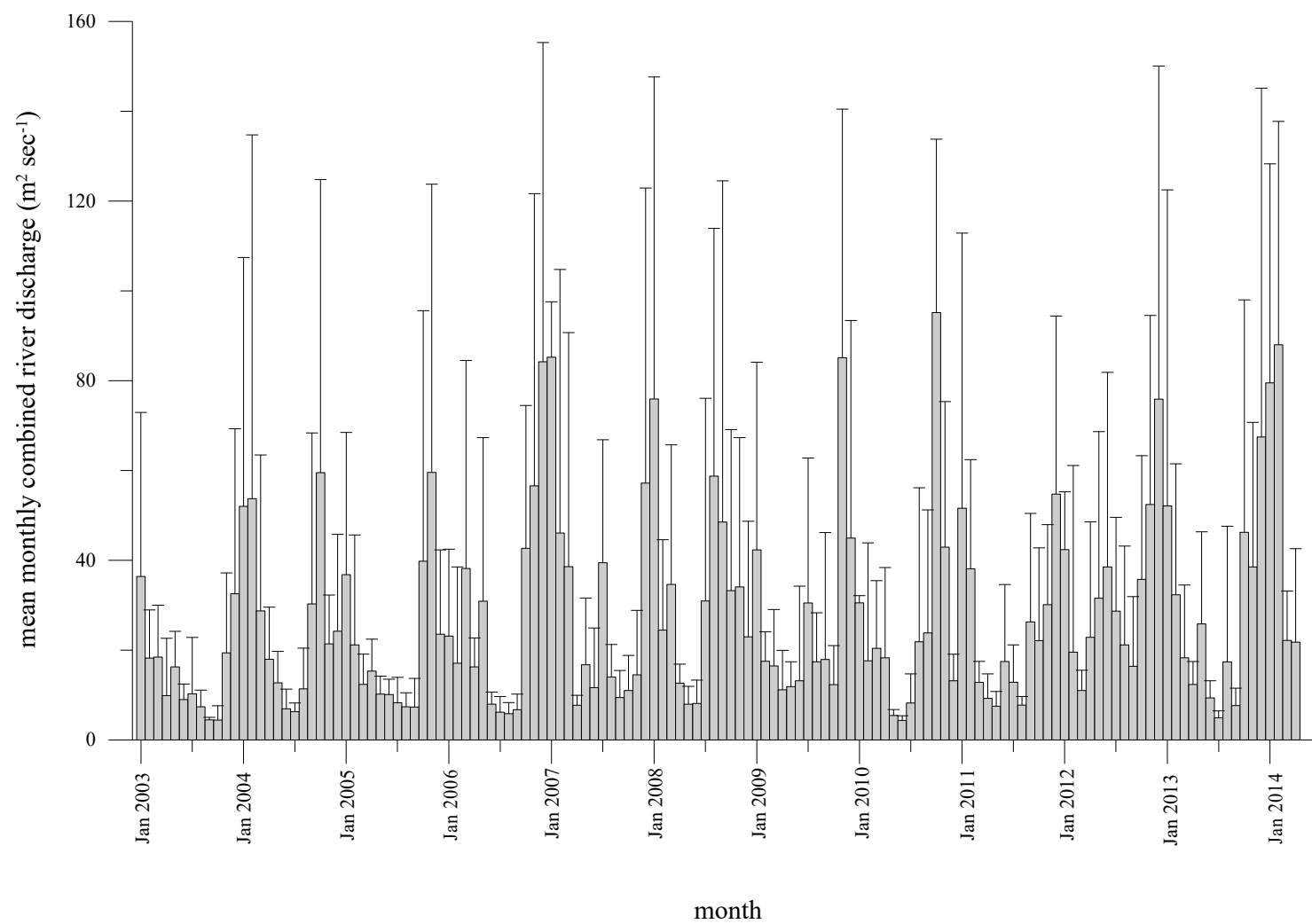


Figure 3.23: Mean monthly combined river discharge from the River Taff and River Ely, data from January 2003 until April 2014 (standard error bars).

Mean annual precipitation correlated weakly with mean annual River Taff discharge ($r = 0.7$, $n = 9$, $P < 0.02$) and not at all with mean annual River Ely discharge or mean annual combined river discharge. During winter, mean seasonal river discharge did not correlate strongly with mean seasonal precipitation (Taff: $r = 0.6$, $n = 10$, $P < 0.05$. Ely: $r = 0.7$, $n = 10$, $P < 0.05$. Combined: $r = 0.7$, $n = 10$, $P < 0.05$). Mean seasonal river discharge and mean seasonal precipitation did not correlate during any other season. On a monthly basis mean precipitation only weakly correlated positively with mean monthly River Taff discharge ($r = 0.2$, $n = 130$, $P < 0.05$), mean monthly River Ely discharge ($r = 0.2$, $n = 130$, $P < 0.05$) and mean monthly combined river discharge ($r = 0.2$, $n = 130$, $P < 0.05$).

As an example of potential climate change risk due to extreme summer rainfall events, the effect of the high peak July precipitation in 2009 on river discharge rate was investigated. A notable increase in river discharge can be observed relative to the adjacent months (Figure 3.23), however the river discharge experienced during July 2009 was not atypical compared to the range of data observed between 2003 and 2014.

3.5.5 The Flushing Rate of Cardiff Bay

There was no statistically significant variation in the mean annual flushing rate of Cardiff Bay between 2003 and 2013 (Figure 3.24). Between January 2003 and April 2014, lake flushing rate demonstrated a seasonal fluctuation (Figure 3.25, seasonal panel), where the seasonal lake flushing rate was statistically different between seasons ($F_{3,44} = 17.3$, $P < 0.01$). Summer experienced the slowest flushing rate with a mean seasonal value of $6.7 \pm 0.9 \text{ days}^{-1}$, followed by spring with a mean seasonal flushing rate of $1.5 \pm 0.4 \text{ days}^{-1}$. Winter had the fastest mean seasonal flushing rate at $0.8 \pm 0.2 \text{ days}^{-1}$ and was significantly faster than spring and summer. Autumn had a mean seasonal flushing rate of $3.3 \pm 0.6 \text{ days}^{-1}$ which was significantly slower than winter but faster than summer (Figure 3.26).

During autumn, mean seasonal lake flushing rate significantly correlated with mean seasonal combined, Taff and Ely river discharge (Figure 3.27). During winter, spring and summer, mean seasonal lake flushing rate and mean seasonal river discharge showed weak correlations

(Appendix Figures 3.1, 3.2 and 3.3). During spring, mean seasonal lake flushing rate correlated with mean seasonal precipitation (Figure 3.28). Mean seasonal lake flushing rate did not correlate with mean seasonal precipitation during any other seasons.

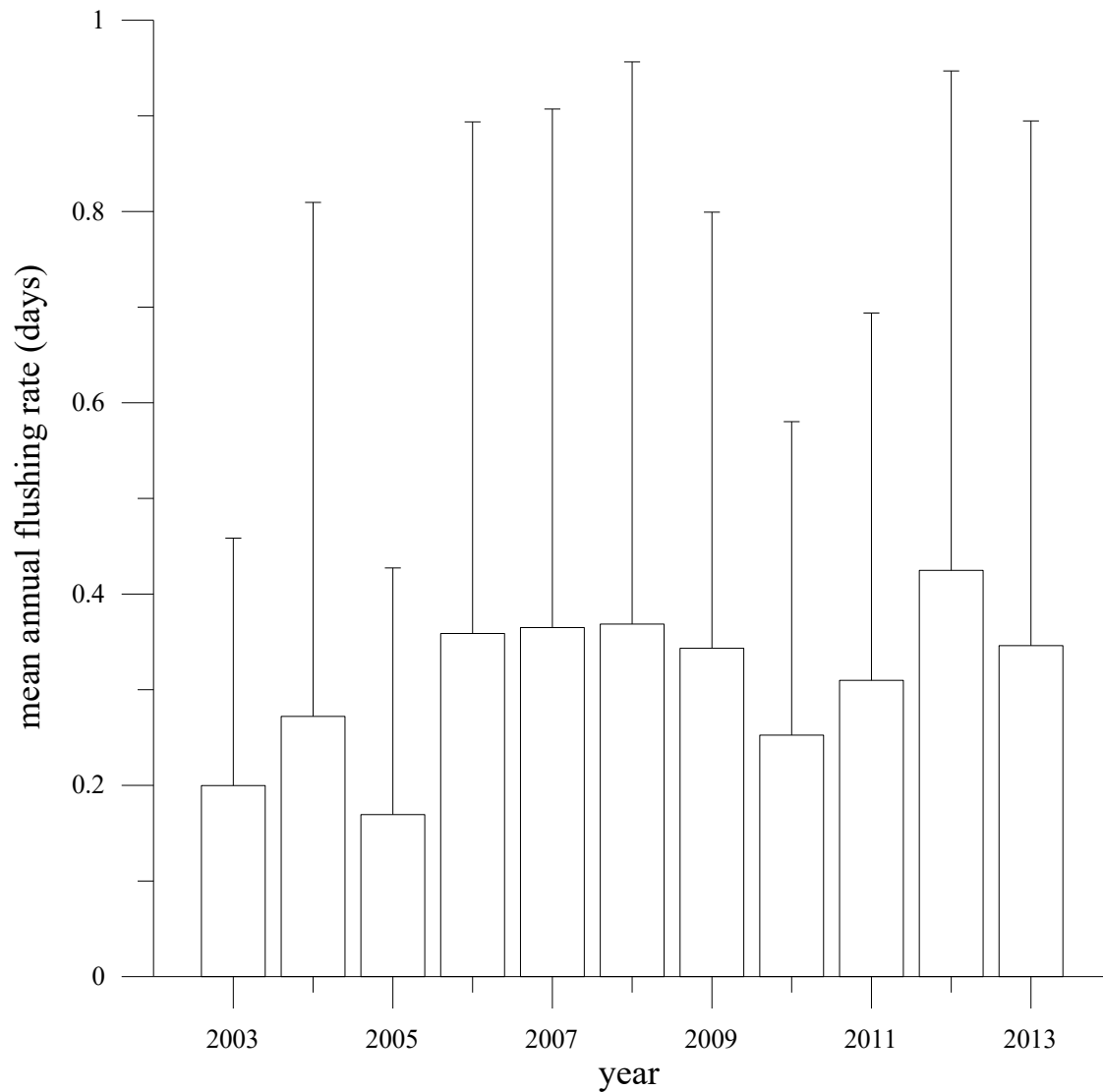


Figure 3.24: Mean annual lake flushing rate of Cardiff Bay, from 2003 until 2013 (standard error bars).

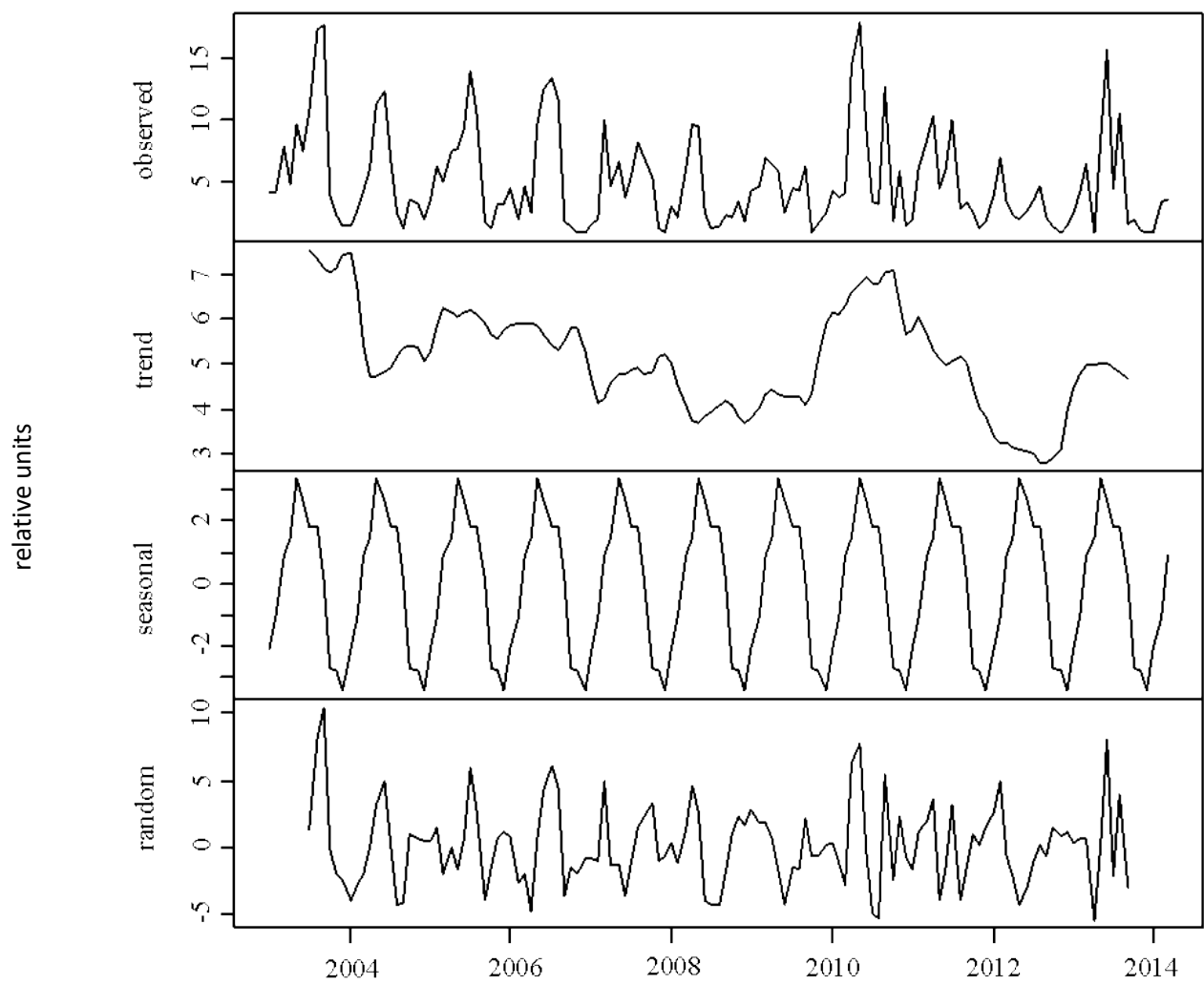


Figure 3.25: Time series analysis of the lake flushing rate of Cardiff Bay. Data recorded from January 2003 until April 2014.

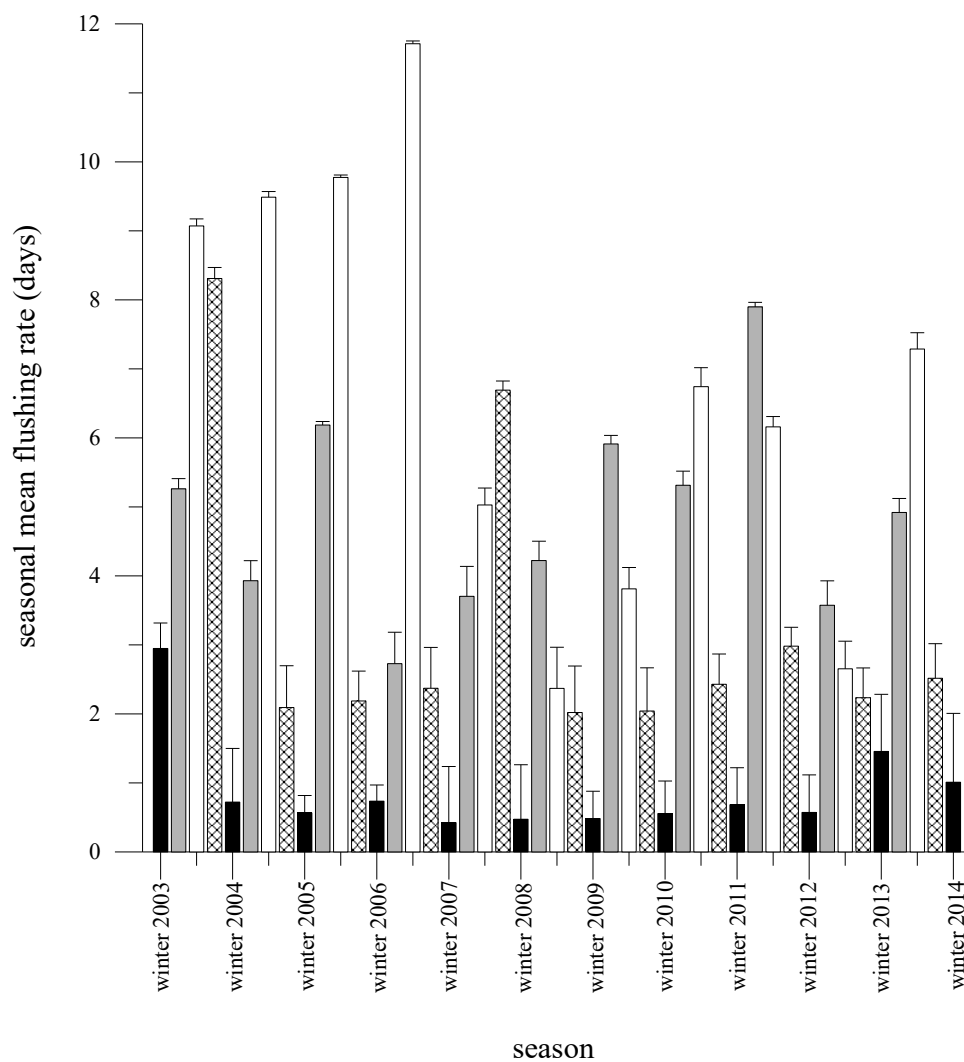


Figure 3.26: Mean seasonal lake flushing rate of Cardiff Bay from January 2003 until April 2014. winter: black bars, spring: grey bars, summer: white bars, autumn: hatched bars (standard error bars).

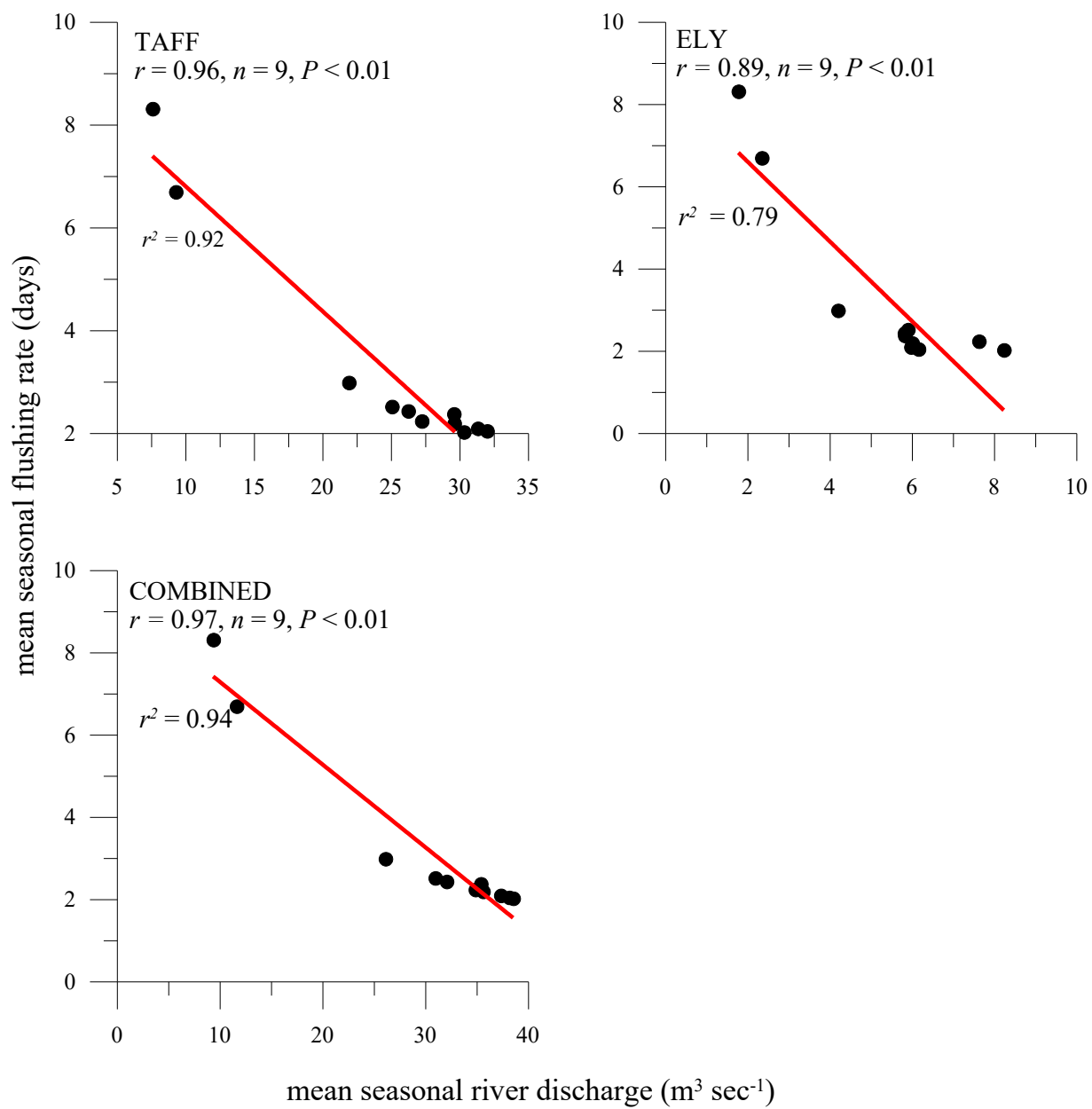


Figure 3.27: Mean autumn River Taff, River Ely and combined river discharge, with mean autumn lake flushing rate. Data from 2003 until 2013.

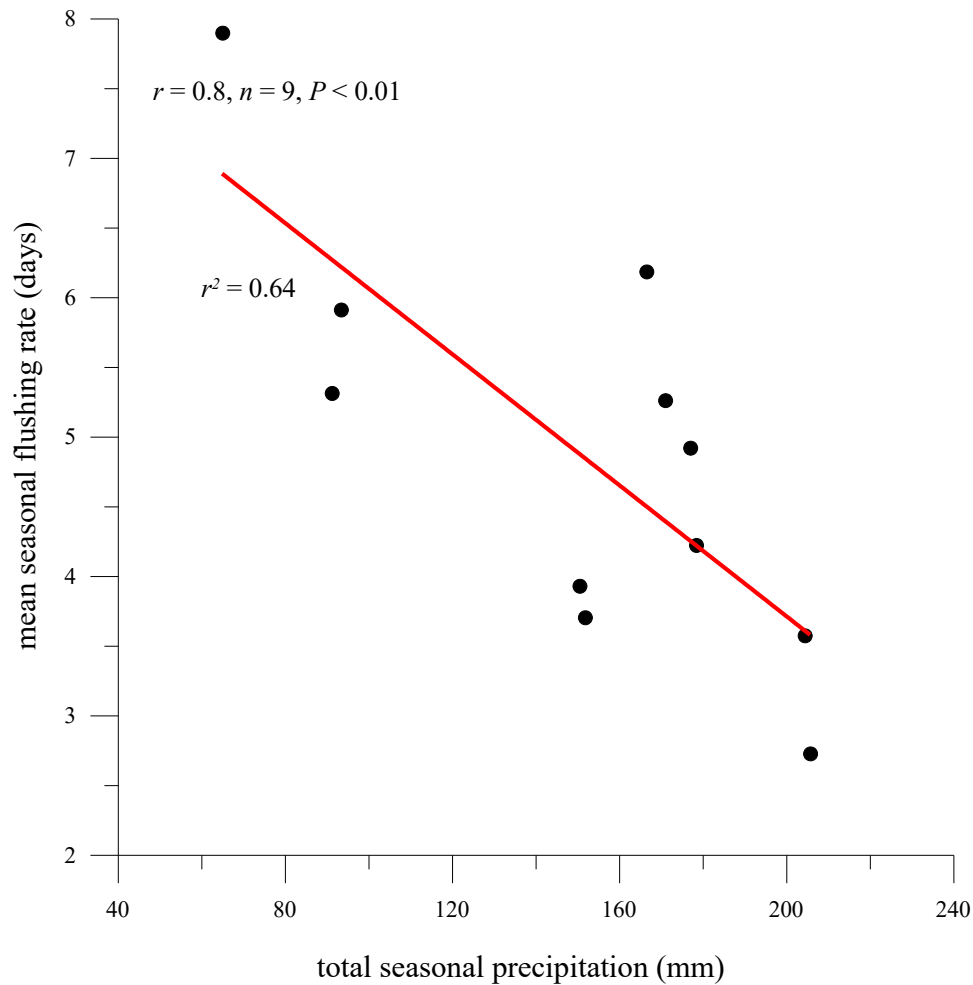


Figure 3.28: Total spring precipitation and mean spring lake flushing rate of Cardiff Bay. Data recorded between 2003 and 2013.

There were significant differences between months of mean monthly lake flushing rate ($F_{11,135} = 8.67, P < 0.01$). The flushing rate in January was faster than that in April, May, June, July, August and September. It was also faster in February than during the months of May to September. During March mean monthly flushing rate was faster than rates in June. The months of November and December experienced faster flushing rates than the months of May, June, July, August and September (Figure 3.29).

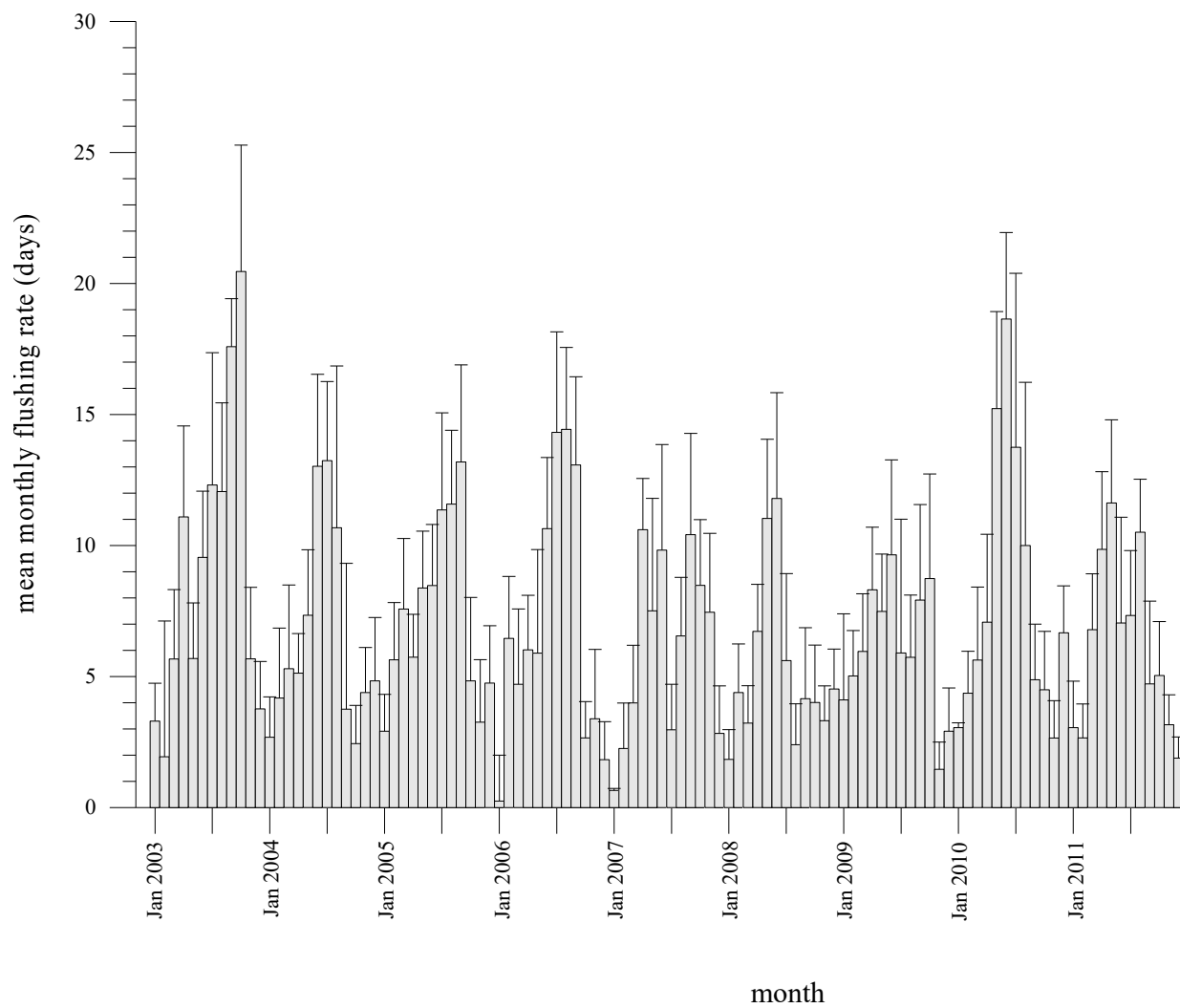


Figure 3.29: Mean monthly flushing rate of Cardiff Bay from January 2003 until April 2014 (stan

The mean monthly lake flushing rate of Cardiff Bay was significantly correlated with mean monthly River Taff discharge ($r = 0.71$, $n = 134$, $P < 0.01$), River Ely discharge ($r = 0.76$, $n = 134$, $P < 0.01$) and mean monthly combined river discharge ($r = 0.72$, $n = 134$, $P < 0.01$) (Figure 3.30), where an increase in river discharge causes the lake flushing rate to quicken. Multiple regression was carried out to determine the extent of this relationship between monthly flushing rate and river discharge. Mean monthly River Ely discharge was the only river discharge to significantly influence the mean monthly lake flushing rate (Equation A).

Equation A: (*flushing rate*) = $28.1 + 13.6$ (*River Ely discharge*); $F_{1,134} = 184.9$, $P < 0.01$, $r^2 = 0.58$.

There was no significant relationship between mean monthly flushing rate and mean monthly precipitation.

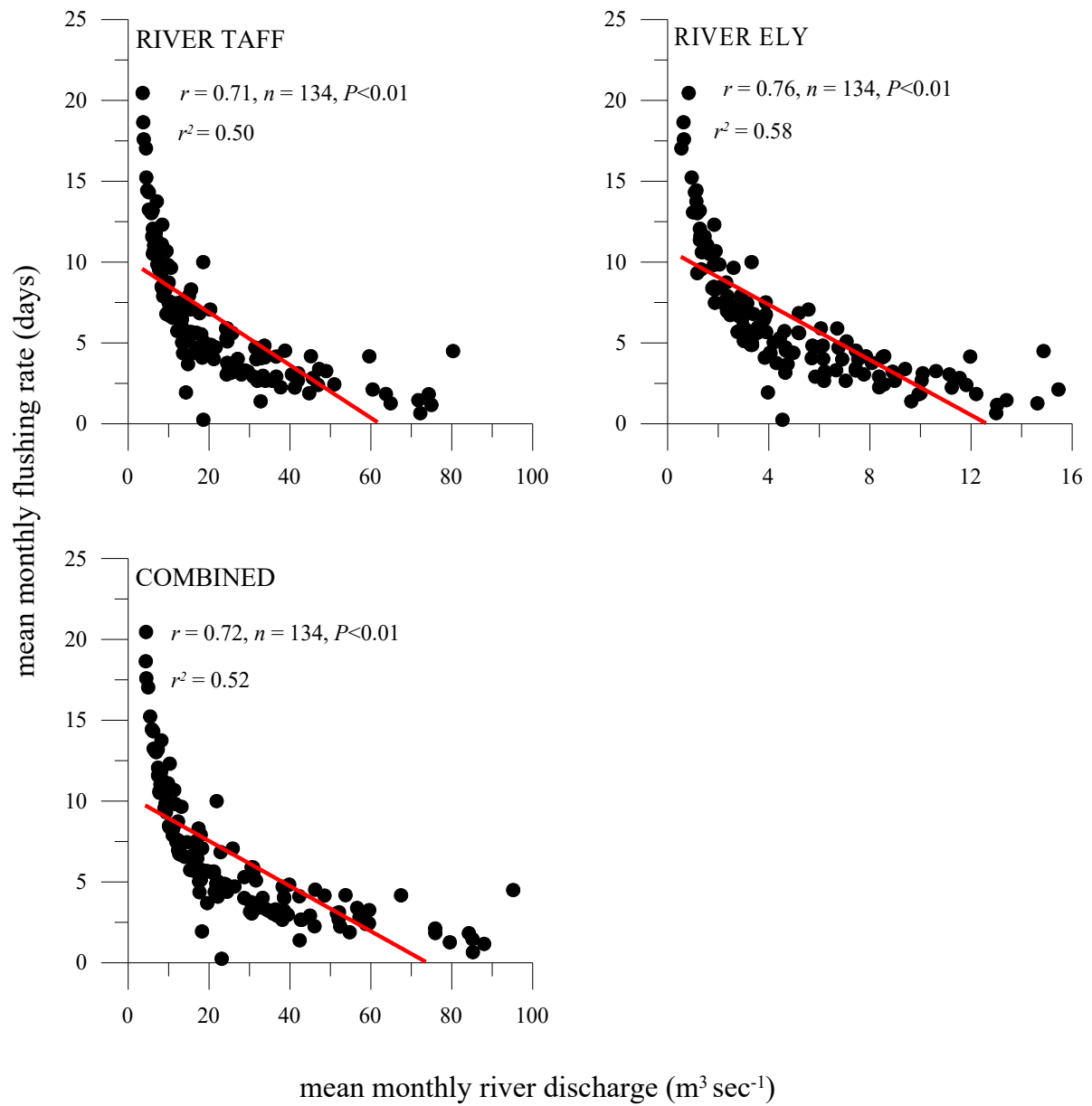


Figure 3.30: Mean monthly River Taff, River Ely and combined river discharge rates, and mean monthly flushing rate of Cardiff Bay. Data from January 2003 until April 2014.

3.5.6 Spatial & Vertical Variation of Water Temperature in Cardiff Bay

There was no statistical difference in mean surface water temperature between sites 4, 5, 6, 9, 10 and 17 on an annual, seasonal or monthly timescale between January 2003 and April 2014.

There was no evidence of thermal stratification within the lake as mean monthly water temperature did not vary with depth at sites 4, 5, 9, 10, A, B, C or D between April 2012 and April 2014.

3.5.7 The Dissolved Oxygen Levels of Cardiff Bay

Mean annual whole-Bay surface (1m depth) dissolved oxygen (DO) concentrations did not vary significantly between years from 2003 to 2013 in Cardiff Bay (Figure 3.31), however they did vary significantly between seasons ($F_{10,131} = 6.6$, $P < 0.01$), (Figure 3.32, seasonal). Statically, mean whole-Bay surface DO in summer was lower than any other season with seasonal means of between 8.5 ± 1.7 and 9.7 ± 1.3 mg l⁻¹. Mean whole-Bay autumn surface DO was also statistically lower than the concentrations experienced in winter and spring but higher than those in summer, with seasonal means of 9.5 ± 1.1 to 10.7 ± 1.6 mg l⁻¹. Winter and spring mean whole-Bay surface DO concentrations were not significantly different from each other, but they were significantly different from the concentrations found in summer and autumn. Winter had seasonal means of 10.9 ± 0.8 to 12.8 ± 1.4 mg l⁻¹ and spring had seasonal means of 10.5 ± 1.9 to 12.1 ± 2.2 mg l⁻¹ (Figure 3.33).

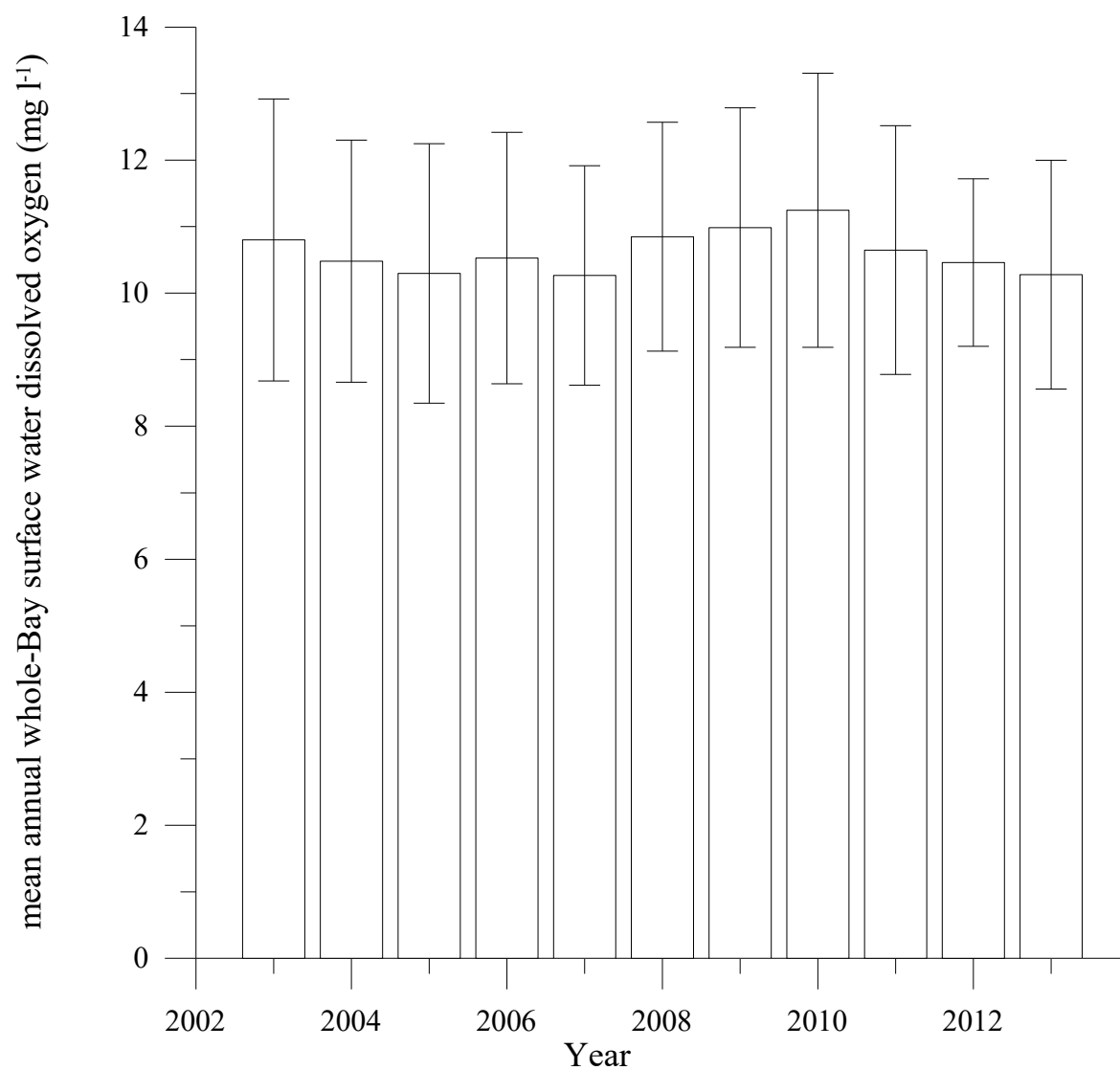


Figure 3.31: Mean annual whole-Bay surface dissolved oxygen concentrations in Cardiff Bay. Data from January 2003 until April 2014 (standard error bars).

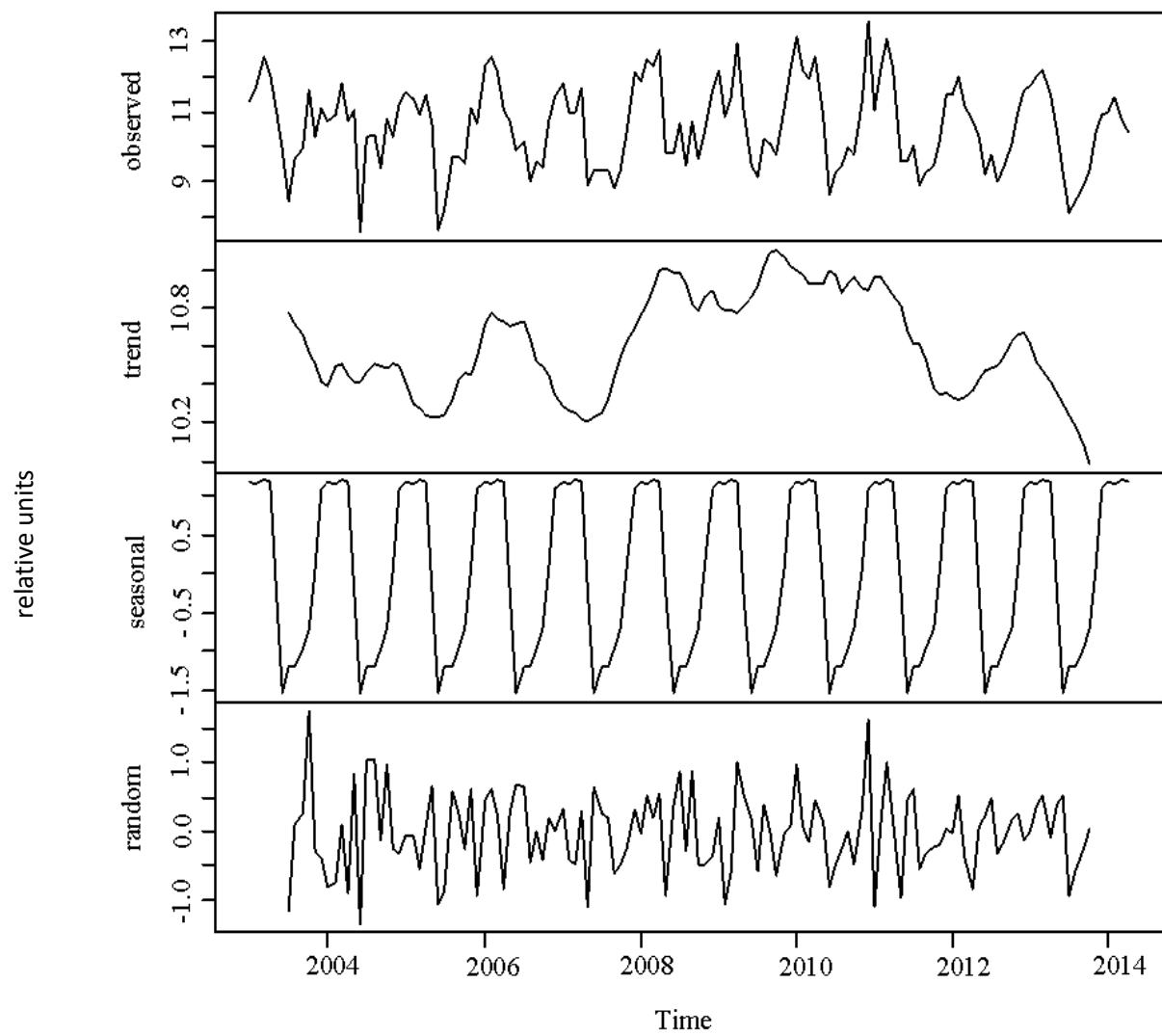


Figure 3.32: Time series analysis of the dissolved oxygen concentrations of Cardiff Bay. Data recorded from January 2003 until April 2014.

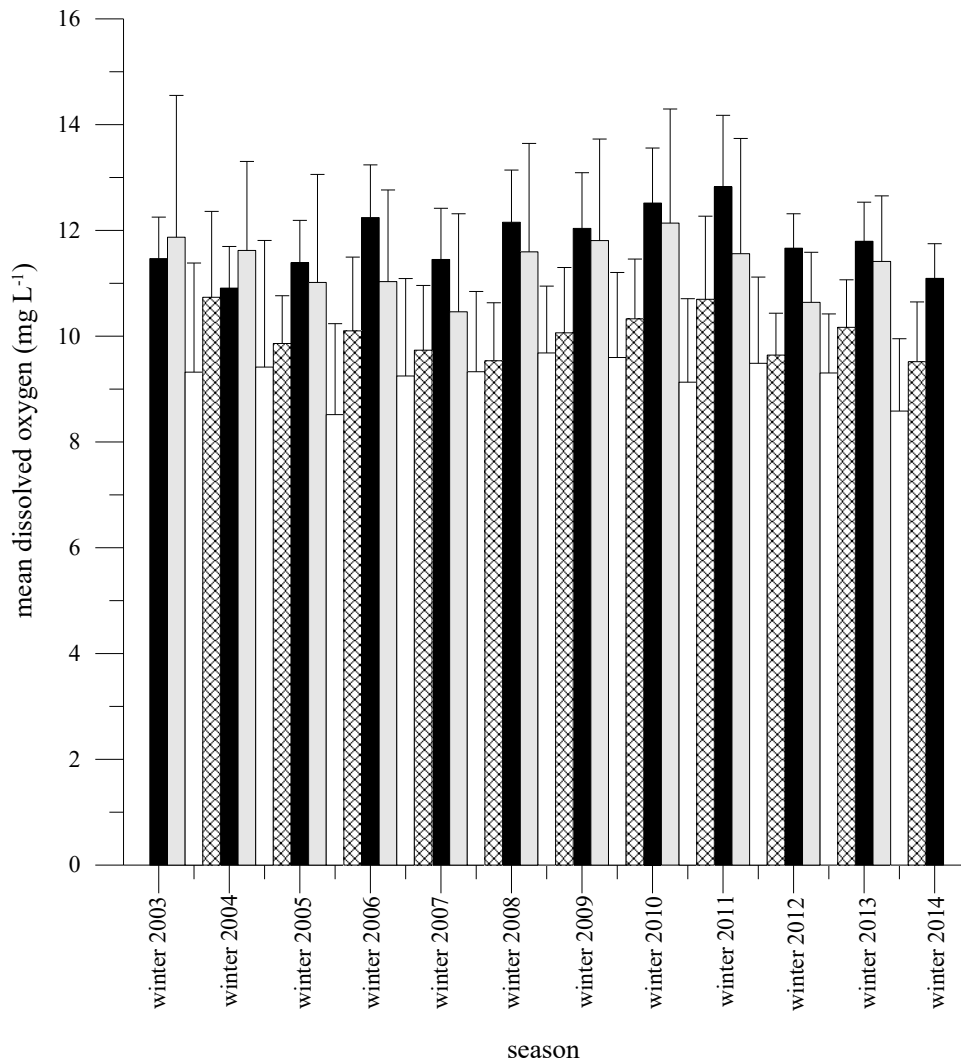


Figure 3.33: Mean seasonal whole-Bay surface dissolved oxygen concentrations in Cardiff Bay from winter 2003 until winter 2014. winter: black bars, spring: grey bars, summer: white bars, autumn: hatched bars (standard error bars).

Mean whole-Bay seasonal surface DO concentrations correlated negatively with mean winter air temperature (Figure 3.34). Mean winter whole-Bay surface water temperature correlated weakly with DO (Figure 3.34) in Cardiff Bay. An increase in temperature caused a decrease in DO. There were no other seasonal relationships between DO and air or water temperature in Cardiff Bay.

Between April 2012 and April 2014, seasonally, dissolved oxygen (DO) concentrations varied significantly ($F_{3,55} = 13.4$, $P < 0.01$). Winter DO concentrations ($11.94 \pm 0.1 \text{ mg L}^{-1}$) were significantly higher than those experienced during summer ($9.4 \pm 0.16 \text{ mg L}^{-1}$) and autumn ($10.26 \pm 0.1 \text{ mg L}^{-1}$), (Figure 3.35) and were weakly correlated with air ($r = 0.6$, $n = 9$, $P <$

0.05) and water temperatures ($r = 0.7, n = 82, P < 0.01$). Autumn DO concentrations were also weakly correlated with air ($r = 0.7, n = 9, P < 0.01$) and water ($r = 0.7, n = 103, P < 0.01$) temperatures. Spring DO concentrations were weakly influenced by air temperature ($r = 0.7, n = 12, P < 0.01$) and were significantly higher than summer DO concentrations by 1.8 ± 0.27 mg L⁻¹ (Figure 3.35) which were weakly correlated with water temperature ($r = 0.5, n = 101, P < 0.01$).

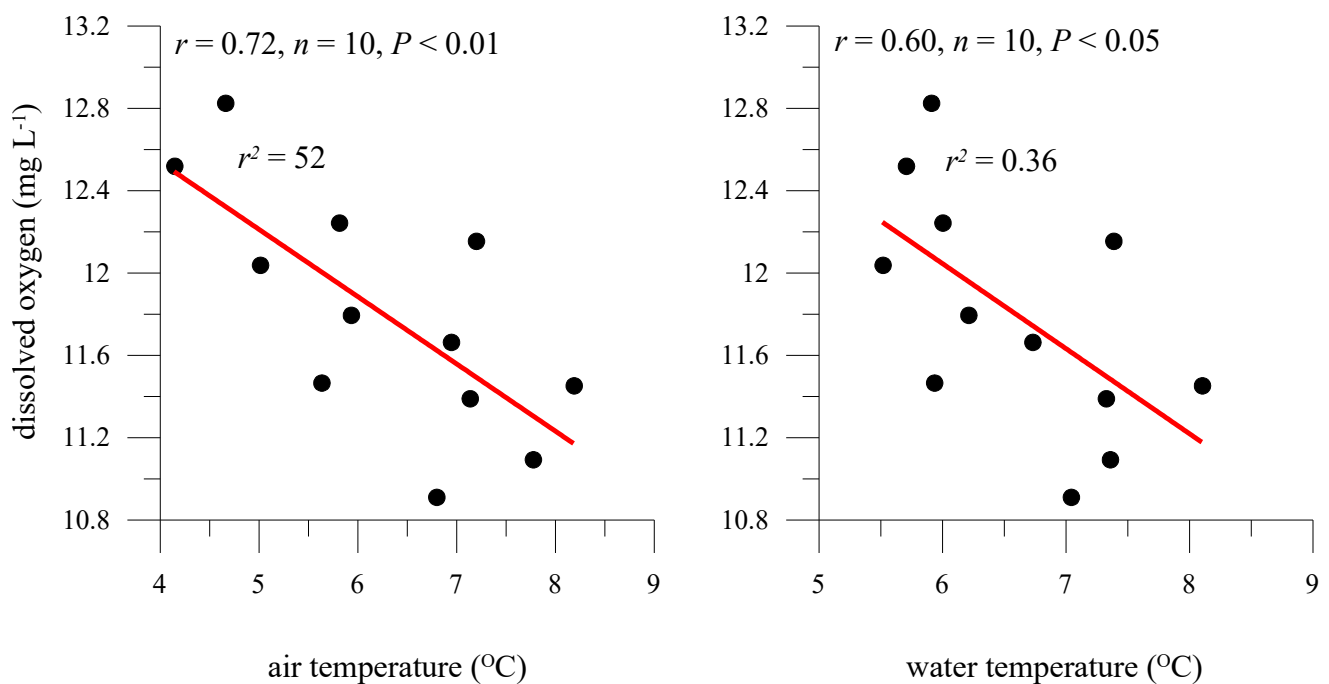


Figure 3.34: Mean winter air temperature and mean winter whole-Bay surface water temperature with mean winter whole-Bay seasonal dissolved oxygen concentrations. Data from winter 2003 until winter 2014.

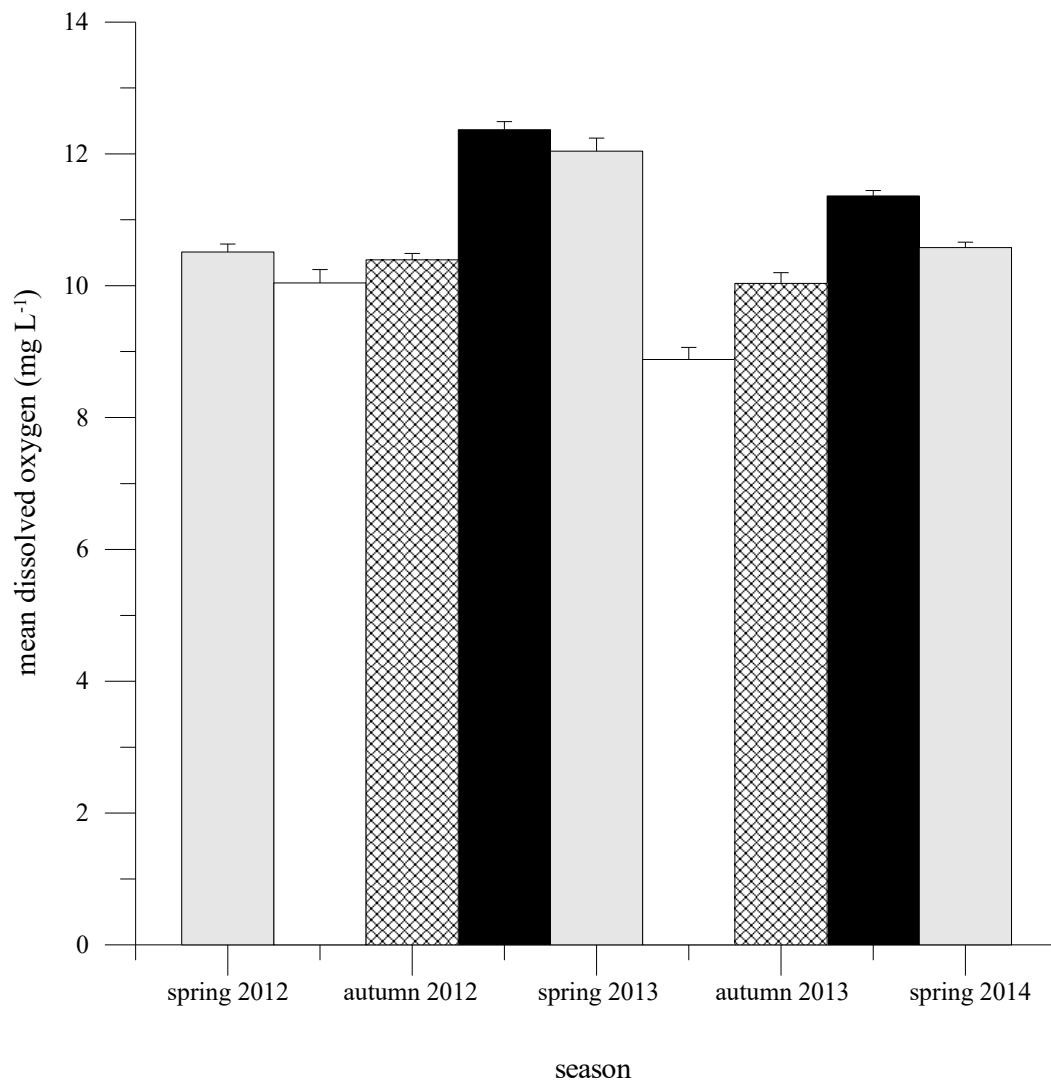


Figure 3.35: Mean seasonal whole-Bay surface dissolved oxygen in Cardiff Bay from spring 2012 until spring 2014. winter: black bars, spring: grey bars, summer: white bars, autumn: hatched bars (standard error bars).

The mean whole-Bay DO concentrations between 2003 and 2014 were significantly different between months ($F_{11,135} = 26$, $P < 0.01$). The months of January, February, March, April and December had significantly higher DO concentrations than May, June, July, August, September, October and November. May had significantly higher DO concentrations than June and July. June, July, August and September also had significantly lower DO concentrations than November (Figure 3.36).

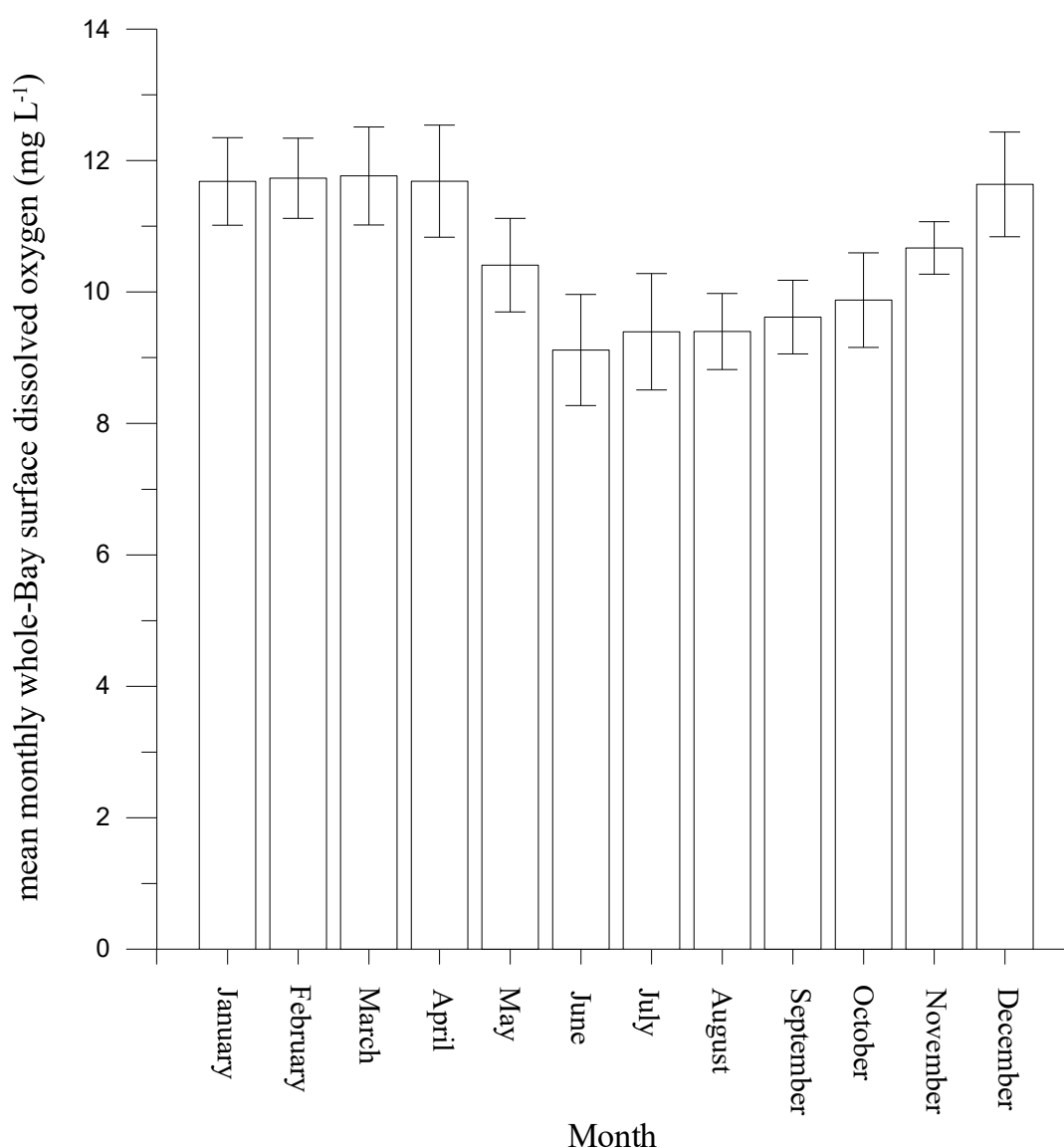


Figure 3.36: The mean monthly whole-Bay surface dissolved oxygen concentrations in Cardiff Bay between January 2003 and April 2014 (standard error bars).

The surface oxygen saturation of Cardiff Bay can be seen in Figure 3.37. For 41 months (30% of the time) (Table 3.2), between January 2003 and April 2014 (totalling 136 months), the surface water of Cardiff Bay was over 100% saturated with oxygen. This oversaturation occurred during at least one month of spring (March, April, May) every year from 2003 until 2012. The year of 2003 experienced the most oversaturation of DO. There were no variations of DO with depth experienced between January 2003 and April 2014 at any of the sites studied, however, the analysis of mean monthly individual site DO concentrations taken between April

2012 and April 2014 at 1m intervals throughout the water column indicated that, with the exception of site A, situated at the mouth of the River Ely, (Figure 3.38), there were no differences in DO throughout the water column in Cardiff Bay. Statistically at site A, the DO concentrations at 7 m were lower than those found at 0-5 m depth ($F_{7,347} = 3.42$, $P < 0.002$). Mean monthly DO concentrations and mean monthly water temperature were significantly negatively correlated throughout the water column at site A (Figure 3.39) during the period between April 2012 and April 2014.

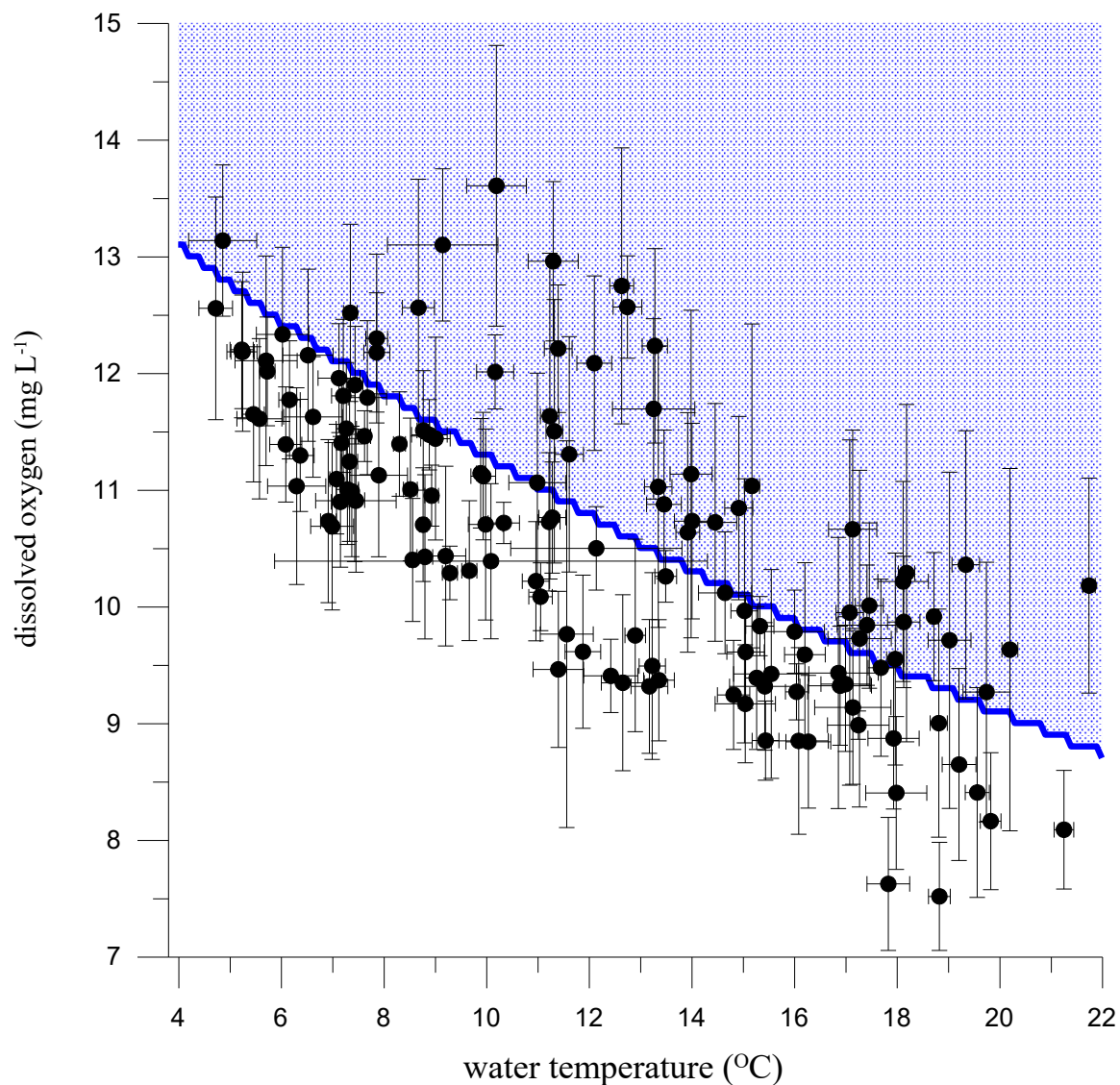


Figure 3.37: The mean monthly whole-Bay surface water temperature and dissolved oxygen concentration in Cardiff Bay (black dots with standard error bars) between January 2003 and April 2014. In comparison to 100% dissolved oxygen saturation in an inert system (blue line and shading).

Table 3.2: The months between January 2003 and April 2014 where mean monthly whole-Bay dissolved oxygen concentrations were over 100% saturation in the surface waters of Cardiff Bay.

Month	DO (mg L⁻¹)	100% DO saturation (mg L⁻¹)
March 2003	12.6	11.7
April 2003	12.1	10.8
May 2003	11.0	10.5
June 2003	9.9	9.3
August 2003	9.6	9.1
September 2003	9.6	9.7
October 2003	11.6	11.0
May 2004	11.0	10.1
July 2004	9.9	9.4
August 2004	10.4	9.2
April 2005	11.5	11.0
May 2005	10.6	10.3
August 2005	9.7	9.3
September 2005	9.7	9.6
May 2006	10.7	10.3
June 2006	9.9	9.4S
July 2006	10.2	8.8
September 2006	9.6	9.5
April 2007	11.7	10.5
February 2008	12.5	12.1
March 2008	12.3	11.9
April 2008	12.8	10.6
June 2008	9.8	9.6
July 2008	10.7	9.6
September 2008	10.7	10.2
February 2009	10.9	10.4
April 2009	13.0	11.0
May 2009	11.1	10.3
August 2009	10.2	9.4
December 2009	12.2	11.9
January 2010	13.1	12.8
April 2010	12.6	10.6
May 2010	10.8	10.1
July 2010	9.3	9.1
November 2010	11.3	10.9
December 2010	13.6	11.3
February 2011	12.2	11.0
March 2011	13.1	11.5
April 2011	12.2	10.5
July 2011	10.0	9.6
February 2012	12.0	11.3

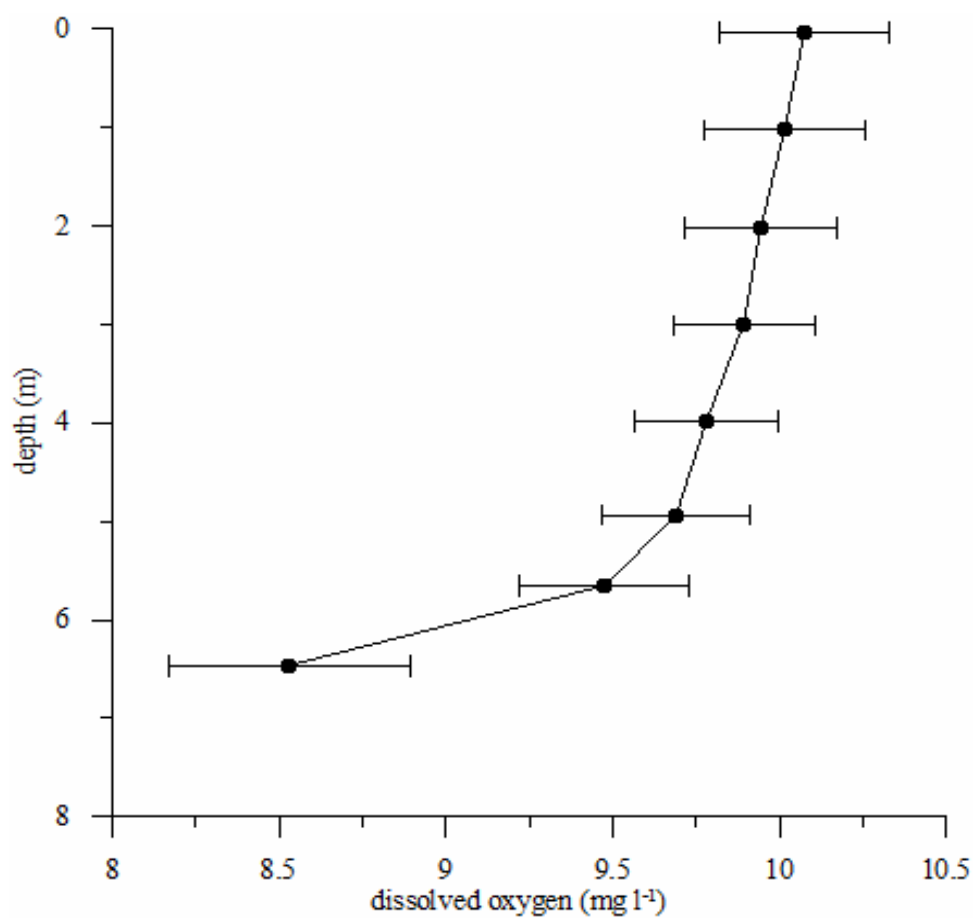


Figure 3.38: Mean dissolved oxygen concentrations with depth at site A in Cardiff Bay. Data taken twice a month from April 2012 until April 2014 (standard error bars).

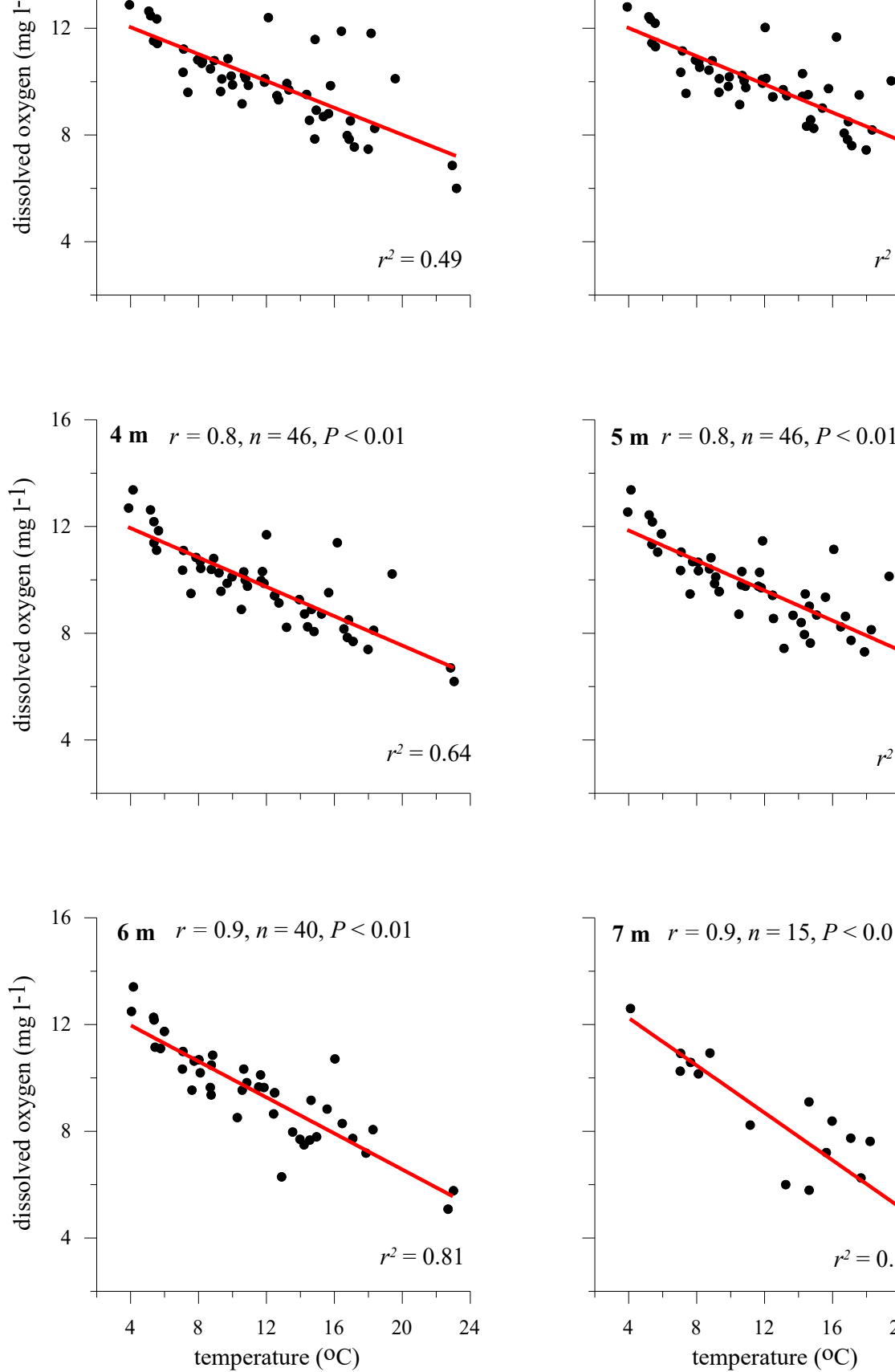


Figure 3.39: Mean monthly water temperature and mean monthly dissolved oxygen concentrations at site A in Cardiff Bay, 2014. Individual graphs show depth below the lake surface.

Mean monthly whole-Bay surface DO correlated with mean monthly air and mean monthly whole-Bay surface water temperature. When air and water temperatures increased, DO concentrations decreased (Figures 3.40 and 3.41 respectively).

Using mean whole-Bay monthly data, backward elimination for surface DO concentrations in Cardiff Bay indicated that mean monthly air temperature had more correlation with DO concentrations between 2003 and 2014 than mean monthly whole-Bay surface water temperature (Equation B). Air temperature was significant ($P < 0.01$), but water temperature was not.

Equation B: (*Dissolved Oxygen*) = $73.0 + 6.3$ (*air temperature*) + 1.2 (*water temperature*); $F_{2,133} = 127.7$, $P < 0.01$, $r^2 = 0.66$).

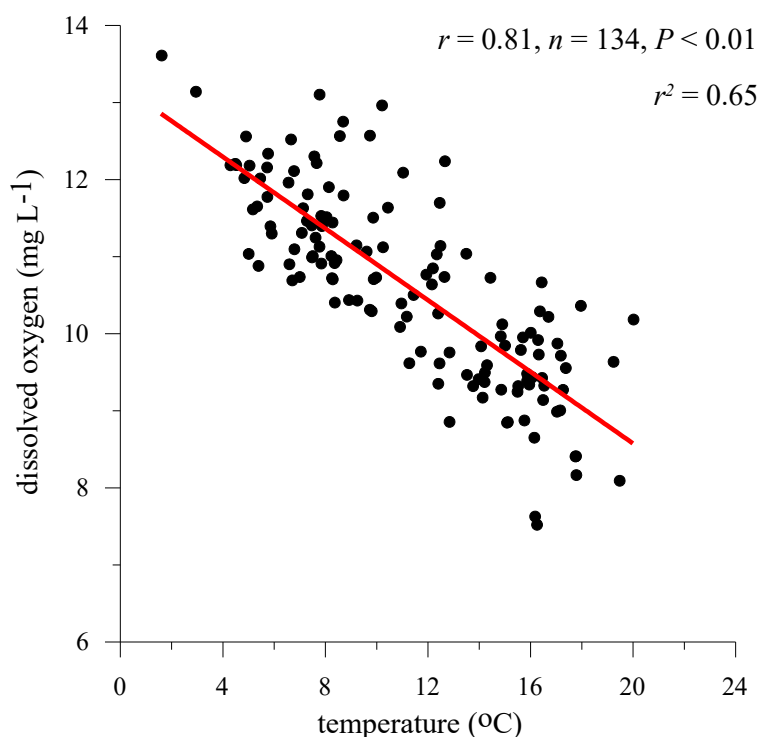


Figure 3.40: Mean monthly air temperature and mean monthly whole-Bay surface dissolved oxygen concentrations in Cardiff Bay recorded between January 2003 and April 2014.

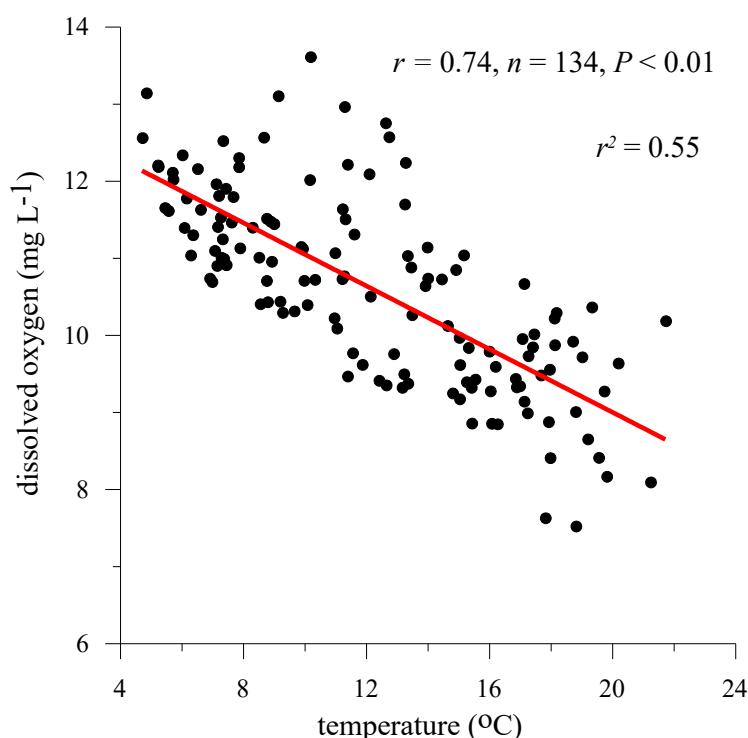


Figure 3.41: Mean monthly whole-Bay surface water temperature and mean monthly whole-Bay surface dissolved oxygen concentrations in Cardiff Bay recorded between January 2003 and April 2014.

Analysis of mean annual surface DO concentrations indicated that there was no statistical difference in concentrations between sites. However, mean seasonal surface DO concentrations were different between sites, (winter: $F_{5,71} = 4.1$, $P < 0.01$, spring: $F_{5,65} = 5.23$, $P < 0.001$, summer: $F_{5,65} = 11.93$, $P < 0.001$, autumn: $F_{5,65} = 9.89$, $P < 0.001$). Mean seasonal surface DO concentrations at site 17 were lower than all the other sites surveyed during summer and autumn. Site 17 also had lower mean spring surface DO concentrations than sites 5, 6, 9 and 10. During winter, site 4 mean seasonal surface DO concentrations were significantly higher than those at site 17. Mean monthly surface DO concentrations at site 17 were significantly lower than other sites throughout the year (Table 3.3). However, analysis of mean monthly surface DO concentrations between April 2012 and April 2014 did not show any difference in DO spatially within the lake.

Month	Site(s)	Statistics
January	4	$F_{5,71} = 3.4, P < 0.01$
February	4	$F_{5,71} = 3.4, P < 0.01$
April	6 & 10	$F_{5,71} = 3.5, P < 0.01$
May	9 & 10	$F_{5,65} = 3, P < 0.05$
June	4	$F_{5,65} = 2.4, P < 0.05$
July	4, 5, 6, 9 & 10	$F_{5,65} = 4.3, P < 0.01$
August	4, 5, 6, 9 & 10	$F_{5,65} = 7.7, P < 0.001$
September	4, 5, 6, 9 & 10	$F_{5,65} = 8.5, P < 0.001$
October	4, 5, 6, 9 & 10	$F_{5,65} = 8.5, P < 0.001$
November	4 & 5	$F_{5,65} = 3.1, P < 0.05$

Table 3.3: The significant difference of mean monthly surface dissolved oxygen concentrations at site 17 in comparison with other sites in Cardiff Bay, taken between 2003 and 2014.

3.5.8 The Photic Depth of Cardiff Bay

Between April 2012 and April 2014 there was no seasonal difference in the mean whole-Bay photic depth of Cardiff Bay (Figure 3.42). The photic depth at site 4 was statistically lower than site C by 1.14 ± 0.46 m ($F_{7,408} 2.2, P < 0.05$) (Figure 3.43), however this difference was not strong. There was no other spatial variation of light penetration within the lake.

Mean monthly photic depth and mean monthly combined river discharge were weakly correlated at sites 10, A, B, C and D. There was no correlation between mean monthly photic depth and water column depth within Cardiff Bay.

The photic depth at site 10 had a weak correlation with lake flushing rate ($r = 0.43$, $n = 23$, $P < 0.05$). As did site A ($r = 0.58$, $n = 23$, $P < 0.01$, $r^2 = 0.34$), site C ($r = 0.59$, $n = 23$, $P < 0.01$, $r^2 = 0.35$), and site D ($r = 0.53$, $n = 23$, $P < 0.01$, $r^2 = 0.28$). Mean whole-Bay photic depth also correlated weakly with lake flushing rate ($r = 0.48$, $n = 23$, $P < 0.05$, $r^2 = 0.23$).

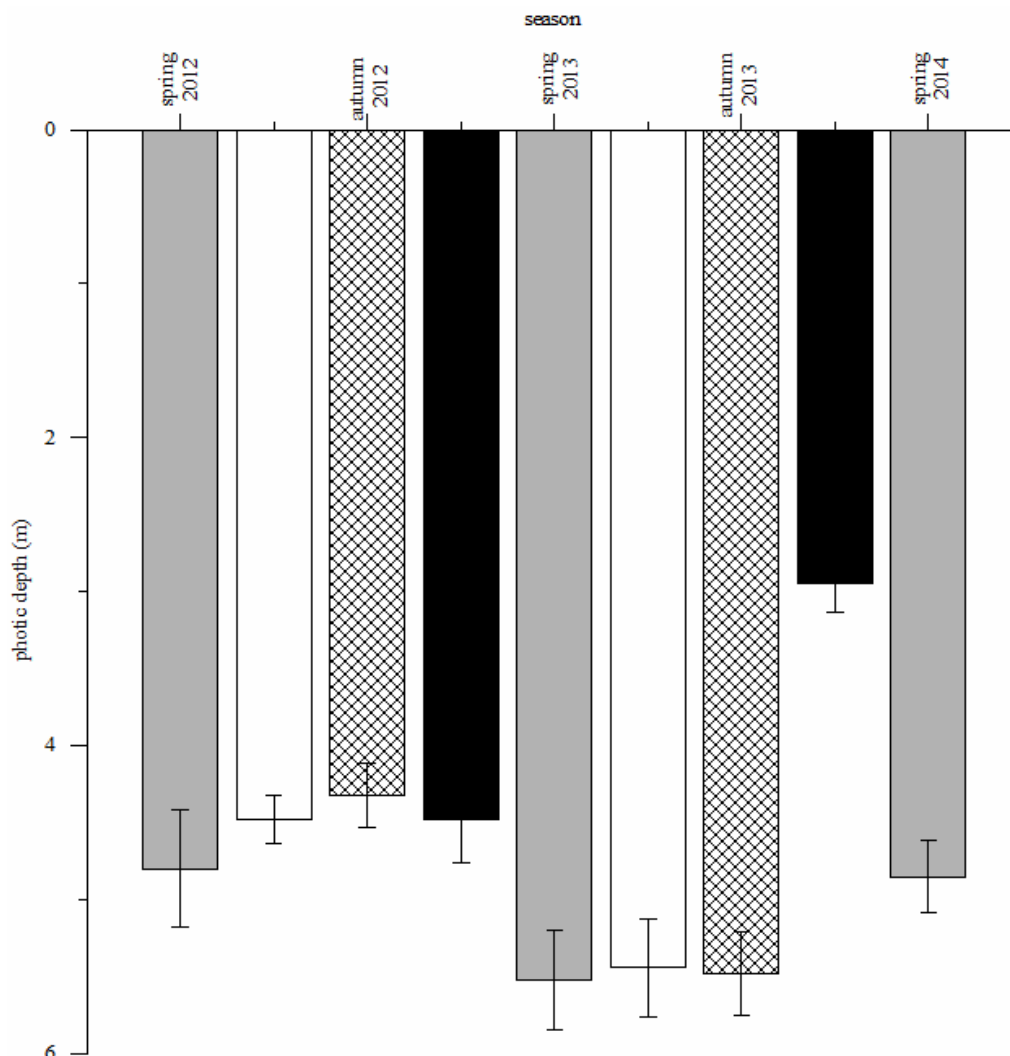


Figure 3.42: Mean seasonal whole-Bay photic depth of Cardiff Bay recorded between April 2012 and April 2014. winter: black bars, spring: grey bars, summer: white bars, autumn: hatched bars (standard error bars).

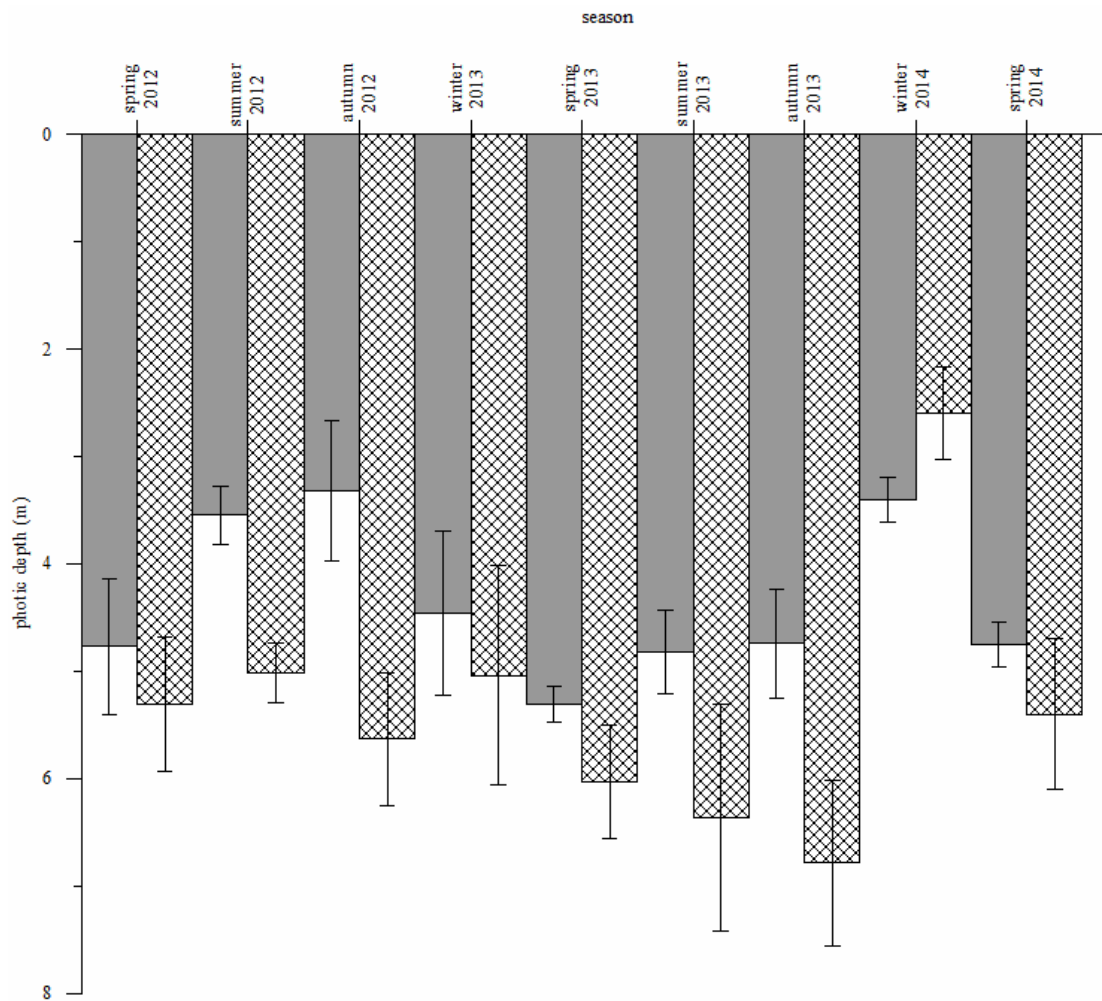


Figure 3.43: Mean seasonal photic depth at site 4 (grey bars) and site C (hatched bars) recorded in Cardiff Bay between April 2012 and April 2014 (standard error bars).

3.6 Discussion

Cardiff Bay was subject to seasonal variation in air and water temperatures, precipitation, river discharge and lake flushing rate. During summer, the combination of higher air and water temperatures with lower precipitation caused river discharge to decrease and therefore, lake flushing rates slowed. The lake was polymictic throughout the year; there was no spatial variation of water temperature, dissolved oxygen concentration or photic depth. Stratification

did not occur at any time or at any location in the lake, causing water temperatures to be uniform with depth. The aeration system within Cardiff Bay appeared to aid the mixing of the water column and ensured that DO concentrations were kept at the legally binding 5 mg/L⁻¹. However, DO concentrations at site 17, in the River Ely were lower than those sites studied within the lake body itself and between 2012 and 2014 site A, also in the River Ely experienced lower DO concentrations at the lake bed. Although the aeration system did not prevent a seasonal variation of DO, it is likely that its use was enough to balance the removal of DO at depth, with DO production. The IPCC has predicted that there will be an increase in temperatures and intense precipitation events. This study suggests that Cardiff Bay will be affected by the impacts of these climatic alterations, therefore future management of the lake needs to take these scenarios into consideration to ensure that phytoplankton productivity does not increase uncontrollably as a result.

3.6.1 Temporal Variation in Air Temperature of Cardiff Bay

The general trend for air temperature in Cardiff Bay has been a decrease post 2006. In contrast, Global surface temperatures have increased by 0.74 °C over the past 100 years, with most of this warming (0.55 °C) post 1980 (Jeppesen et al., 2009). Although Wales has witnessed a steady warming of mean annual air temperatures since the early 1900s (Defra, 2012b), there has been a notable slowing in this rate of increase since 2005 (Figure 3.44). Hence this slowing of air temperature in Wales is reasonably close to the decrease in air temperature in Cardiff Bay. Note that the magnitude of this decrease in Cardiff Bay was 0.8 °C (annual average air temperature of 2003 – 2006 compared to 2007 - 2013). Throughout the UK a peak in air temperature during 2006 and a decrease in 2010 was experienced (Figure 3.45) (Met Office, 2009d). Compared to the mean temperatures experienced in the UK between 1981 and 2010, the year of 2006 experienced unusually high temperatures during every month of the year in Wales except March, April and November (Figure 3.46) (Met Office, 2009b), and 2010 experienced unusually low temperatures during 7 months of the year (Figure 3.47). December 2010 experienced a difference of -5 °C compared to the 1981-2010 average in Wales (the coldest temperature in the UK apart from those experienced in Northern Ireland) (Met Office, 2009c). This was reflected in the number of ‘cold days’ witnessed in Cardiff Bay during 2010.

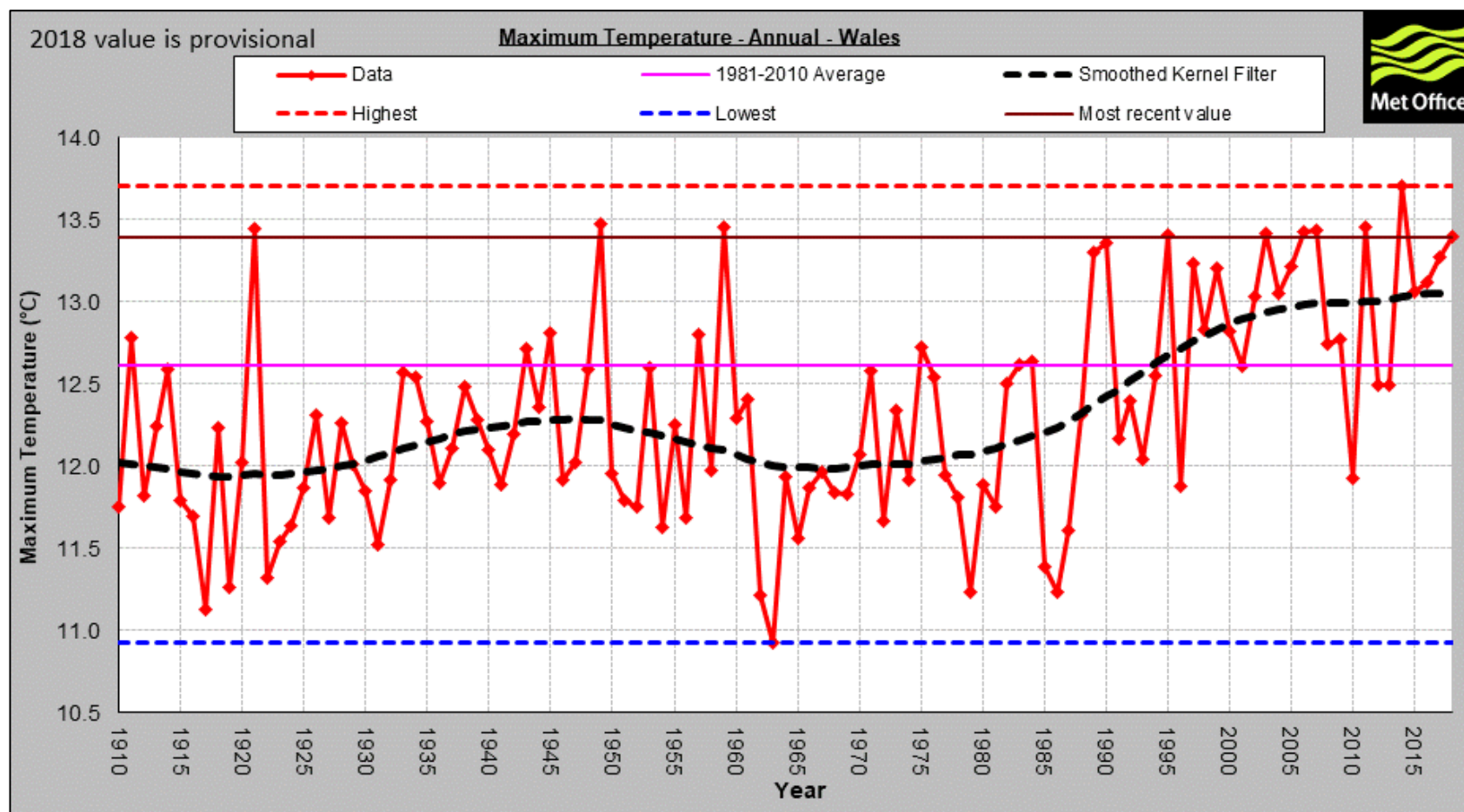


Figure 3.44: Maximum annual air temperatures experienced in Wales since 1910. Met Office sourced graph (Met Office, 2009d).

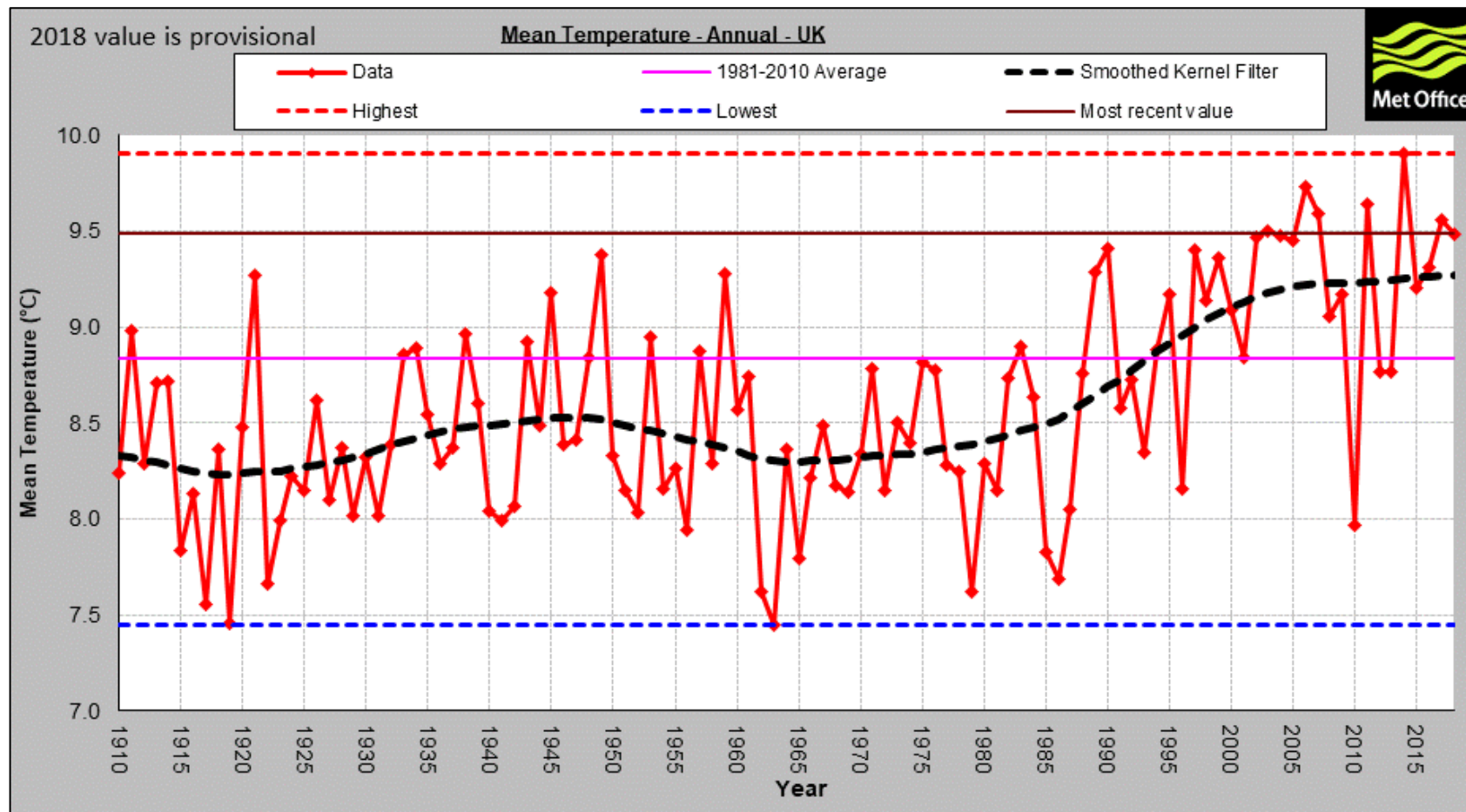


Figure 3.45: Mean annual air temperature in the UK since 1910. Met Office sourced graph (Met Office, 2009c).

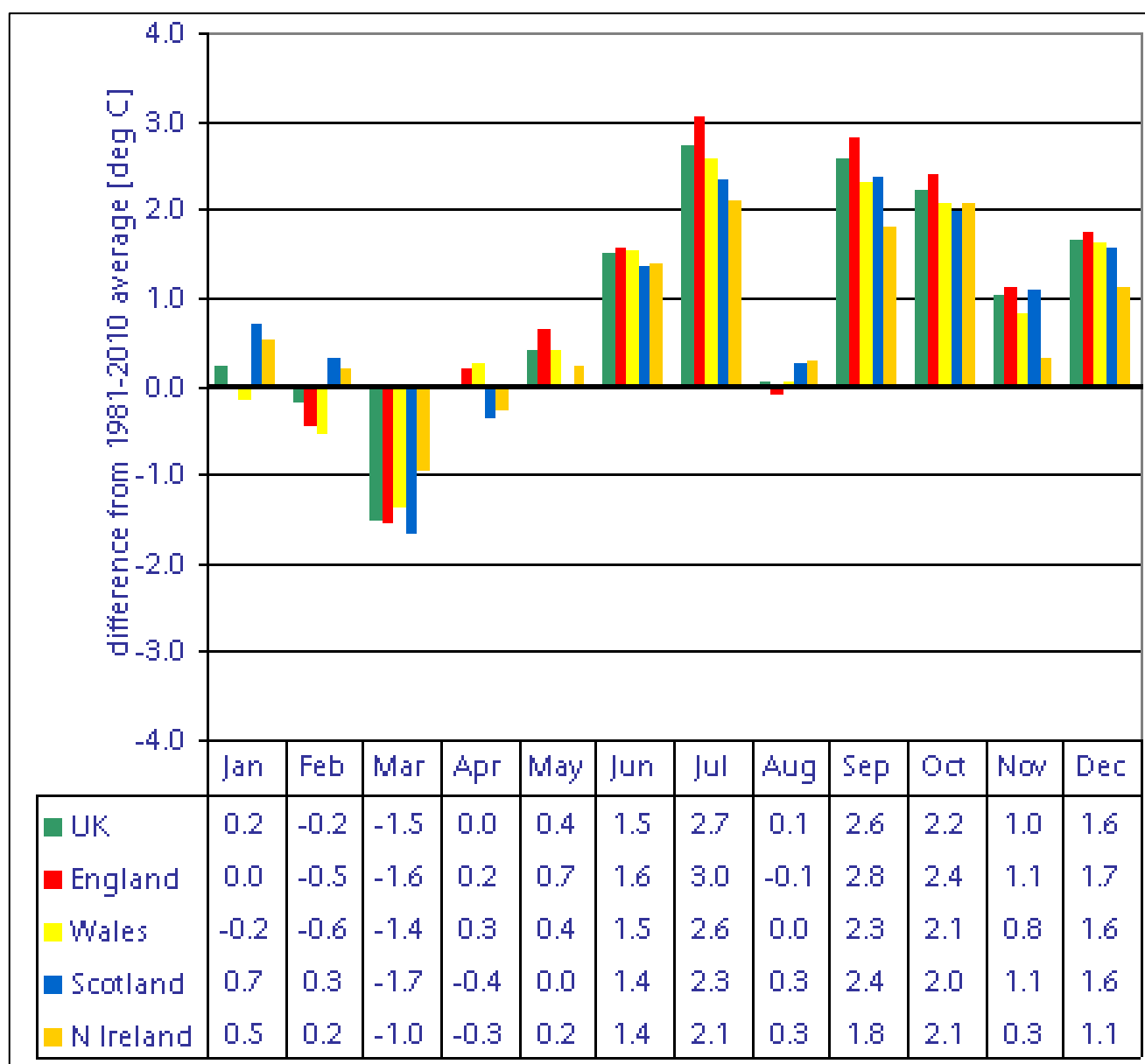


Figure 3.46: Mean air temperatures (1981 – 2010), anomalies for 2006. Met Office sourced graph (Met Office, 2009b).

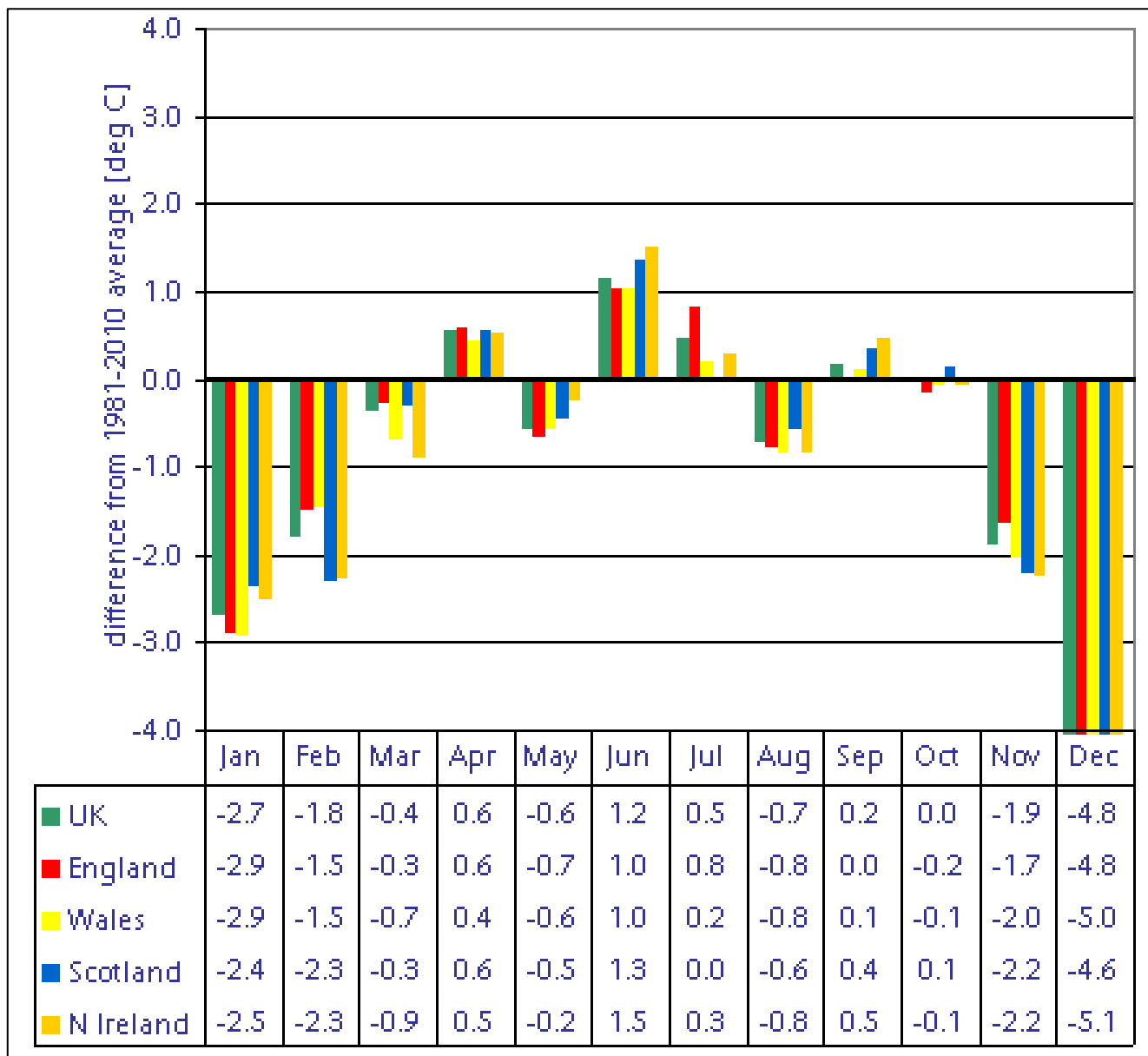


Figure 3.47: Mean air temperatures (1981 – 2010), anomalies for 2010. Met Office sourced graph (Met Office, 2009b).

The lowest winter NAO Index was experienced in 2010, corresponding with the lowest air temperatures experienced between 2003 and 2014 in Cardiff Bay. The North Atlantic Oscillation (NAO) is primarily a winter phenomenon (Gerten & Adrian, 2000b, Ottersen et al., 2001, Blenckner et al., 2007a) explaining why its connection with air temperature is strongest during the winter. This strong seasonal correlation was also experienced during autumn,

suggesting that the NAO began to influence the air temperature of Cardiff Bay before winter began. This indicates that large scale climatic influences on temperature local to Cardiff Bay may be an important consideration when predicting changes in chemical and biotic processes in the water system of Cardiff Bay itself.

Whilst climate models will never be able to exactly reproduce the real climate system, the UKCP09 projections demonstrate that there is enough similarity between the UKCP09 model and the observed world temperature to assume that it provides plausible projection of future climate changes (Murphy et al., 2009). UKCP09 indicates that the air temperature of Wales will increase over time with an annual average increase of 3-4 °C from the 1961 – 1990 baseline. The annual average increase in air temperature in Wales has been projected to rise by 1.3 °C by the 2020s, 2.0 °C by the 2040s and 3.3 °C by the 2080s under the medium emissions scenario (Welsh Assembly Government, 2009b).

3.6.2 The Relationship between Air & Surface Water Temperature in Cardiff Bay

An increase in air temperature caused an increase in the surface water temperature of Cardiff Bay. Water has a large heat capacity, therefore a change in air temperature alters the amount of atmospheric radiation on the lake surface, affecting the rate of latent and sensible heat exchange (Livingstone, 2003, Fink et al., 2014). The year of 2010 experienced lower annual mean air temperatures than any other year studied between 2002 and 2013, however due to water having a high heat capacity, which causes lake waters to cool slower than the atmosphere (Wetzel & Likens, 2000), the mean annual surface water temperature of Cardiff Bay did not reflect this.

The relationship between air temperature and surface water temperature is not primarily one of cause and effect. The heat balance within a lake is determined by heat exchange processes causing numerous variables, including evaporation and condensation rates at the lake surface (Livingstone & Lotter, 1998), to have an impact on the surface water temperatures of Cardiff Bay. This will cause the seasonal relationship between air and water temperature to vary. Cooler temperatures experienced during winter reduces evaporation rates and the relationship between air and water temperature becomes less variable.

3.6.3 Temporal Variation in Precipitation Changes in Cardiff Bay

There was no annual variation of precipitation in Cardiff Bay. This corresponds with research by Jenkins et al. (2009), where precipitation in the UK has shown no annual trend since 1766. Although intensive precipitation events are becoming more common, they are not represented in annual precipitation data sets (Defra, 2012b). Annually, there are no overall changes (-10 – 0 %) in precipitation projected for Wales up to the 2080s when compared to the 1961 – 1990 baseline under the IPCC medium emissions scenario (Welsh Assembly Government, 2009a).

Cardiff Bay displayed a seasonal change in precipitation, with autumn and winter levels being higher than those in summer and spring. A study of trends in UK extreme precipitation by (Jones et al., 2013) found that there has been an increase in the intensity of long-duration events and an increase in winter precipitation, but a decline in intensity and duration during summer (Prosdocimi et al., 2013). Despite the NAO influence on seasonal air temperatures, there was no relationship in Cardiff Bay between the NAO and precipitation. Relationships between NAO and precipitation are very likely during certain timescales but are often not obvious or easily retrieved in precipitation records (Massei et al., 2007). However, the UKCP09 and IPCC predicted that mean summer precipitation will decrease over time (Table 3.4) and average winter precipitation will increase (Table 3.5); changes which are likely to occur within Cardiff Bay in the future.

In Cardiff Bay, July 2009 precipitation levels were unusually high. Throughout the UK, July 2009 was the wettest July on record (in a series from 1914), and it was also the wettest July recorded in Wales (Met Office, 2013), experiencing 272% of the 1961 – 1990 mean precipitation (Met Office, 2009a).

The variability and unpredictability of precipitation, along with the potential for large precipitation events have the ability to cause variability in river flow (Prosdocimi et al., 2013) and lake nutrient loadings (Morales et al., 2001). Therefore, changes in precipitation and how we adapt to them are becoming increasingly important when managing lake systems. However, it is acknowledged that there is a difficulty in detecting changes in precipitation behaviour, particularly as patterns are currently experiencing change (Jones et al., 2013). This highlights the importance of understanding how the phytoplankton community of Cardiff Bay respond directly and indirectly to precipitation variability.

Table 3.4: Seasonal average air temperature projections for Cardiff Bay in comparison to the 1961-1990 baseline. Data re UKCP09 (Welsh Assembly Government, 2009a).

Emissions scenario	Winter			Spring			Summer			Autumn
	2020	2050	2080	2020	2050	2080	2020	2050	2080	2020
Low	1.21	1.85	2.5	1.29	1.83	2.47	1.68	2.57	3.11	1.63
Medium	1.3	2.14	2.92	1.26	2.17	3.07	1.61	2.89	4.18	1.57
High	1.13	2.36	3.53	0.89	2.47	3.86	2.77	3.23	5.25	2.01

Table 3.5: Seasonal average precipitation projections for Cardiff Bay in comparison to the 1961-1990 baseline. Data re UKCP09 (Welsh Assembly Government, 2009a)

Emissions scenario	Winter			Spring			Summer		
	2020	2050	2080	2020	2050	2080	2020	2050	2080
Low	7.4	14.36	22.91	1.08	0.81	0.49	-6.52	-12.7	-15.02
Medium	8.47	20.44	28.8	-0.39	-0.28	0.39	-7.04	-19	-23.25
High	7.69	21.02	39.37	0.19	-0.22	0.87	-3.93	-19.57	-28.84

3.6.4 The Impact of Precipitation on River Taff and River Ely Discharge

The correlation between the river discharges of both the Taff and Ely and precipitation was weak, and when this was integrated up to annual mean values, insignificant. This is likely due to the nature of the catchment for each river regarding their geology and hydrology, which is beyond the scope of this study. However, findings did show that the lower discharge of the River Ely was still significant to determine the overall river discharge and its influences on the lake system. From a management perspective, it is important that the monitoring of both river discharges is continued to determine the total river discharge into Cardiff Bay to allow the modelling of the flushing rate of the Bay as a whole.

Although the relationship between precipitation and river discharge in winter was found to be weak, it is still relevant as precipitation rates are unpredictable. During winter, precipitation had the highest influence on River Taff and River Ely discharge where an increase in precipitation caused an increase in discharge rates. This is a result of river systems being fed primarily by precipitation and ground water, as well as being influenced by temperature through evapotranspiration rates (Kriauciuniene et al., 2012). The highest levels of precipitation in Cardiff Bay were found during autumn and winter, resulting in a direct relationship in winter between precipitation and river discharge. The indirect transfer of precipitation into rivers by groundwater flow significantly influences river discharge rates (Andermann et al., 2012). Ground water levels are often at their highest during winter due to saturation by precipitation (Changnon et al., 1988), supplying the rivers with more runoff and increasing volume and therefore discharge of rivers. In the UK, total annual evapotranspiration can amount to more than 50% of the total annual precipitation (Dunn & Mackay, 1995), making it an important factor to consider when interpreting the effect of precipitation on river discharge. During winter, temperatures are at their lowest therefore rates of evapotranspiration are low, allowing a higher percentage of precipitation to reach rivers. In contrast to this, during summer, evapotranspiration rates are higher than at other times of the year and soils are less saturated, reducing surface run off and groundwater flow into the rivers. This can lead to longer lag times, possibly explaining the reduced effect of the high precipitation event in July 2009 on River Taff and River Ely discharge rates.

The extreme precipitation event in July 2009 is a potential example of what Cardiff Bay may expect to experience more frequently in the future if the IPCC climate change scenario prediction of increasing extreme weather event frequency transpires. With reduced summer rainfall, these events can have a greater magnitude of effect and may cause either a) a short-term peak in river discharge that flushes out the Bay or b) may cause a pulse of nutrients that stimulate phytoplankton productivity (Sadro & Melack, 2012).

3.6.5 The Flushing Rate of Cardiff Bay

The absence of annual variation in the flushing rate of Cardiff Bay is likely to be caused by the absence of annual variation in precipitation and combined river discharge, both strong influences on lake flushing rate. Summer experienced the lowest flushing rates; this season also experienced the lowest river discharge and the second lowest precipitation levels between 2003 and 2014. The flushing rate of Cardiff Bay is relatively quick. In comparison; Trasimeno lake, Italy has a mean depth of 4 m (0.5 m shallower than Cardiff Bay) and a volume of 590,000,000 m³ ($\times 10^2$ more than Cardiff Bay) but has a mean flushing rate of 21.6 years (Ambrosetti et al., 2003) and Bassenthwaite Lake, Cumbria, UK has a mean depth of 21 m (16.6 m deeper than Cardiff Bay) and a volume of 27,700,000 m³ (10 times larger than Cardiff Bay) but has a flushing rate of 19 days (Thackeray et al., 2010).

Understanding the flushing rate of Cardiff Bay by measuring the inputs and outputs of the lake is essential for its management. It enables the movement of nutrients in and out of the lake to be tracked and the extent of nutrient and phytoplankton dilution and removal to be determined (Interagency Freshwater Group, 2015). A low flushing rate can exaggerate nutrient loading problems and increase the likelihood of eutrophication because lake waters are not replenished often enough to prevent the accumulation of nutrient-rich runoff which can lead to increased amounts of nutrients in the lake sediment. Therefore, the reduced flushing rate experienced during summer in Cardiff Bay has the potential to encourage phytoplankton blooms (Interagency Freshwater Group, 2015), making it a key factor to analyse for Cardiff Bay lake management.

3.6.6 Spatial & Vertical Variation of Water Temperature in Cardiff Bay

The lack of spatial variation in surface water temperature, and the absence of a thermocline within Cardiff Bay indicates that the lake is well-mixed and can be classified as polymictic. This has important management implications as reduced variation of water temperature, nutrient loading and dissolved oxygen concentrations between surface and bottom waters characteristic of a polymictic lake make it more prone to changes in climate (Wilhelm & Adrian, 2008). Mixing can cause hypolimnetic temperatures to rise as well as supply them with dissolved oxygen, the combination of which can cause organic matter mineralisation rates to increase (Fortino et al., 2014).

If water temperatures rise as a result of the predicted increase in summer air temperature like many lakes worldwide, Cardiff Bay could be more prone to stratification during warm periods (Trolle et al., 2011), making the need for the aeration system within the lake more prominent. If used effectively the aeration system within Cardiff Bay has the potential to prevent the development of a thermocline and therefore stratification through the process of artificial circulation (Kortmann et al., 1994).

However, if the aeration system within Cardiff Bay is not able to prevent stratification from occurring in the lake, there is a risk that it could become dimictic. Therefore, there is the potential for seasonal changes in temperature to cause alternative stratification and mixing events in the lake (Wilhelm & Adrian, 2008). Multiple nutrient pulses into the euphotic layer can take place during summer, altering the nutrient budget of the lake. Nutrient release can also occur at the sediment-water interface during periods of stratification as oxygen levels are depleted and hypolimnetic temperatures are higher (Wilhelm & Adrian, 2008). This could not only create low DO conditions in the lake of Cardiff Bay, particularly at depth, but it could also create ideal conditions for the growth of cyanobacteria. The strength and turbulence of mixing within a lake, and the influence this has on water temperatures and nutrient availability, influences phytoplankton growth and competition; during periods of stratification, phytoplankton growth is often stimulated (Wilhelm & Adrian, 2008).

Cardiff Bay is a relatively small and shallow lake and both characteristics often prevent thermocline development (Burt, 2002). It also has a relatively large input of water from the River Taff and River Ely which contributes to the fast flushing rate, therefore the turnover of

the water in the lake is high and causes constant mixing to take place. The lake can be considered to be more like a slow-flowing river system rather than a typical lake. This makes understanding the likelihood of Cardiff Bay stratifying in the future essential for robust lake management strategies to be created (Reynolds, 1980) as phytoplankton species composition in a shallow, polymictic lake is often very similar to that in an inflowing lowland river (Köhler & Hoeg, 2000). Shallow lakes are often efficient at converting available phosphorus into phytoplankton biomass due to the constant turnover of water and light availability (Nixdorf & Deneke, 1997) and the combination of lake flushing rate and the presence of stratification have the ability to modify phytoplankton structure within Cardiff Bay (Köhler & Hoeg, 2000). Therefore, the ability to detect or predict any changes in flushing rate or the presence of stratification and its duration within Cardiff Bay can provide CHA with the ability to predict how the phytoplankton community structure will change as a result.

3.6.7 The Dissolved Oxygen Levels of Cardiff Bay

Despite water temperature changes being the most common cause of annual variation in lake DO concentrations (Langman et al., 2010), this relationship was not apparent in Cardiff Bay, suggesting that other factors are more prominent when determining DO fluctuations long-term. Reaeration and photosynthetic oxygen production are often the main oxygen inputs to system, and respiration is the main oxygen sink. The amplitude of these inputs and outputs are dependent on the presence of macrophytes and abundance of fish within the lake (Ansa-Asare et al., 2000). These processes and their effect on DO in Cardiff Bay are discussed in Chapter 4.

Seasonally, Cardiff Bay lake DO only display a relationship with air and water temperature during winter. Variation in DO at this timescale is often as a result of changes in water temperature and external loading (Langman et al., 2010). The loading of oxygen in a lake can be controlled by the thermal stratification pattern, which is often seasonal; an increased duration of stratification in the summer causes surface DO concentrations to be higher than those found at depth (Zhang et al., 2015), whereas during winter the water column of a lake is commonly well mixed. The absence of this seasonal occurrence of thermal stratification in

Cardiff Bay may explain why winter is the only season to experience a relationship between DO and temperature.

On a short-term (monthly) timescale DO concentrations in the lake varied as a result of both air and water temperature. This is as a result of the capacity of the lake to hold oxygen; oxygen solubility decreases with increasing water temperatures, causing DO concentrations to decrease, therefore oxygen solubility increases with a reduction in water temperature which allows DO concentrations to rise (Zhang & Tromans, 1998, Zhang et al., 2015). Lake thermal structure, irradiance, and nutrient availability has also been found to affect DO concentrations in lakes on this scale, these are discussed in Chapter 5.

Cardiff Bay had DO levels above 100% saturation frequently during spring. Although data is unavailable to determine when the aeration system was in use, it is likely that because temperatures start to warm during this season, the aeration system was used more frequently, causing supersaturation of DO despite an increase in water temperature. However, the decrease in DO during summer may indicate that the aeration system is unable to keep up with the rate of increase in water temperature increase, allowing the seasonal fluctuation in DO to occur. The year of 2003 experienced the most DO saturation, and no saturation occurred after February 2012. It is possible that the aeration system was overused after its initial implementation in 2003, and by 2012 the use of the aeration system had been decreased as a result of a better understanding of the lake processes and how often the diffuser is required, alongside an effort to reduce costs.

Site 4 and site 17 were situated within the River Taff and River Ely respectively. DO concentrations at site 4 were higher than those at site 17, likely due to the well-mixed water column at site 4 which will have exposed more water to the surface, enabling higher levels of oxygen uptake from the atmosphere; which is one of the main sources of oxygen input in a lake (Hasan et al., 2014) than at the slower moving site 17.

The DO concentrations at site A reduced at 7m depth. Site A is situated within the boat moorings at the mouth of the River Ely, subjecting it to higher levels of organic matter accumulation at the lake bed than other sites in Cardiff Bay. Biochemical oxygen demand (BOD) and sediment oxygen demand (SOD) processes both utilise oxygen from the water column to break down organic material in the water (BOD) and sediment (SOD), reducing surrounding oxygen concentrations (Mallin et al., 2006). The SOD of Cardiff Bay was

calculated as between 51 and 61 mmol m⁻² d⁻¹ in 2012 (Thomas, 2014), suggesting that oxygen was utilised from the water column at this site, but at relatively low levels. In the Arroya Colorado River mean SOD values have been recorded at 19.8 mol m⁻² d⁻¹ (Wallace et al., 2016).

The aeration system within Cardiff Bay was designed to ensure that lake bottom waters reach the surface to allow oxygen exchange with the atmosphere to occur so that DO levels can be maintained above 5 mg l⁻¹. The DO concentrations within Cardiff Bay did not fall below 7.5 mg l⁻¹ and were oversaturated 30% of the time, indicating that the aeration system is working effectively. However, it is not solely responsible for the regulation of lake DO concentrations as seasonal fluctuation indicates that there is also a natural variation of DO in the lake caused by temperature. Studies of lake BOD pre-and post-aeration indicate that BOD increases without aeration as there is an insufficient availability of oxygen to decompose organic matter. Lower BOD values in post-aerated waters indicate that aerobic decomposition of organic matter takes place (Astuti et al., 2014). It is possible that the aeration system balances the oxygen concentrations within the lake, by ensuring that the oxygen consumption by microbial decomposition of organic matter does not outweigh the oxygen production in the lake.

The concentration and distribution of dissolved oxygen in a lake is essential for a healthy lake system to maintain a good nutrient and oxygen balance (Hasan et al., 2014). Photosynthesis, driven by irradiance, and respiration can control DO variability day-to-day within lake systems (Langman et al., 2010), however DO in lakes often shows complex patterns suggesting that concentrations are often controlled by multiple mechanisms from biological and physical processes, as well as management processes such as aeration (Bella, 1970, Langman et al., 2010). The use of aeration is not only for increasing DO levels in a lake, it can also be used to limit phytoplankton growth and minimise phytoplankton concentrations as well as preventing gas escaping from benthal deposits (Hasan et al., 2014). For effective management of Cardiff Bay, all the benefits of the aeration system need to be assessed to ensure that management methods are correctly implemented. The influence that the phytoplankton population in Cardiff Bay has on lake DO concentrations can be found in Chapter 4, Section 4.5.4 and lake sediment process are discussed in Chapter 5, Section 5.5.7.

3.6.8 The Photic Depth of Cardiff Bay

The photic depth of Cardiff Bay did not show any seasonal variation between April 2012 and April 2014. Photic zone measurements often reflect the turbidity of the water column, and therefore display seasonal variation as a result of changes in river discharge, wind and phytoplankton abundance (James et al., 2004, Zhang et al., 2006). The presence of seasonal variation in River Taff and River Ely discharge but the lack of photic zone variation suggests that the turbidity of the water column was not primarily controlled by river discharge. However, river discharge is not the only source of suspended and dissolved particulate matter in a lake which causes turbidity within the water column; terrestrial erosion can enter the system by runoff, and phytoplankton are a natural source that cause turbidity (Minnesota Pollution Control Agency, 2008).

Sediment resuspension is dependent on the concentration of particulate and dissolved matter within in the water column and the extent of wind mixing. Shallow lakes like Cardiff Bay are often vulnerable to sediment resuspension which increases the total suspended solids (TSS) in the water column and decreases photic depth (James et al., 2004, Zhang et al., 2006). The presence of the aeration system in Cardiff Bay and the short flushing rate created by the River Taff and Ely inputs are likely to affect the light attenuation of the water column through sediment resuspension.

Site C, situated in the marina, had a significantly deeper photic depth than site 4. Site C was situated on the East shore of the lake next to a jetty and during sampling macrophytes were observed. The presence of macrophytes can prevent sediment resuspension, allowing deeper light attenuation (Acharyya et al., 2012). In contrast, site 4 was situated at the mouth of the River Taff and experienced lower photic depths. The water column here was only 2-3 m deep and turbidity caused by the river discharge rates will have caused sediments to become suspended in the water column, reducing the photic depth.

Phytoplankton growth and photosynthesis rates are often related to the ratio of photic depth to mixing; defined as either the water column depth or the surface layer depth in a stratified water column (Alpine & Cloern, 1988). High concentrations of turbidity can light-limit

phytoplankton photosynthesis and productivity, restricting biomass growth (Dokulil, 1994). Understanding the relationship of light availability and the phytoplankton community within Cardiff Bay can be found in Chapter 4 and is essential for the implementation of effective management in Cardiff Bay to ensure a healthy ecosystem.

3.7 Conclusions

The environmental variables and physical properties of Cardiff Bay were determined for the period between January 2003 and April 2014. Lake management strategies based on continuous scientific data covering inter-annual and seasonal variation are arguably the most resilient (De Hoyos & Comín, 1999), therefore understanding the temporal and spatial variations of the environmental and physical properties of Cardiff Bay is critical when assessing the future management of the lake with regard to phytoplankton community structure and abundance and the impact it can have on water quality.

Using data collected by Cardiff Harbour Authority from January 2003 until April 2014, and additional data collected between April 2012 and April 2014, it was determined that the lake of Cardiff Bay was polymictic. The lake was well-mixed with no spatial differences in temperature or DO, and there was no thermocline present at any time. The combination of a fast lake flushing rate and relatively shallow depth, along with the presence of an aeration system, prevented the lake from stratifying. The water in the lake was being constantly mixed and replaced every few days.

Despite the absence of a thermocline in the lake, the Bay was still subject to natural seasonal changes. Seasonal variations in air temperature resulted in the seasonal succession of water temperature within the lake. Warmer temperatures experienced during summer have been predicted to increase in the future under IPCC climate scenarios and the occurrence of extreme precipitation events such as that experienced in the Bay in July 2007, are also predicted to increase. This suggests that Cardiff Bay will be affected by climatic changes in the future. The months of summer will be critical for determining processes within the lake as the combination of higher temperatures and a reduction in precipitation will result in a decrease in river discharge and a slowing of the lake flushing rate. To maintain the current lake water quality, management policies will need to consider the impacts of these climatic changes on the lake

phytoplankton community, and nutrient availability. These relationships are discussed further in Chapter 4 and Chapter 5, respectively.

The presence of the aeration system within Cardiff Bay appeared to aid the mixing of the lake body but did not influence the river system with respect to DO concentrations, where differences at depth were observed. The use of the aeration system in the lake will be required in the future to counteract the degenerative effects of a warmer system and a slower flushing rate. DO concentrations in the lake must be maintained at 5 mg L^{-1} and despite the water column being oversaturated with oxygen 33% of the time; the presence of seasonal variation suggests that extreme changes in climate will have the ability to negatively influence DO concentrations, therefore, to prevent any degeneration in DO, the aeration system will be required. At present the aeration system is balancing the removal of DO at depth with the DO being produced in the lake, however, with changes in climate occurring, this balance also has the potential to break down.

Cardiff Bay has a high flushing rate as a result of river input. If precipitation changes in the future as predicted by the IPCC, variations in river input will occur as a result, ultimately influencing and altering the flushing rate of the lake. To ensure that these changes do not negatively impact the lake of Cardiff Bay, potential extreme weather events and alterations in lake flushing rate need to be considered within management plans. The flushing rate of the lake has the potential to be the pivotal characteristic controlling the health of the water body by controlling the flushing loss of phytoplankton biomass and nutrients (Elliott et al, 1999); these relationships are discussed in detail in Chapters 4 and 5.

Due to the absence of data, the direct impact that the Cardiff Bay aeration system had on the lake was not able to be studied, therefore the comparison of natural and artificial mixing within the lake was not analysed. If data allows in the future, studying this relationship would provide an insight into the exact role that the aeration system plays in Cardiff Bay and the extent to which it will be required in the future. The calculation of lake flushing rate was limited by the restriction of water input and output data; evaporation and transpiration rates not being used; and the role of wind induced mixing not being assessed. Despite this, the presence of the barrage to maintain lake water levels, and the artificial mixing caused by the aeration system,

allowed sufficiently accurate lake flushing rates to be calculated and the influence rates had on the lake system to be investigated.

This research has combined and interpreted a valuable 11-year data set to provide an understanding of how temperatures and precipitation vary temporally; over longer (annual) and shorter (monthly) timescales in Cardiff Bay. Current seasonal variability and the potential for change in the future has been discussed and it is essential that it is considered for the successful management of this lake, as well as to provide examples for the management of other artificial temperate lakes. The data collected will be integral to the management strategy used by CHA and has demonstrated how the use of an aeration system in a polymictic lake can still allow for natural seasonal variations to take place, without compromising lake water quality.

THE PHYTOPLANKTON COMMUNITY OF CARDIFF BAY

4.1 Summary

The phytoplankton community structure of the artificial lake of Cardiff Bay, South Wales was analysed between January 2003 and April 2014 using chlorophyll *a* as a proxy for biomass, variable chlorophyll fluorescence as a proxy for productivity, and species identification. The state of the phytoplankton population and the impact of environmental drivers in the lake were determined. The phytoplankton community was found to be healthy, but low in biomass and dominated by diatoms. Cyanobacteria were present in insignificant concentrations and do not threaten lake water quality at present. Phytoplankton was not responsible for variations in dissolved oxygen in the lake and productivity was not light limited. The polymictic nature of the lake prevented any spatial or vertical differences of phytoplankton biomass from occurring but concentrations were found to be lower at the river study sites compared to those in the main body of the lake. The phytoplankton biomass of the lake was low and prevented the formation of strong relationships with environmental factors. However, there is the potential for these relationships to strengthen in the future, particularly in response to changes in climate.

4.2 Introduction

4.2.1 The Importance of Phytoplankton

Phytoplankton provide the base of a lake food-web, they convert solar energy to chemical energy and store it as organic matter, providing energy to the entire lake trophic system (Peeters et al., 2007, Kumari et al., 2018). Phytoplankton community structure and abundance are often driven by a blend of nutrient supply processes and environmental variables, and in turn this alters the structure of higher trophic levels and hence the biodiversity and conservation quality of a lake. This “conservation quality” is also intrinsically linked with water quality, determining

a balance between algal dominated systems, susceptible to algal blooms, and clear-water macrophyte dominated systems following classical bi-stable lake theory (Scheffer et al., 2001). Hence, phytoplankton dynamics within a lake are often linked to fluctuations of water temperature, water column mixing and nutrient availability (Kumari et al., 2018).

4.2.2 Seasonal Variability in Phytoplankton

The phytoplankton community of a lake increases in complexity from winter to summer, in the annual plankton succession (Peeters et al., 2007). During spring, phytoplankton growth often increases abruptly, determining the start of phytoplankton succession for the year (Peeters et al., 2007). The typical decline in phytoplankton experienced during autumn-winter is often a result of a reduction in solar radiation and an increase in vertical mixing (Talling, 1971).

4.2.3 Temperature & Phytoplankton

Temperature changes, particularly increasing temperatures, have the ability to alter the phenology of dominant phytoplankton within a lake. For example, an increase in temperature can cause spring blooming species to peak earlier in the year (Elliott et al., 2006). Increasing temperatures also favour cyanobacteria, causing this group of phytoplankton to dominate phytoplankton biomass, particularly during summer and often leading to harmful algal blooms (HABs) and associated water quality issues (Elliott et al., 2006).

It has been predicted that in a future warmer climate, the onset of the spring phytoplankton bloom will occur earlier than it does at present due to the combined result of higher air temperatures and low wind speeds (Peeters et al., 2007). There is also a likely potential for longer summer algal blooms increasing costs of maintaining lake use and water quality (Sanseverino et al., 2016). Hence understanding this process is key for the management of lakes and reservoirs, including systems such as the heavily managed Cardiff Bay.

4.2.4 Precipitation and Phytoplankton

High levels of precipitation can reduce lake stratification by mixing the water column alongside increasing the concentration of DO (Kumari et al., 2018). Conversely lower levels of summer precipitation, as predicted by the IPCC, may result in reduced mixing, particularly within shallower lakes and reservoirs at the time when cyanobacteria blooms are most likely. Such conditions greatly favour cyanobacteria and hence also raise poor water quality risk.

4.2.5 River Discharge & Phytoplankton

Partially linked with precipitation, river discharge is a major integrating factor related to important process affecting phytoplankton communities. For example, it affects the rate of turbulent mixing, nutrient supply, and light attenuation as a result of suspended matter (Descy, 1993). It is also possible for discharge to control phytoplankton growth through dilution as significant phytoplankton populations can only be created when their net growth rate exceeds the dilution rate (Descy, 1993). This is effectively synonymous to a laboratory chemostat system, with wash out proportional to flow rate. In general, fast-growing, disturbance-tolerant phytoplankton are favoured in high discharge areas (Descy, 1993). In warm summer conditions, this is also likely to promote cyanobacteria productivity and biomass. However, there are also reports where cyanobacteria dominated during periods of low river discharge (Descy, 1993) due to the shallow conditions within a lake, combined with potential stratification, leading to nutrient supply (P and N) that give the cyanobacteria a competitive advantage over other phytoplankton groups.

Phytoplankton production is often low in environments with high turbidity, due to light limitation, particularly those caused by river inputs of suspended particulate matter and/or resuspension of bottom sediments, as light is rapidly attenuated in the water column, reducing the photic zone (Cloern, 1987). In a lake with high river discharge inputs, a higher sediment loading may therefore occur and control phytoplankton biomass by increasing light limitation, whilst at the same time, the greater river discharge can dilute and/or wash out phytoplankton biomass.

The influence of river discharge on phytoplankton assemblage often decreases during summer as lake conditions stabilise (Descy, 1993).

4.2.6 Flushing Rate & Phytoplankton

A fast flushing rate often results in constant losses due to displacement (Muylaert et al., 2000). As explained above, this is the chemostat theory whereby an external inflow rate and combined outflow (if proportional to the inflow when lake volume is high) lead to simple out-flushing of cells at a faster rate than can be balanced by production. In such situations, independent of nutrient supply and/or potential nutrient limitation, phytoplankton biomass removal leads to a low lake production overall with low phytoplankton biomass. Such systems can often favour macrophyte growth and hence, so long as turbidity is low, result in a clear-water system supporting a macrophyte dominated bi-stable state (Scheffer et al., 2001).

4.2.7 Stratification & Phytoplankton

Spatial and temporal variability in phytoplankton biomass and productivity can often be explained by variations in photic depth and water column mixing (Cloern, 1987). As a result, phytoplankton blooms often coincide with stratification events due to the reduction in mixing of the surface layer (Cloern, 1987). The onset of phytoplankton growth is predominantly determined by mixing conditions which affect the exposure of phytoplankton to light (Peeters et al., 2007). Stratification may also result in lower oxygen potential in deeper hypolimnetic waters due to increased BOD. In such conditions anaerobic processes may lead to internal loading of nutrients, such as phosphorus and hence increase phytoplankton productivity during lake-turnover or periods of localised mixing (Perkins & Underwood, 2001b). The balance between mixing depth control of nutrient supply and light availability is crucial in determining phytoplankton productivity. However as explained above, if flushing rate is high due to high river discharge into the lake and high outflow, then a) stratification is unlikely due to hydrodynamic mixing, and b) out-flushing of phytoplankton may be greater than phytoplankton productivity. This demonstrates the need to understand lake hydrology, environmental variables and nutrient dynamics in lake management.

4.3.8 Management & Phytoplankton

Understanding the multiple factors which drive the ecological changes in phytoplankton communities under current climatic changes is a fundamental challenge of aquatic ecology but is required for the implementation of effective lake management (Alpine & Cloern, 1992, Kumari et al., 2018). In Cardiff Bay, the focus of this study, this is of paramount importance due to the system being a totally artificial impounded freshwater system that supports a high investment in housing and associated infrastructure attracted to the Bay area. Management must ensure a clean water system suitable for recreation and aesthetic exploitation, and thus the prevention of algal blooms is essential. As described above, reduced summer rainfall and higher summer temperatures increase the risk of a decline in water quality if phytoplankton productivity increases to drive HABs. Hence determining the roles of nutrient limitation, environmental variables and flushing rate currently providing a relatively clean-water system is highly important for risk assessment and climate change resilience management.

4.3 Hypotheses

To determine the drivers of the phytoplankton community structure and biomass of Cardiff Bay the following thesis objective will be addressed within this chapter:

The phytoplankton within the lake will be assessed using chlorophyll-*a* as a proxy for biomass, variable chlorophyll fluorescence as a proxy for productivity, and species identification.

Using data collected between January 2003 and April 2014 in Cardiff Bay, the following hypotheses have been tested:

The phytoplankton biomass, productivity and assemblage within the lake does not vary spatially or with depth within the Bay.

There will be seasonal variation in phytoplankton biomass, productivity and assemblage within the lake.

Phytoplankton biomass is controlled in part by flushing rate, such that increases in flushing rate reduces biomass due to out-flushing, and low flushing rate reduces biomass out-flushing.

4.4 Materials and Methods

The formally tidal, artificial freshwater lake of Cardiff Bay was formed in 2001 as a result of a barrage being built across the Taff-Ely estuary. Even before the barrage was built, it was recognised that there were high environmental risks in creating the lake, therefore it is managed by Cardiff Harbour Authority (CHA), with a focus on preventing the formation of toxic phytoplankton blooms (HABs). The features of Cardiff Bay are introduced in Chapter 2.

4.4.1 Data Processing

Cardiff Harbour Authority (CHA) collected bbe (biological. biophysical. engineering. Moldaenke FluoroProbe) data weekly from January 2003 to April 2014; a spectrofluorometer was used to determine the biomass of chlorophytes, cyanobacteria, cryptomonads, diatoms and total phytoplankton using chlorophyll *a* at c1 m depth at sites 4, 5, 6, 9, 10 and 17 (see Chapter 2 for site locations). Mean annual, seasonal and monthly whole-Bay phytoplankton biomass was calculated using all data collected at c1m during each time scale. Individual site means were calculated for all sites using all data collected at c1 m on an annual, seasonal and monthly basis from January 2003 until April 2014. Total phytoplankton biomass refers to the sum of biomass for chlorophytes, cyanobacteria, cryptomonads and diatoms, represented by chlorophyll-*a* concentrations.

Throughout the 11-year study period, the sites for phytoplankton data collection altered. Data were collected from sites 4, 6 and 10 from January 2003 until September 2010. Site 5 data were collected from August 2004 until April 2014. Site 9 data were collected from November 2007 until April 2014. Site 17 was the only site where phytoplankton biomass data were collected throughout the whole study; January 2003 until April 2014. Due to the nature of the data, interpolation of missing data was not suitable, therefore any gaps in data were not filled in. All graphs contain standard deviation error bars.

Water samples were collected once a fortnight at sites 4, 5, 9, 10, A, B, C and D within the lake (see Chapter 2 for locations) from April 2012 until April 2014 from the surface (c1 m), middle and bottom (c1 m from lake sediment) of the water column at each site. Fluorescence measurements were made using a Walz Water (Walz GmbH Germany) PAM in the field (Calculations in Chapter 2, Section 2.2.2.1), to determine the photosynthetic activity of the phytoplankton at every site and depth, as well as samples taken for chlorophyll-*a*, *b* and *c* concentrations. Samples taken at the surface, middle and bottom of the water column were merged at each site for phytoplankton identification to take place. Mean seasonal data was calculated for each site individually and mean whole-Bay data were calculated using all data collected at each time scale.

The data processing that took place for air temperature, water temperature, precipitation, dissolved oxygen, photic depth, river discharge and flushing rate data used within this chapter can be found in Chapter 3.

4.4.2 Statistical Analysis

Mean monthly values for all data sets were used individually to conduct the decomposition of additive time series in ‘Relevance’ (R) to analyse the seasonal variation in phytoplankton biomass, the script used can be found in Chapter 3, Section 3.4.2.

Principle Component Analysis (PCA) was used to determine the difference in phytoplankton assemblage between sites and was carried out using ‘PAleontological STatistics’ (PAST) (Hammer et al., 2001).

Pearson’s correlation was undertaken using R to determine significant relationships between phytoplankton (biomass, abundance and productivity), and the environmental factors of the lake (air temperature, water temperature, precipitation, river discharge, lake flushing rate, dissolved oxygen and photic depth).

Nested Analysis of Variance (ANOVA) was carried out using MINITAB statistical software to determine if there was any variation in data on a seasonal or monthly basis, or throughout the water column. Single-factor ANOVA was used on data sets shown to be significant in

nested ANOVA to identify more specifically where differences lie using *post hoc* Tukey test ($\alpha = 0.05$) in MINITAB.

Monthly and seasonal data were used for linear regression analysis using R to determine if there was a “cause and effect” relationship between the physical variables of Cardiff Bay and the phytoplankton population. Following significant linear regression results, multiple regressions using R were undertaken using backward elimination to refine the “cause and effect” relationships.

4.5 Results

4.5.1 The Spatial & Vertical Distribution of Phytoplankton in Cardiff Bay

Using bbe (biological. biophysical. engineering. Moldaenke FluoroProbe that uses chlorophyll-*a* to determine phytoplankton groups) data, between January 2003 and August 2004, the phytoplankton assemblage at site 17 was different to that at sites 4, 6 and 10 (Figure 4.1, Table 4.1). The latter sites were dominated by diatoms and chlorophytes, with a low assemblage of cyanobacteria at site 4. The overall chlorophyll-*a* concentrations were lower at site 17 and it was dominated by diatoms. Between September 2004 and November 2007, the phytoplankton assemblage at sites 4 and 17 were similar, with lower levels of chlorophyll-*a* comprised mostly of diatoms. Larger concentrations of diatoms and chlorophytes, along with some cryptophytes were found at sites 5 and 10. Site 6 was different to the other sites as chlorophyll-*a* levels were much lower (Figure 4.2, Table 4.1). Chlorophyll-*a* concentrations were low at sites 6 and 17 between December 2007 and September 2010 making them different to the other sites in the lake, but also different to each other as site 6 contained more diatoms than site 17, and site 17 contained more chlorophytes than site 6 (Figure 4.3, Table 4.1). Site 9 was significantly different to sites 5 and 17 between October 2010 and April 2014 with low levels of chlorophyll-*a* (Figure 4.4, Table 4.1), the phytoplankton assemblage at site 5 and 17 were dominated by diatoms and chlorophytes.

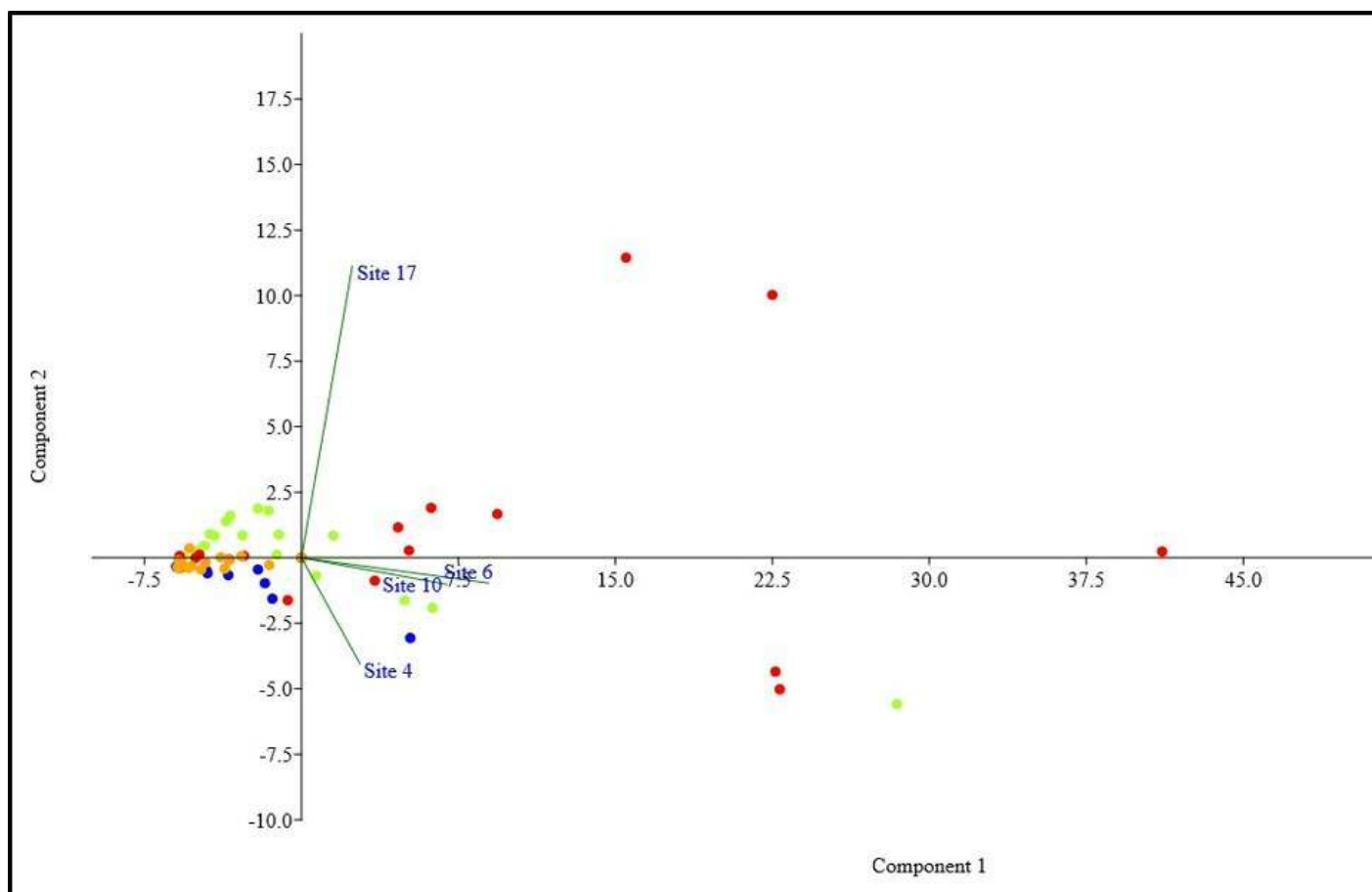


Figure 4.1: Principle Component Analysis of the phytoplankton assemblage between January 2003 and August 2004
cyanobacteria, green = chlorophytes, orange = cryptophytes, red = diatoms.

September 2004 – November 2007	4	$r = 0.20, n = 156, P < 0.05$	$r = 0.58, n = 156, P < 0.001$
	5	$r = 0.55, n = 156, P < 0.001$	
	6	$r = 0.57, n = 156, P < 0.001$	$r = 0.39, n = 156, P < 0.001$
	10	$r = 0.56, n = 156, P < 0.001$	
	17		$r = 0.84, n = 156, P < 0.001$
December 2007 – September 2010	4	$r = 0.85, n = 136, P < 0.001$	
	5	$r = 1.0, n = 136, P < 0.001$	
	6	$r = 0.97, n = 136, P < 0.001$	
	9	$r = 0.99, n = 136, P < 0.001$	
	10	$r = 0.99, n = 136, P < 0.001$	
	17	$r = 0.83, n = 136, P < 0.001$	$r = 0.56, n = 136, P < 0.001$
October 2010 – April 2014	5	$r = 0.20, n = 172, P < 0.001$	$r = 0.94, n = 172, P < 0.001$
	9	$r = 1.0, n = 172, P < 0.001$	
	17		$r = 0.30, n = 172, P < 0.001$

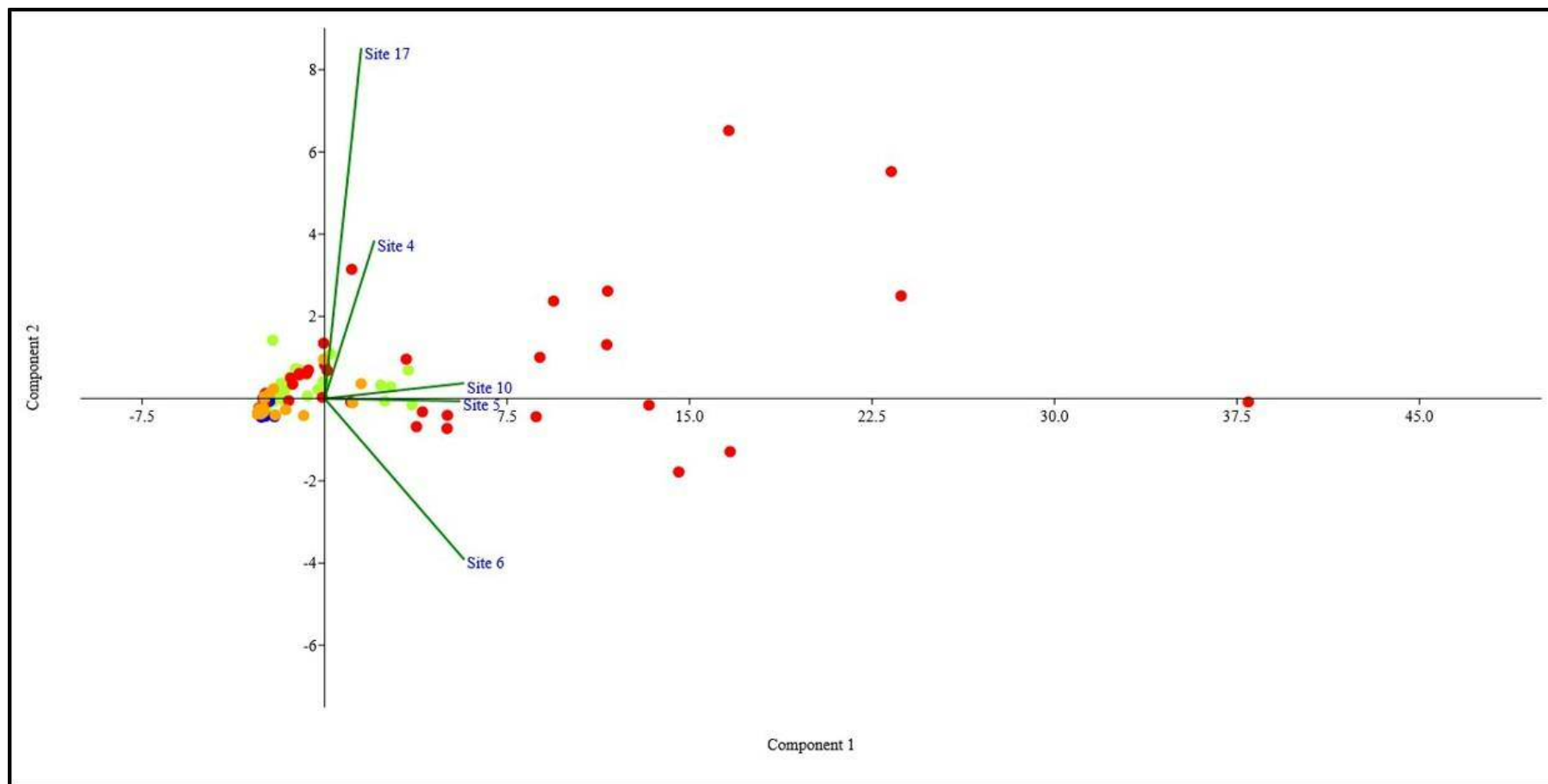


Figure 4.2: Principle Component Analysis of the phytoplankton assemblage between September 2004 and November 2007 in Cardiff Bay. Data points: blue = cyanobacteria, green = chlorophytes, orange = cryptophytes, red = diatoms

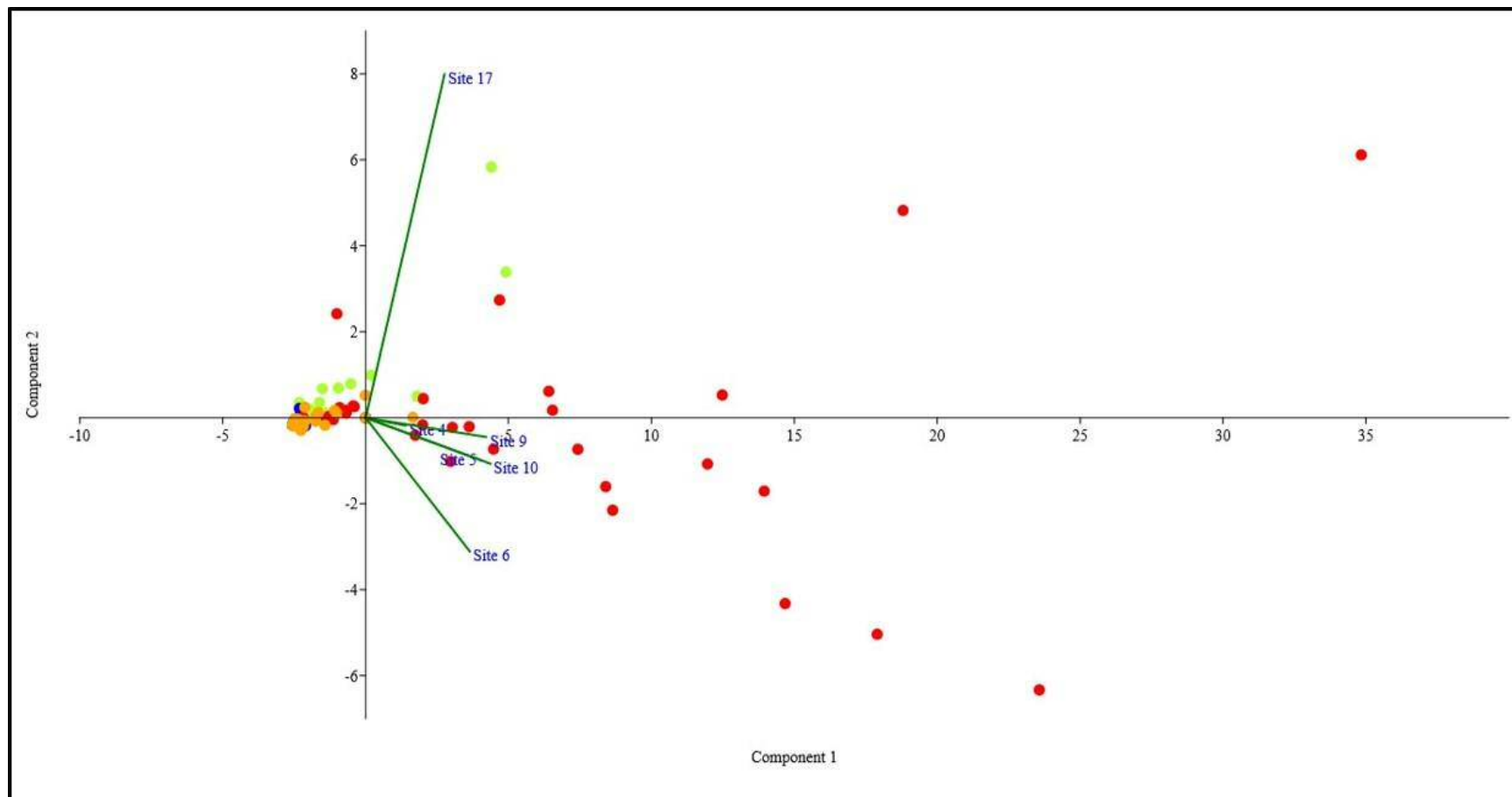


Figure 4.3: Principle Component Analysis of the phytoplankton assemblage between December 2007 and September 2010 in Cardiff Bay. Data points: blue = cyanobacteria, green = chlorophytes, orange = cryptophytes, red = diatoms.

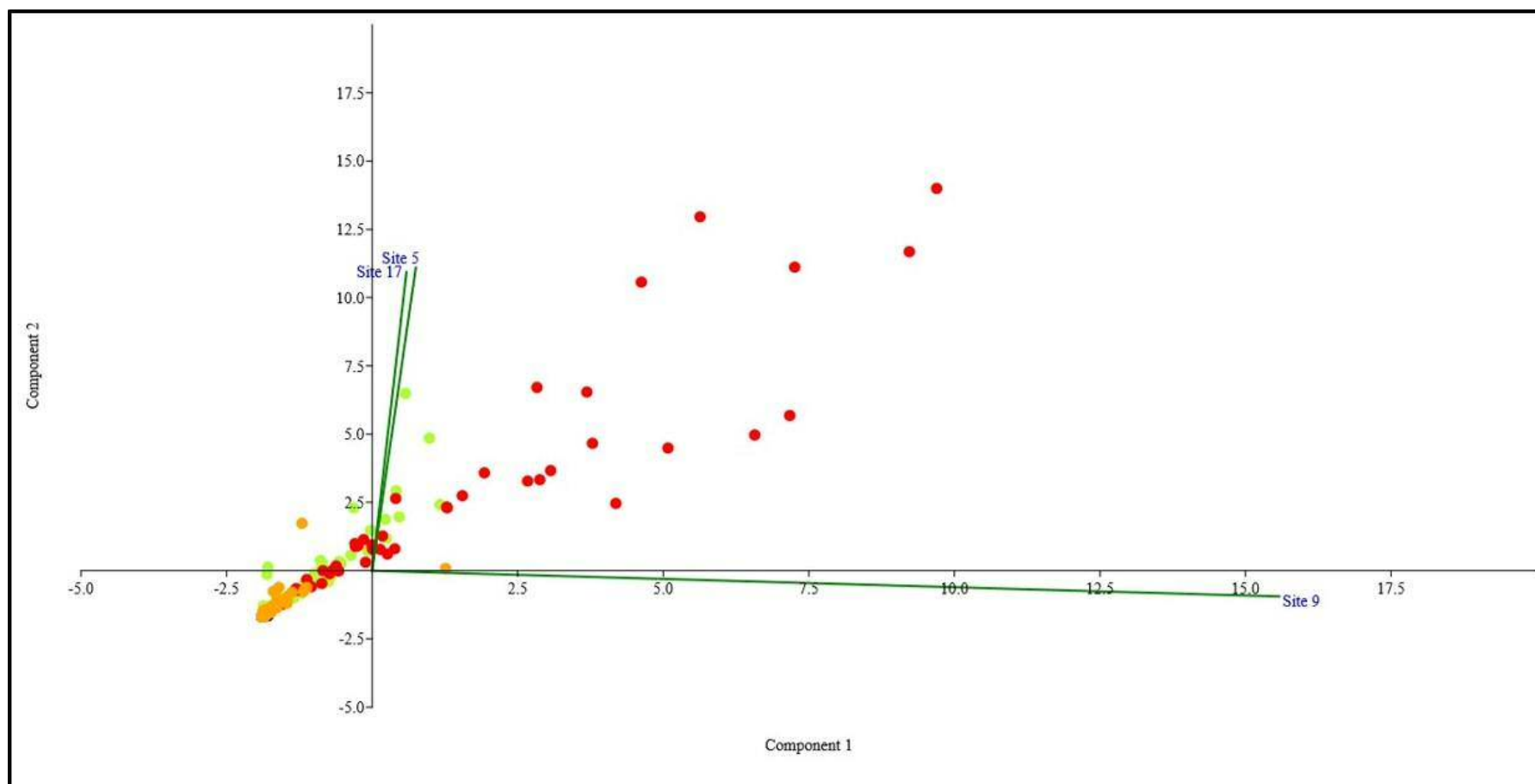


Figure 4.4: Principle Component Analysis of the phytoplankton assemblage between October 2010 and April 2014 in Cardiff Bay. Data points: blue = cyanobacteria, green = chlorophytes, orange = cryptophytes, red = diatoms.

Phytoplankton productivity was measured using proxy measurements obtained with variable chlorophyll fluorescence rapid light curves. All light curve parameters showed no significant difference between site, depth or between seasons (due to the relatively large variation in data indicated by sample standard errors, Table 4.2). Values of maximum light use efficiency (F_v/F_m) indicated a healthy actively photosynthetic community of phytoplankton, with an average for the whole data set of 0.52 ± 0.01 for the whole sample size of 223 rapid light curves (Table 4.2). This was supported by the relative electron transport rate ($rETR_{max}$ rel. units), light use coefficient (α , rel. units) and light saturation coefficient (E_k , $\mu\text{mol PAR m}^{-2} \text{ s}^{-1}$) with comparatively high values for all three parameters.

Table 4.2 Fluorescence data obtained for sites 4, 5, 9, 10, A, B, C and D in top (T), middle (M) and bottom (B) water samples in Cardiff Bay. Maximum light use efficiency (F_v/F_m), maximum relative electron transport rate ($rETR_{max}$ re. units), light use coefficient (α , rel. units) and light saturation coefficient (E_k , $\mu\text{mol PAR m}^{-2} \text{ s}^{-1}$) are shown with standard errors and sample sizes. Also shown are the average values with minimum and maximum values for the whole data set. Only light curves with significant curve fitting coefficients (at $p < 0.01$) were accepted for further analysis, with non-saturating light curves also excluded due to errors in calculating coefficients.

SITE and Depth	SAMPLE SIZE	F_v/F_m	$rETR_{max}$	A	E_k	F_v/F_m se	$rETR_{max}$ se	α se	E_k se
10B	2	0.61	262	0.30	912	0.037	7	0.059	204
10M	14	0.59	169	0.21	971	0.03	18	0.03	117
10T	22	0.52	171	0.21	837	0.02	9	0.01	48
4B	0	ND	ND	ND	ND	ND	ND	ND	ND
4M	9	0.46	140	0.21	815	0.03	24	0.02	191
4T	1	0.31	161	0.16	1014	n/a	n/a	n/a	n/a
5B	0	ND	ND	ND	ND	ND	ND	ND	ND
5M	10	0.47	238	0.19	1210	0.04	64	0.03	141
5T	18	0.47	196	0.21	970	0.02	36	0.02	109
9B	2	0.45	195	0.22	1067	0.03	51	0.07	565
9M	9	0.48	144	0.17	901	0.03	15	0.02	69
9T	20	0.53	187	0.18	1112	0.03	31	0.02	114
AB	0	ND	ND	ND	ND	ND	ND	ND	ND
AM	19	0.56	183	0.18	1065	0.03	11	0.01	70
AT	16	0.49	173	0.20	965	0.03	10	0.02	85
BB	0	ND	ND	ND	ND	ND	ND	ND	ND
BM	12	0.63	214	0.20	966	0.03	53	0.02	183
BT	14	0.51	209	0.18	1325	0.04	27	0.02	202
CB	0	ND	ND	ND	ND	ND	ND	ND	ND
CM	14	0.50	176	0.21	929	0.03	27	0.02	155
CT	5	0.54	204	0.29	967	0.05	63	0.12	199
DB	1	0.57	292	0.17	1750	n/a	n/a	n/a	n/a
DM	14	0.50	158	0.15	1463	0.03	12	0.02	440
DT	21	0.54	192	0.17	1248	0.02	20	0.01	222
Average	223	0.52	188	0.20	1067	0.01	7	0.01	45
Min	0	0.31	140	0.15	815				
Max	22	0.63	292	0.30	1750				

4.5.2 The Phytoplankton Community Structure of Cardiff Bay

Phytoplankton counts taken between April 2012 and April 2014 determined that the dominant phytoplankton group at each site within Cardiff Bay was chlorophytes. Except at site 4, where diatoms were dominant (Figure 4.5). Cyanobacteria were the least dominant group of phytoplankton, except at site B, where dinoflagellates were the least dominant phytoplankton group (Figure 4.5). The dominant species of diatom observed within Cardiff Bay were *Cocconeis*, *Navicula* and *Synedra*, the dominant species of chlorophyte were *Sphaerocystix*, *Chorella* and *Ulothrix* whilst the dominant species of dinoflagellates identified were *Cryptomonas*, *Phacus* and *Rhodomanas* (Table 4.3).

Mean monthly whole-Bay surface cryptomonad chlorophyll *a* displayed positive correlation with mean monthly whole-Bay surface chlorophyll *a*, diatoms, chlorophytes and cyanobacteria. However, despite these relationships being statistically significant, they were not strong, with $r^2 = <0.5$ (Appendix Figures 4.1, 4.2, 4.3 and 4.4).

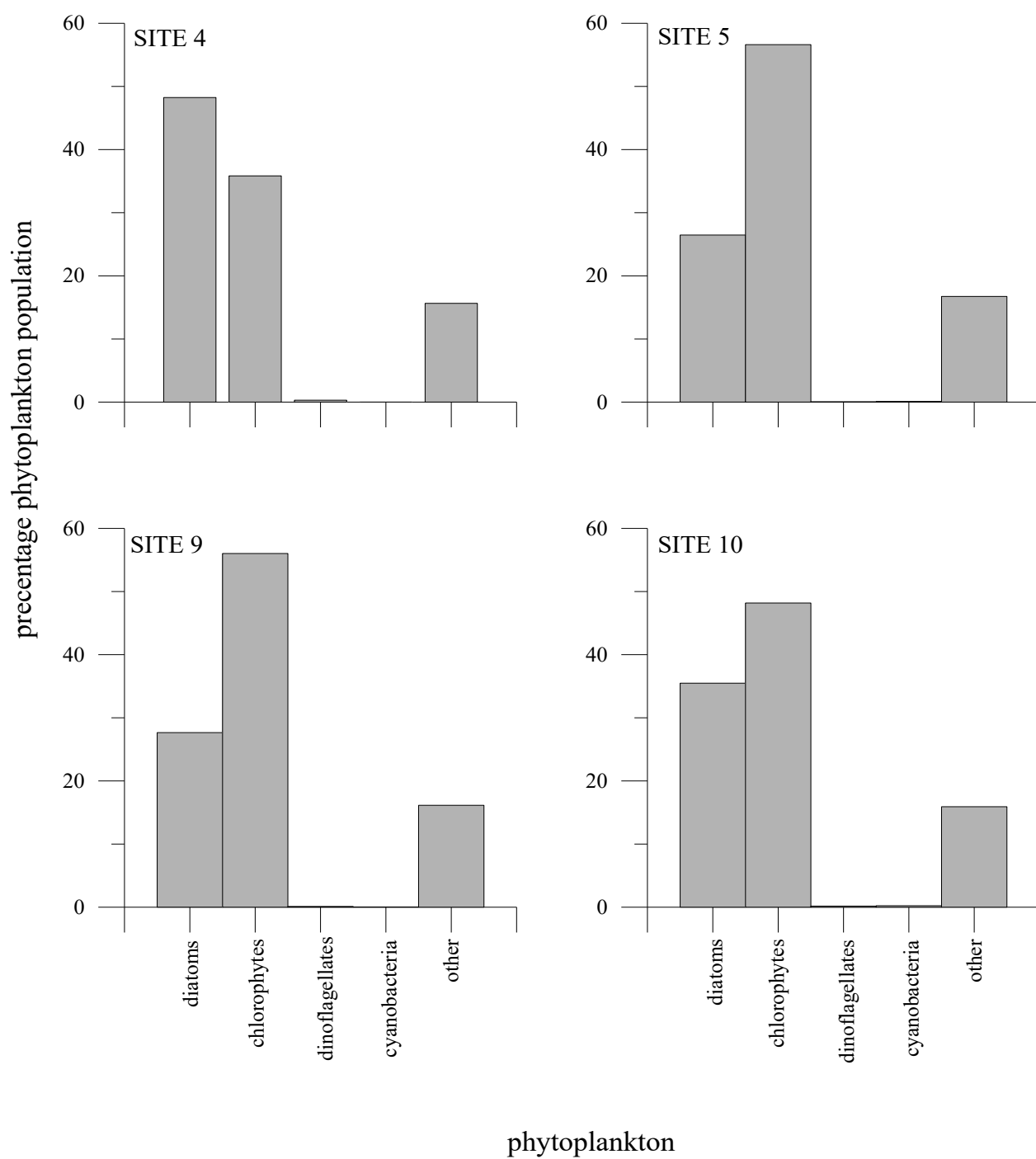


Figure 4.5: Percentage phytoplankton population composition at 8 survey

Table 4.3: Phytoplankton species identified in Cardiff Bay between April 2012 and April 2014, in declining order of abundance.

Diatoms	Chlorophytes	Cyanobacteria & Others
Cocconeis	Sphaerocystix	Oscillatoria
Navicula	Chorella	Nostoc
Synedra	Ulothrix	Anabena
Frustulia	Euglena	Gomphosphaeria
Diatoma	Tetraedron	Microcystis
Nitzschia	Chlamydomonas	Ceratium
Melosira	Trachelomonas	Peridinium
Synedra	Dictyosphaerium	
Neidium	Ankistrodesmus	
Rhoicospenia	Scenedesmus	
Stephanodiscus	Trochiscia	
Tabellaria	Pediastrum	
Gomphonema	Closterium	
Cymbella	Stigeoclonium	
Meridion	Actinastrum	
Fragillaria	Staurastrum	
Calonesis	Cryptomonas	
	Rhodomonas	
	Phacus	

4.5.3 Seasonal Variations of Phytoplankton in Cardiff Bay

Whole-Bay mean seasonal total chlorophyll-*a* concentrations (from bbe data) indicated that phytoplankton biomass peaked during spring (Figure 4.6). The spring peak was mostly comprised of diatoms, which accounted for between 74% (2013) and 97% (2009) of the phytoplankton biomass between 2003 and 2014 (Figure 4.7). Diatom biomass peaked during spring, except for during 2006, 2007 and 2012, where diatom biomass reached their maximum during summer (Figure 4.8). The rest of the community in spring was composed of

chlorophytes and cryptomonads. Cyanobacteria biomass was at its lowest during this season (Figure 4.7).

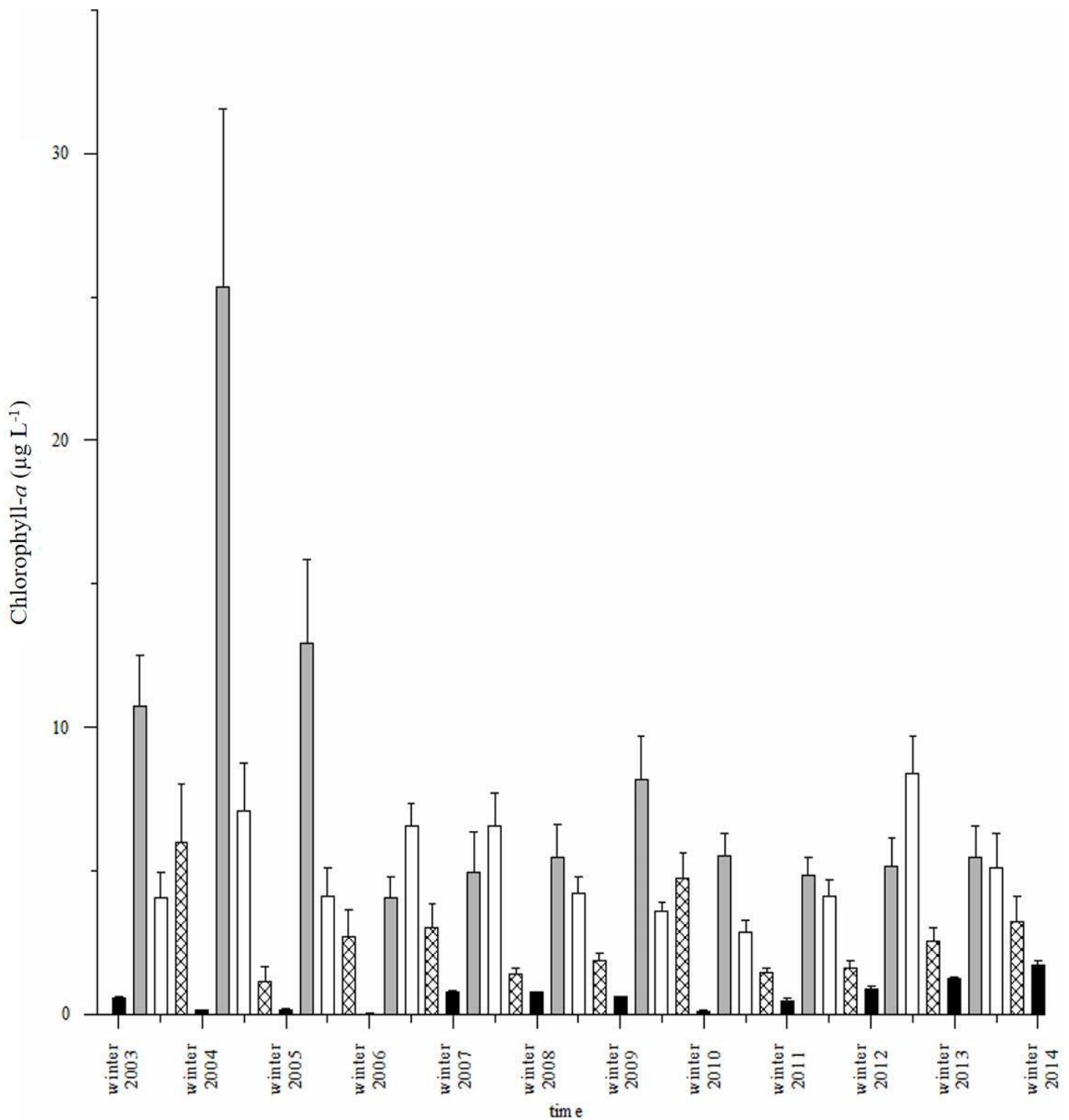


Figure 4.6: Mean seasonal whole-Bay total chlorophyll *a* in Cardiff Bay from 2003 until 2014. winter: black bars, spring: grey bars, summer: white bars, autumn: hatched bars (standard error bars).

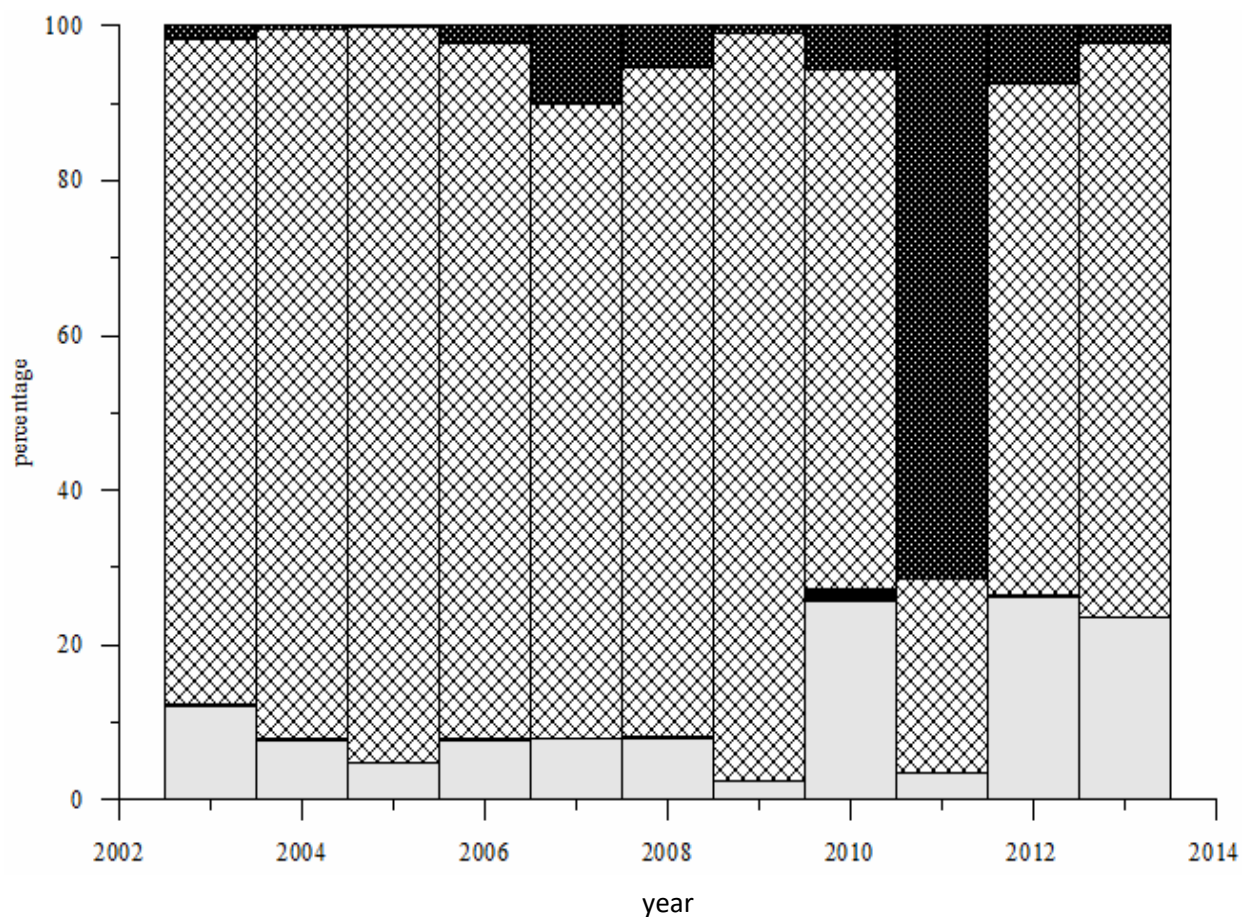


Figure 4.7: Whole-Bay spring percentage chlorophyll a phytoplankton community composition in Cardiff Bay between 2003 and 2014. Grey bar: chlorophytes, black bar: cyanobacteria, hashed bar: diatoms and black-spotted bar: cryptomonads.

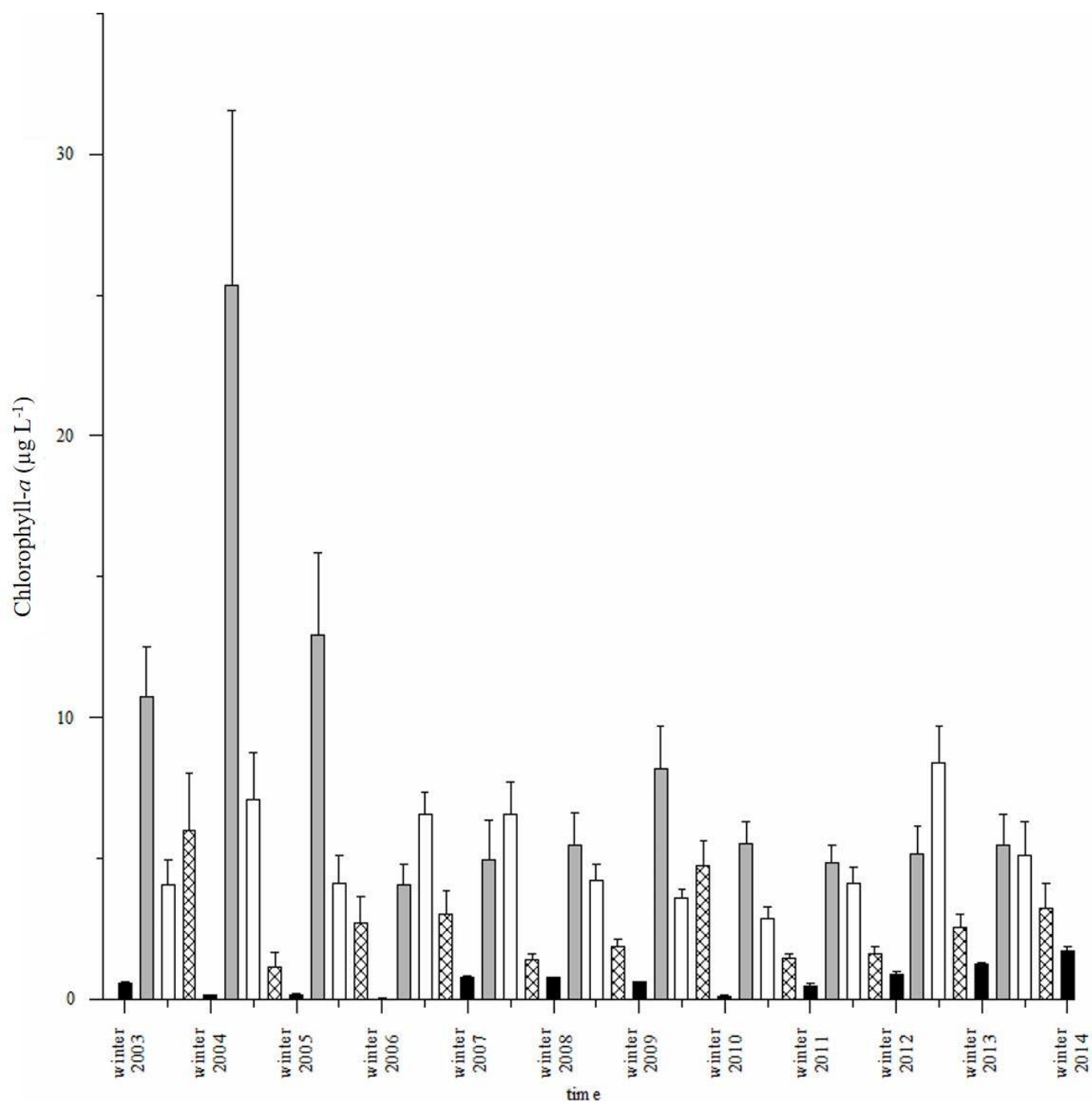


Figure 4.8: Mean seasonal whole-Bay diatom chlorophyll *a* in Cardiff Bay from 2003 until 2014. winter: black bars, spring: grey bars, summer: white bars, autumn: hatched bars (standard error bars).

During summer, phytoplankton biomass remained high and, like spring, was mainly composed of diatoms, with chlorophytes and cryptomonads making up the rest of the community (Figure 4.9). Chlorophyte biomass peaked during summer (Figure 4.10). The seasonal succession of cryptomonads varied between 2003 and 2014, where maxima was experienced during spring, summer and autumn, but during 5 out of 11 years it occurred in summer (Figure 4.11).

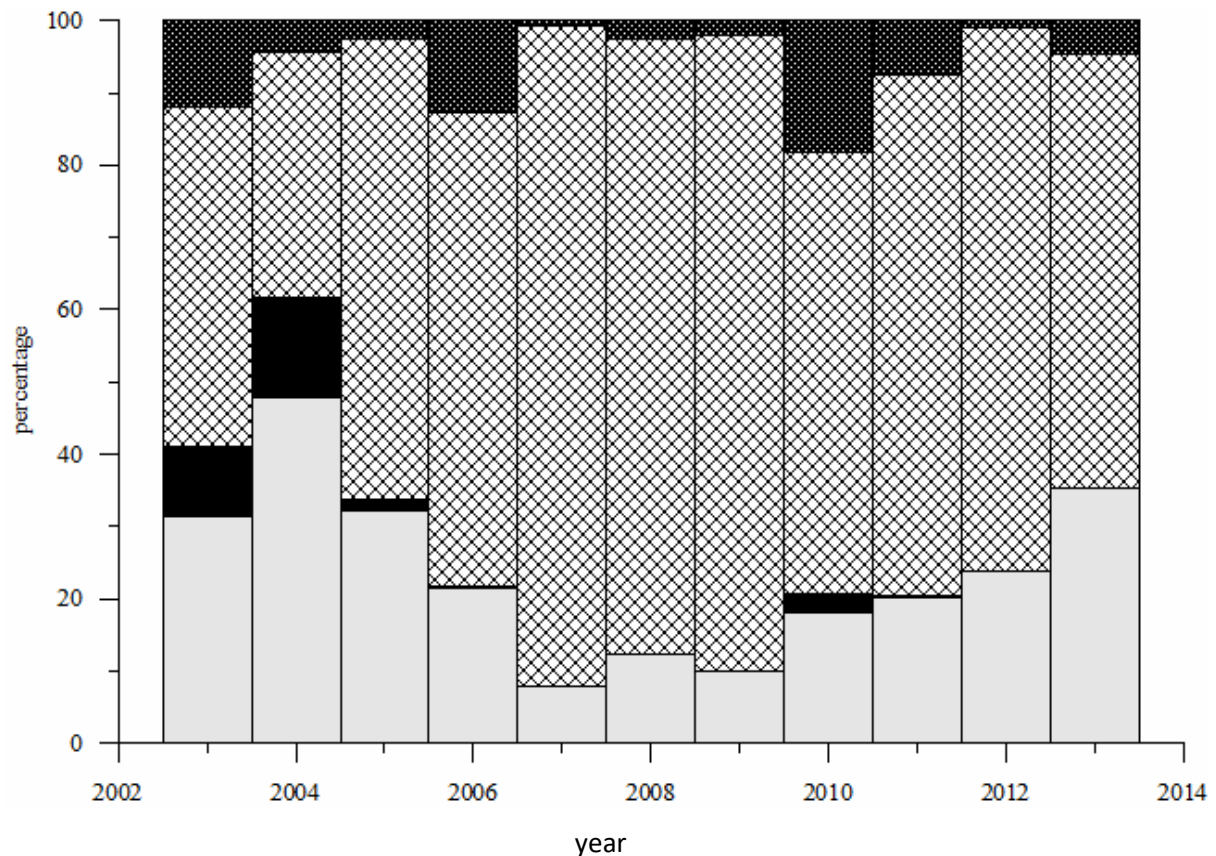


Figure 4.9: Whole-Bay summer percentage chlorophyll *a* phytoplankton community composition in Cardiff Bay between 2003 and 2014. Grey bar: chlorophytes, black bar: cyanobacteria, hashed bar: diatoms and black-spotted bar: cryptomonads.

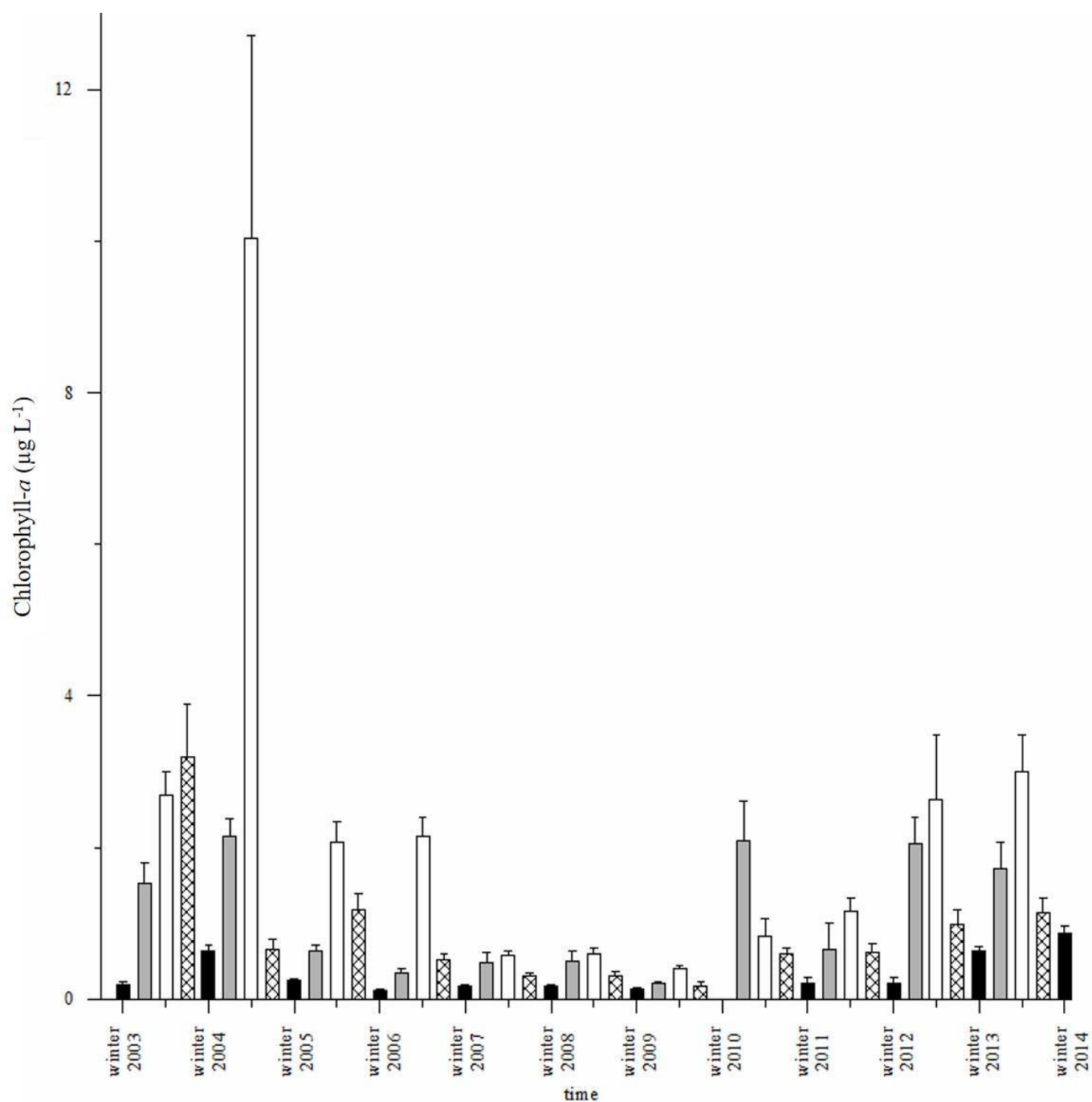


Figure 4.10: Mean seasonal whole-Bay chlorophyte chlorophyll *a* in Cardiff Bay from 2003 until 2014. winter: black bars, spring: grey bars, summer: white bars, autumn: hatched bars (standard error bars).

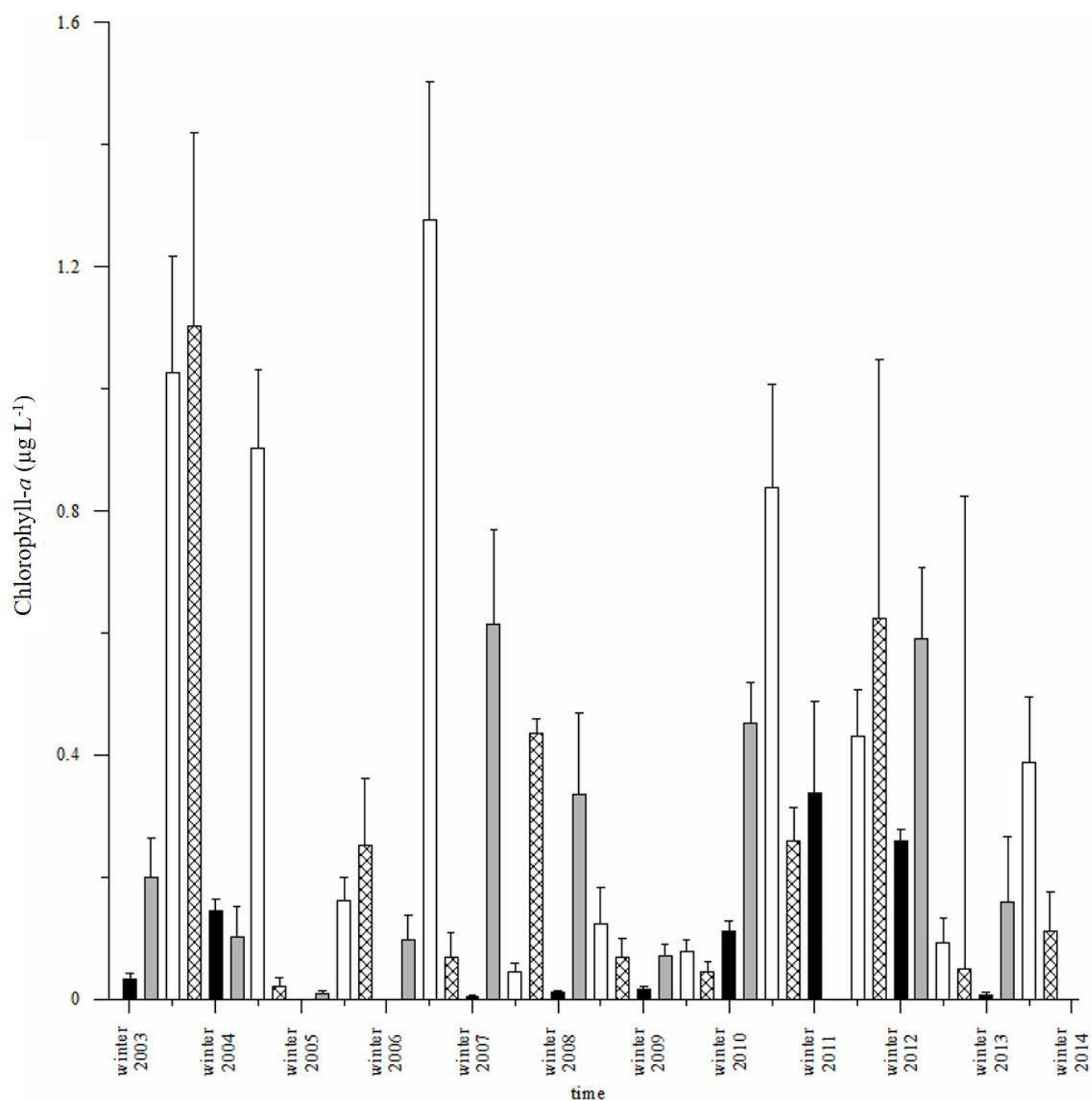


Figure 4.11: Mean seasonal whole-Bay cryptomonad chlorophyll a in Cardiff Bay from 2003 until 2014.
 winter: black bars, spring: grey bars, summer: white bars, autumn: hatched bars (standard error bars).

The phytoplankton biomass during autumn was mainly composed of diatoms, followed by chlorophytes and cryptomonads, with a small percentage of cyanobacteria (Figure 4.12). During winter, the phytoplankton biomass in Cardiff Bay was at its lowest (Figure 4.6). Diatom, chlorophyte and cryptomonad biomass were all at their lowest during this season. Despite this, the phytoplankton community of Cardiff Bay was mainly composed of chlorophytes and diatoms during winter (Figure 4.13). Although since 2007 diatoms have made up the majority (between 35% in 2010 and 80% in 2008) of the winter phytoplankton community, winter was the season that chlorophytes and cyanobacteria reached their highest contribution to the phytoplankton population. Cyanobacteria chlorophyll-*a* concentrations were very low between 2003 and 2014 and demonstrated no obvious seasonal succession (Figure 4.14).

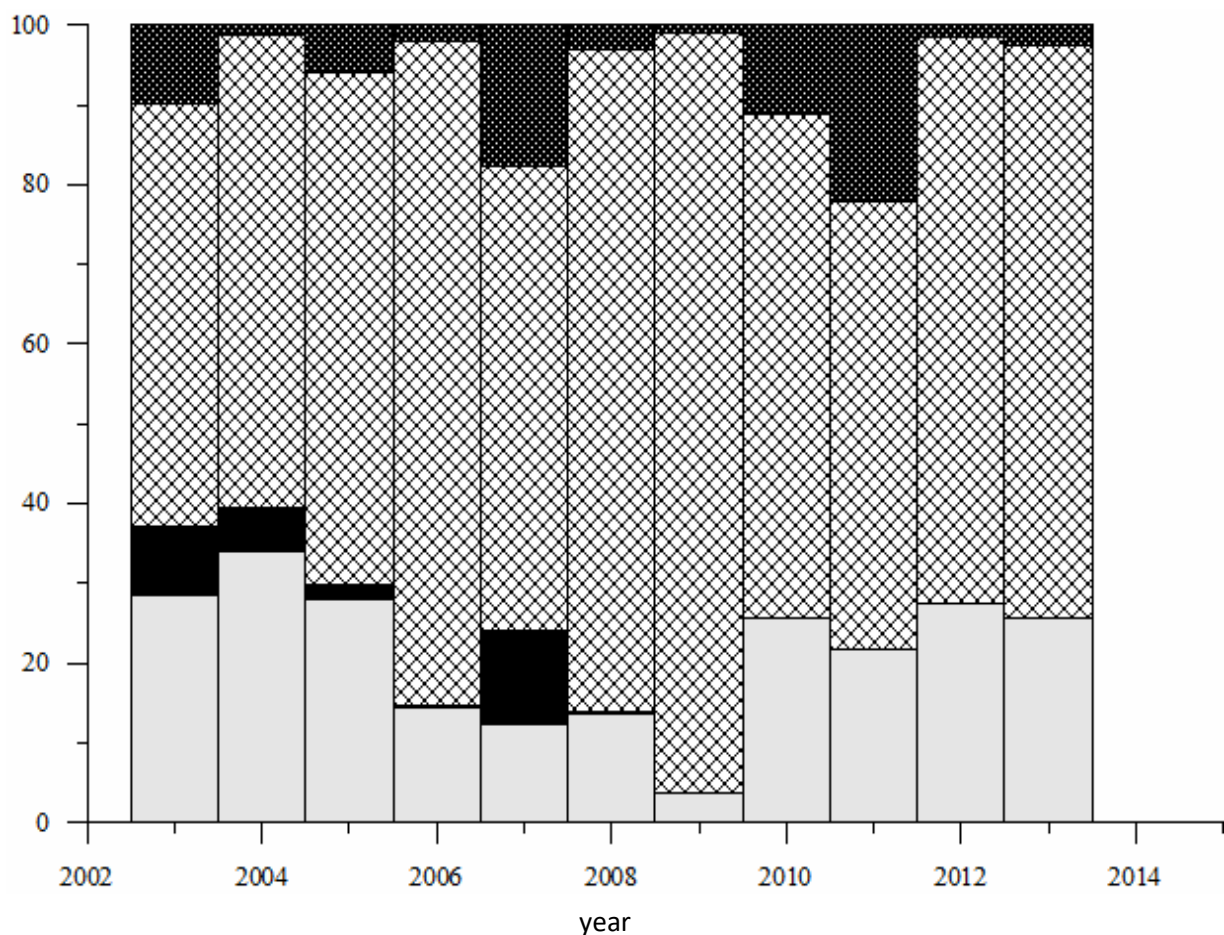


Figure 4.12: Whole-Bay autumn percentage chlorophyll a phytoplankton community composition in Cardiff Bay between 2003 and 2014. Grey bar: chlorophytes, black bar: cyanobacteria, hashed bar: diatoms and black-spotted bar: cryptomonads.

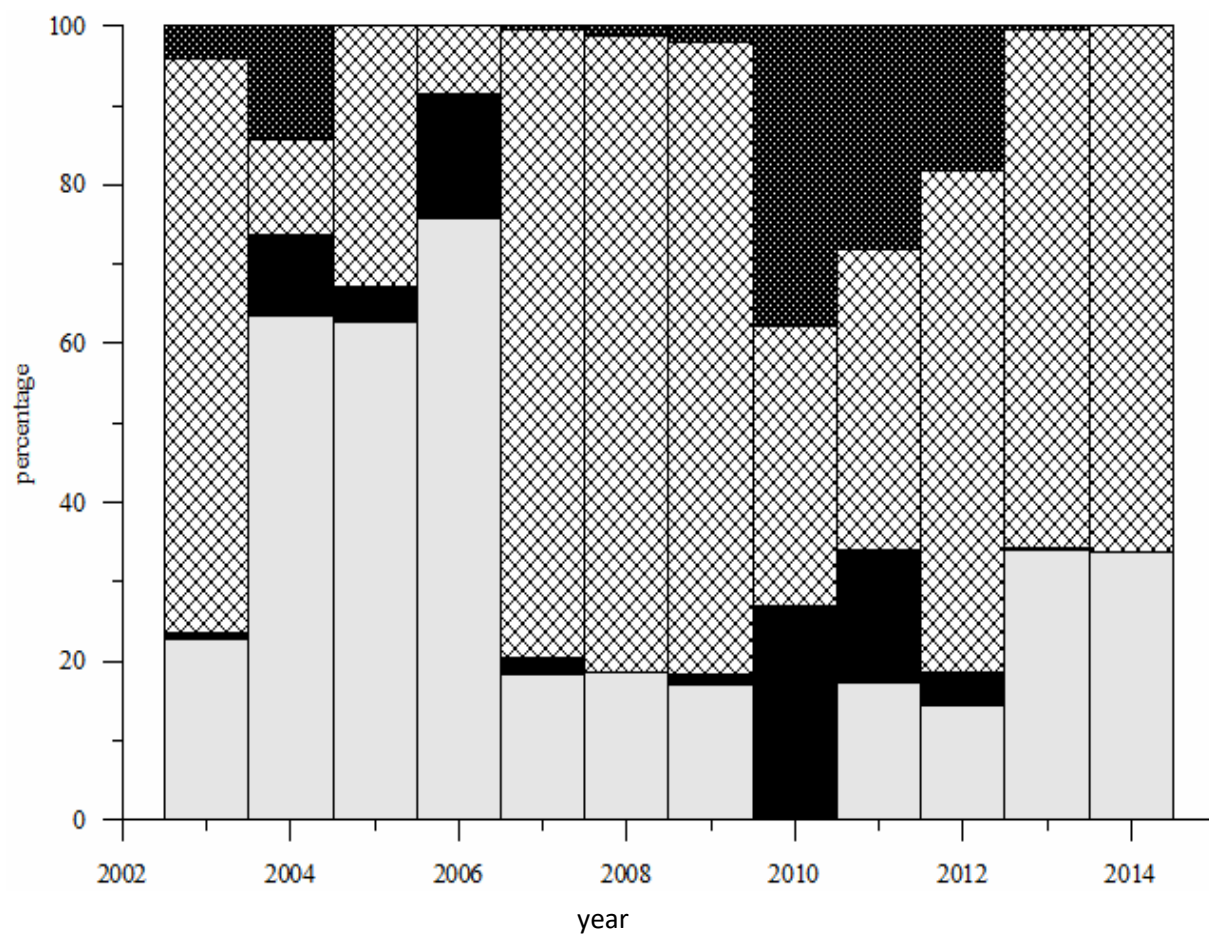


Figure 4.13: Whole-Bay winter percentage chlorophyll *a* phytoplankton community composition in Cardiff Bay between January 2003 and April 2014. Grey bar: chlorophytes, black bar: cyanobacteria, cross-hashed bar: diatoms and black-spotted bar: cryptomonads.

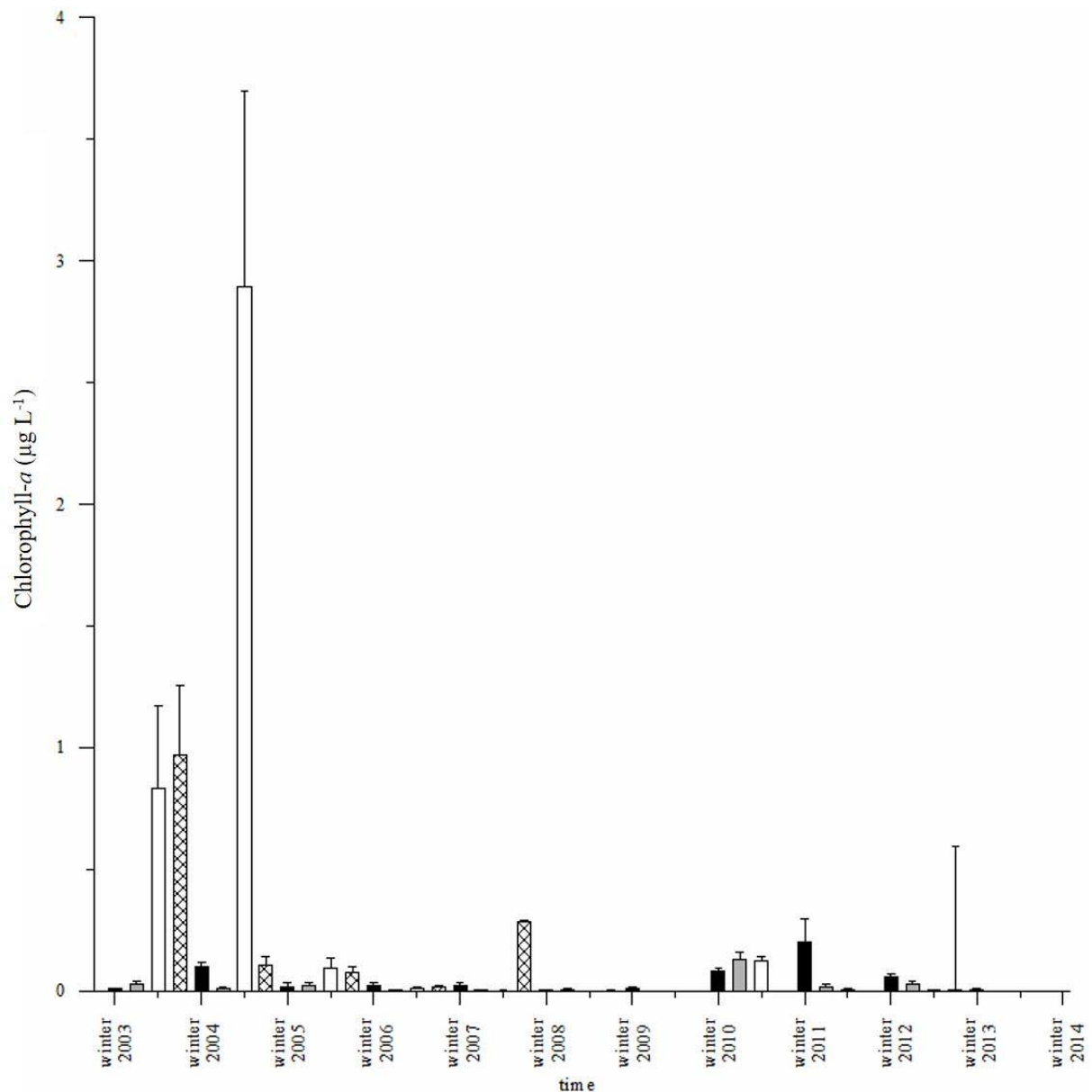


Figure 4.14: Mean seasonal whole-Bay cyanobacteria chlorophyll a in Cardiff Bay from 2003 until 2014. winter: black bars, spring: grey bars, summer: white bars, autumn: hatched bars (standard error bars).

There was a significant difference in mean whole-Bay surface chlorophyll *a* concentration between months ($F_{11,133} = 8.94$, $P < 0.01$). January and February chlorophyll *a* concentrations were significantly lower than those in April to August. March concentrations were significantly lower than those in April and May. April and May concentrations were significantly higher than those in October to December. July chlorophyll *a* was significantly higher than concentrations in December. August concentrations were significantly higher than those in November and December (Figure 4.15).

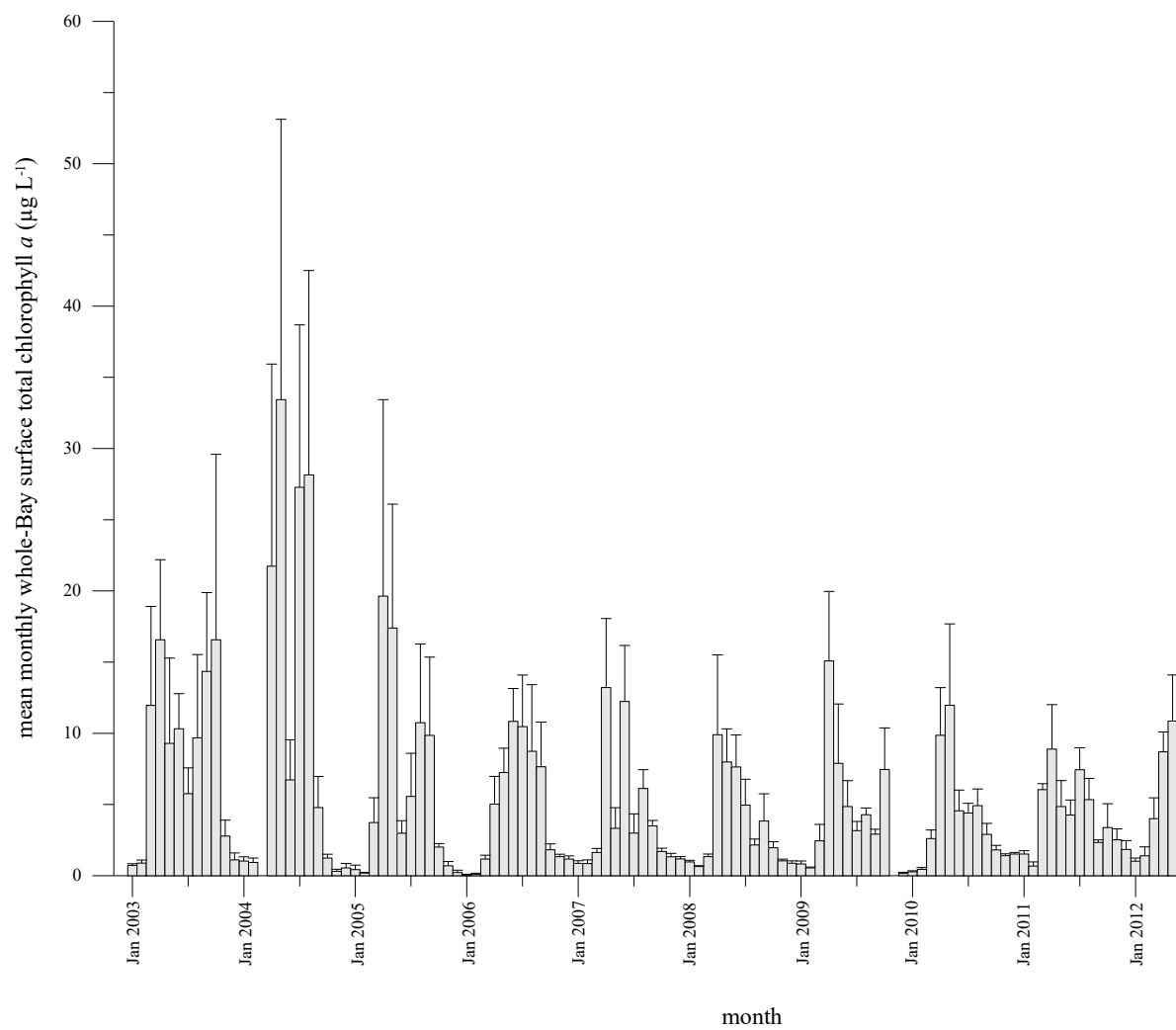


Figure 4.15: Mean monthly whole-Bay surface total chlorophyll *a* in Cardiff Bay from January 2003 until April 2012

4.5.4 The Relationship between Environmental Factors & Phytoplankton in Cardiff Bay

Mean autumn whole-Bay surface cryptomonad and cyanobacteria chlorophyll *a* was significantly negatively correlated with mean autumn lake flushing rate (Figure 4.16 and 4.17). Mean summer whole-Bay surface cryptomonad chlorophyll *a* was also significantly negatively correlated with mean summer lake flushing rate (Figure 4.18).

Mean monthly surface air temperature and mean monthly whole-Bay surface water temperature were positively correlated with mean monthly whole-Bay surface chlorophyll *a* but were not strong (Appendix Figure 4.5 and 4.6). Mean monthly precipitation was significantly negatively correlated with mean monthly surface whole-Bay chlorophyll *a* and mean monthly surface whole-Bay diatom, chlorophyte, cryptomonad and cyanobacteria chlorophyll *a*, but not strongly (Appendix Figure 4.7, 4.8, 4.9, 4.10 and 4.11).

There was a significant negative correlation between mean monthly whole-Bay total chlorophyll *a* and mean monthly River Taff discharge (Appendix Figure 4.12), River Ely discharge (Appendix Figure 4.13), and combined river discharge (Appendix Figure 4.14), however these relationships were weak.

Mean seasonal lake flushing rate positively correlated during autumn with mean seasonal whole-Bay surface chlorophyte chlorophyll *a*, and during spring with mean seasonal whole-Bay surface cryptomonad chlorophyll *a*, but the relationships were weak. Mean monthly lake flushing rate also significantly positively correlated with mean monthly whole-Bay surface total chlorophyll *a*, where the chlorophyll *a* level increased as the lake flushing rate slowed. However, this relationship was also weak.

There was no statistically significant annual relationship between mean annual whole-Bay surface total chlorophyll *a* and mean annual air temperature, mean annual whole-Bay surface water temperature, mean annual precipitation, mean annual River Taff, River Ely and combined river discharge, or mean annual lake flushing rate. There was also no statistically significant relationship between mean annual whole-Bay surface diatom, chlorophyte, cryptomonad and cyanobacteria chlorophyll *a* with mean annual air temperature, mean annual

whole-Bay surface water temperature, mean annual precipitation or mean annual lake flushing rate.

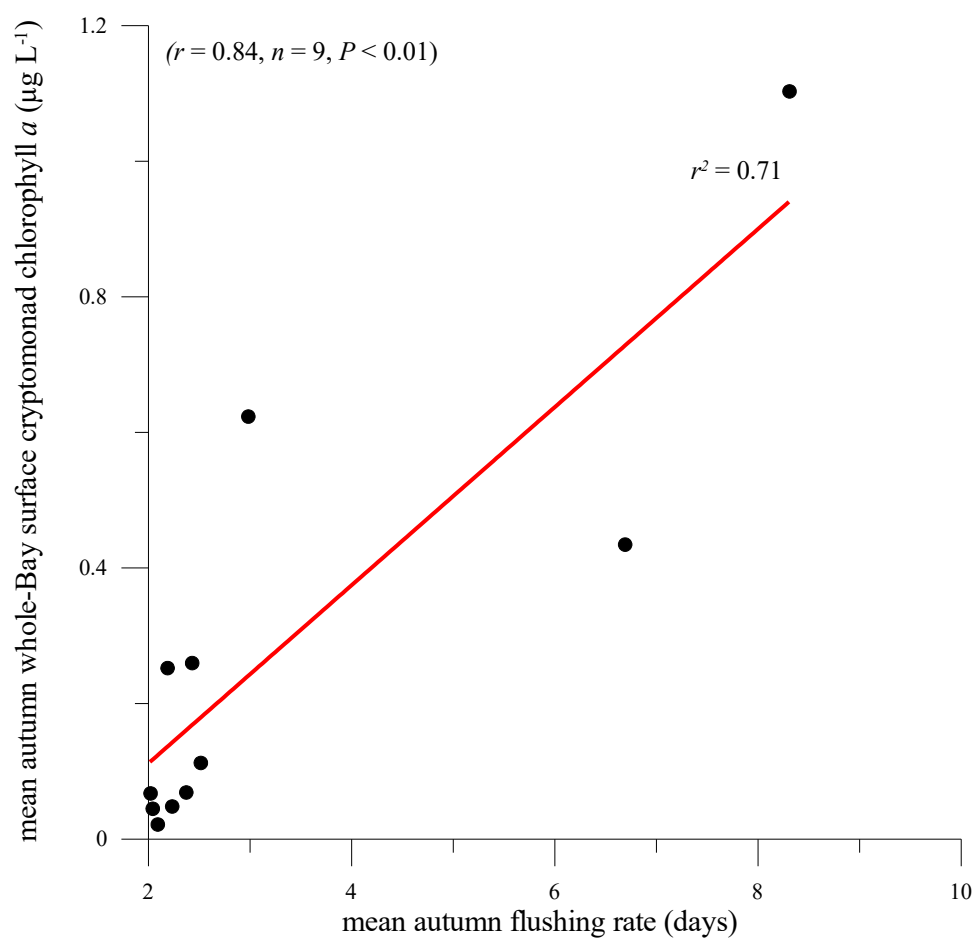
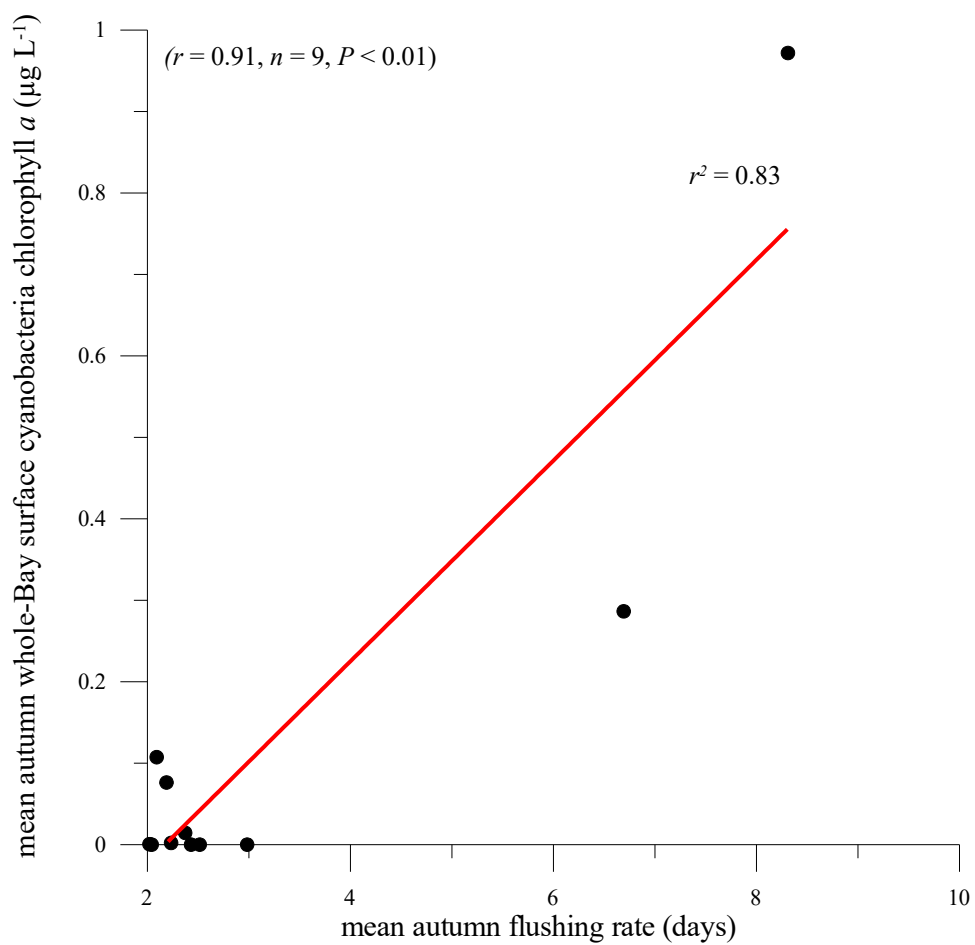


Figure 4.16: Mean autumn whole-Bay surface cryptomonad chlorophyll *a* and mean autumn lake flushing rate of Cardiff Bay between 2003 and 2013.



4.17: Mean autumn whole-Bay surface cyanobacteria chlorophyll *a* and mean autumn lake flushing rate of Cardiff Bay between 2003 and 2013.

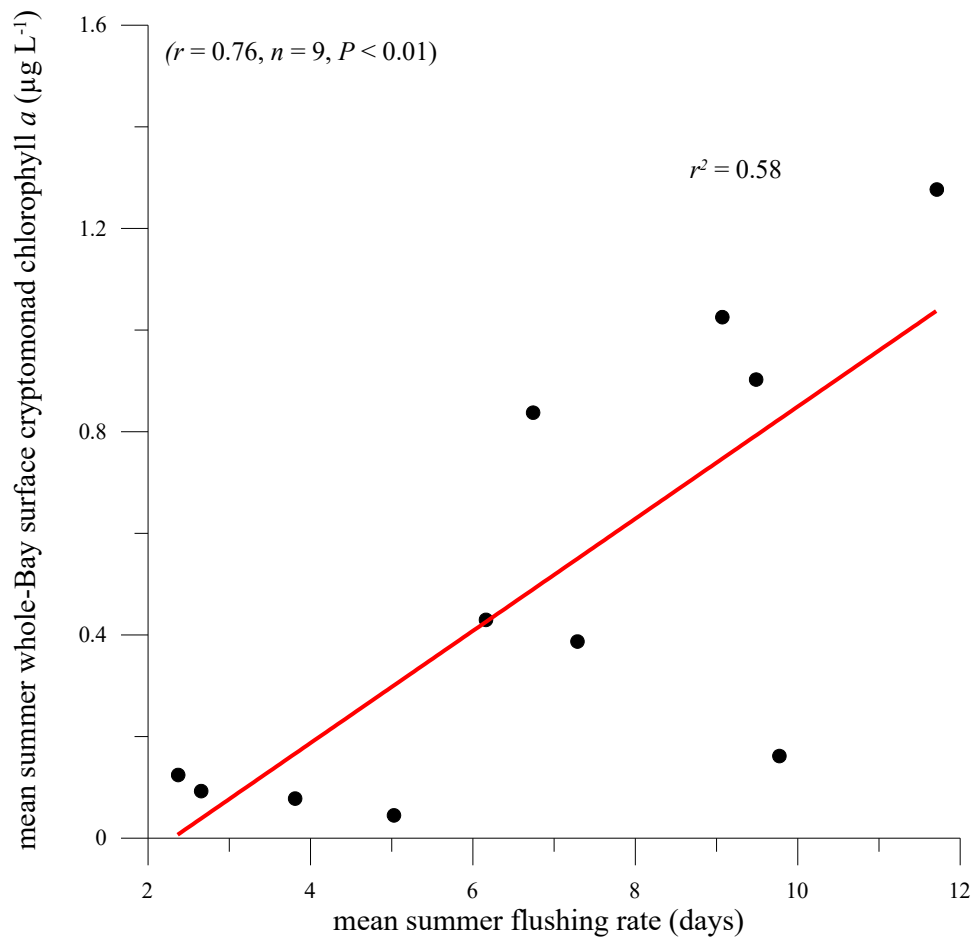


Figure 4.18: Mean summer whole-Bay surface cryptomonad chlorophyll *a* and mean summer lake flushing rate of Cardiff Bay between 2003 and 2013.

There were no statistically significant seasonal relationships during any season between mean seasonal whole-Bay surface diatom, chlorophyte, cryptomonad, cyanobacteria and total chlorophyll *a* with mean seasonal air temperature, mean seasonal whole-Bay surface water temperature, mean seasonal precipitation or mean seasonal River Taff, Ely or combined discharge.

Mean monthly air temperature, mean monthly whole-Bay surface water temperature (Appendix Figure 4.15, 4.16, 4.17 and 4.18), and mean monthly River Taff, Ely and combined discharge did not have a statistically significant relationship with mean monthly whole-Bay surface diatom, chlorophyte, cryptomonad or cyanobacteria chlorophyll *a*. The mean monthly flushing rate of Cardiff Bay did not have a statistically significant relationship with mean

monthly whole-Bay surface diatom, chlorophyte, cryptomonad, cyanobacteria or total chlorophyll *a*. Mean monthly surface total chlorophyll *a* correlated with River Taff and River Ely discharge at every site within the lake and rivers but all of the relationships were weak.

There was a weak statistically significant relationship between mean winter whole-Bay surface DO and mean winter whole-Bay surface total chlorophyll *a*. Mean monthly whole-Bay surface DO also displayed a weak statistically significant relationship with mean monthly whole-Bay surface diatom and cyanobacteria chlorophyll *a*. Annually, and during spring, summer and autumn there was no relationship between mean whole-Bay surface DO and mean whole-Bay surface diatom, chlorophyte, cryptomonad, cyanobacteria or total chlorophyll *a*. Mean monthly whole-Bay surface DO did not have a relationship with mean monthly whole-Bay surface chlorophyte or cryptomonad chlorophyll *a*.

Between 2012 and 2014 there was no relationship between mean whole-Bay surface diatom, chlorophyte, cryptomonad, cyanobacteria or total chlorophyll *a* and mean whole-Bay photic depth on an annual, seasonal and monthly basis.

4.6 Discussion

Cardiff Bay supported a healthy, productive community of phytoplankton, but biomass levels were low. This made it difficult for strong relationships to be established between phytoplankton and environmental factors in the lake. However, phytoplankton biomass did display a seasonal succession where levels peaked during spring and gradually decreased through summer and autumn, before being reaching their lowest levels in winter. Diatoms made up most of the phytoplankton population and were dominant during every season. Cyanobacteria displayed the lowest levels of biomass throughout the study and were often not present. There was no spatial or vertical variation in phytoplankton biomass with the main lake body, however sites situated in the River Taff and River Ely displayed lower phytoplankton biomass than the body of the lake. Phytoplankton were not found to influence DO variations and productivity was not light limited. Lake flushing rate controlled cryptomonad and cyanobacteria biomass during summer and autumn (respectively), no other relationships between phytoplankton biomass and environmental variables were found.

4.6.1 The Characteristics of Cardiff Bay

The analysis of the environmental variables affecting Cardiff Bay (reported in Chapter 3) found that the lake was subject to seasonal variations in air and water temperature. These seasonal changes in temperature have the potential to influence the timing and rate of phytoplankton growth within temperate lakes like Cardiff Bay. As temperatures increase during spring (March, April and May), diatom biomass often abruptly increases (Descy, 1993), determining the start of phytoplankton succession for the year (Peeters et al., 2007). Succeeding this, the warmer summer months of June, July and August usually encourage the increase of chlorophyte and cyanobacteria biomass (Descy, 1993). During summer Cardiff Bay air and water temperatures were at their warmest, providing the ideal conditions for phytoplankton biomass growth. However, during this season precipitation levels were at their lowest, causing a reduction in discharge from the River Taff and River Ely and therefore a slowing of lake flushing rate. Analysing the combination of these environmental variables is essential to understand potential changes within the phytoplankton community of a lake to ensure that phytoplankton productivity does not increase uncontrollably. Alterations in river discharge and lake flushing rates can influence the growth of phytoplankton through dilution. Significant phytoplankton populations can only be created when their net growth exceeds the dilution rate within the lake (Descy, 1993), this makes the management of river discharge and the lake flushing rate of Cardiff Bay essential, as during periods of low discharge and flushing; reducing dilution, lakes become susceptible to HABs (Mitrovic et al., 2003). These conditions are likely to occur in the future if the IPCC scenario that projected an increase in air temperature and decrease in precipitation will occur during summer take place (ref) is correct, the effects of which, the management of Cardiff Bay need to be prepared for.

Cardiff Bay does not stratify during any time of the year as a result of fast lake flushing rates and the constant mixing of the water column, these conditions prevent any spatial variation in the physical parameters of the lake from occurring, at the surface or with depth, within the lake, making it polymictic. As introduced in Chapter 1, stratification often occurs during spring and summer, as temperatures increase and river discharge decreases which often causes seasonal variation in phytoplankton dominance. Chapter 3 also found that the availability of light in the water column of Cardiff Bay does not vary spatially. Therefore, the absence of stratification and the impact this has on light availability in Cardiff Bay makes the assessment of the

phytoplankton community structure and its temporal variation paramount when making management recommendations as it is unlikely to respond to changes in climate like most other lakes.

The aeration system within Cardiff Bay is also likely to aid the mixing of the water column and prevent the onset of stratification within the lake. Its use is likely to be increased in the future to ensure that DO concentrations are maintained at 5 mg L^{-1} . However, during the timescale of this study, seasonal fluctuations still occurred within the lake despite the use of the system. Understanding the potential relationship between DO concentrations and the phytoplankton biomass within the lake is important when assessing the potential extent of use of the aeration system in the future.

4.6.2 The Phytoplankton Community of Cardiff Bay

The surface phytoplankton biomass (represented by chlorophyll *a* concentrations) in Cardiff Bay varied on a mean monthly basis of between 0.06 and $33.4 \text{ } \mu\text{g L}^{-1}$, a variation which is not uncommon in lakes due to the influence of seasonal changes in temperature, nutrient availability and mixing regimes (Maberly et al., 2011). Despite this fluctuation of phytoplankton biomass in the lake, Cardiff Bay experienced low levels of biomass throughout the year relative to other freshwater lakes. In the English Lake District, UK samples taken from wastewater in 2010 had a chlorophyll *a* concentration of $330 \text{ } \mu\text{g L}^{-1}$ and the waters of Blelham Tarn experienced $55,200 \text{ } \mu\text{g L}^{-1}$ during July 2010 (Maberly et al., 2011). As the chlorophyll *a* in Cardiff Bay was typically below $10 \text{ } \mu\text{g L}^{-1}$ between January 2003 and April 2014, the lake can be classed as mesotrophic, a low trophic status for a lake in lowland UK (Istvánovics, 2009).

Phytoplankton productivity data showed that the phytoplankton community of Cardiff Bay was healthy and highly productive with cells having high light accumulation ($\text{EK} \sim 1,000$). This indicates that the phytoplankton community are unlikely to be nutrient limited therefore another variable within the lake must be restricting the phytoplankton biomass.

Although the phytoplankton biomass in Cardiff Bay was low, there was still a phytoplankton group that dominated the lake. The polymictic nature of Cardiff Bay allowed diatoms to

dominate the phytoplankton community (Wilhelm & Adrian, 2008). Diatoms are one of the heavier groups of phytoplankton, therefore they benefit from a mixed water column which allows them to reach areas of the lake where growth is favoured (Patrick, 1948). The second most abundant phytoplankton in the lake were chlorophytes, this species is typical of a lake system as they can adapt to a broad range of environmental conditions (Salmaso, 2000). Cryptomonads and cyanobacteria were the least present phytoplankton group within Cardiff Bay. Cryptomonads prefer moderate turbulence and reduced grazing as they are 'high quality' food for herbivores and are readily digested by herbivores in a lake system (Barone & Naselli-Flores, 2003), therefore the conditions within Cardiff Bay may not be optimal for cryptomonad growth. Cyanobacteria often dominate in low N and P environments, when other phytoplankton do not thrive (Downing et al., 2001); the nutrient concentration and their relationship with phytoplankton in Cardiff Bay are discussed in Chapter 5.

4.6.3 The Distribution of Phytoplankton within Cardiff Bay

Phytoplankton biomass data calculated using bbe chlorophyll *a* data between January 2003 and April 2014 indicated that the phytoplankton biomass of Cardiff Bay varies spatially between the river and the main lake body. Site 4, situated in the mouth of the River Taff, and site 17, in the River Ely, both experienced a lower phytoplankton biomass than the sites situated within the main body of the lake. Although phytoplankton biomass was low in both the lake of Cardiff Bay and the rivers, the slower-moving water of the lake allowed the phytoplankton biomass to increase quicker than the phytoplankton present in the rivers.

Overall, analysis of the 11-year data-set suggested that there was little difference in phytoplankton biomass between sample points within the Bay (bbe data). This was supported by the fact that phytoplankton counts taken within the lake between 2012 and 2014 did not indicate any spatial difference in phytoplankton between sites within the lake. The exception to this was that the river sites (4 and 17) had higher biomass due to the impact of the river inflows, but once this water was within the Bay, the polymictic nature of the Bay largely removed any spatial variation and the Bay could be considered to have a well-mixed phytoplankton community. Small differences between some sample sites were treated with

caution due to the fact that the reliability of using a bbe FluoroProbe has been questioned (Kring et al., 2014).

4.6.4 Seasonal Succession of Phytoplankton within Cardiff Bay

The phytoplankton of Cardiff Bay displayed a typical seasonal succession where the highest biomass was present during spring and was maintained through summer, before a decline in autumn until the lowest biomass was reached in winter (Reynolds, 1980). However, whilst most lake phytoplankton communities experience succession from a spring diatom dominance, to chlorophytes, followed by cyanobacteria (Reynolds, 1980), Cardiff Bay did not. Diatoms were the dominant phytoplankton group during every season of the year. Diatoms did show seasonal succession though; diatom biomass was at its highest most commonly during spring, followed by summer. Diatom biomass then declined through autumn until reaching their lowest levels during winter. This is the normal seasonal succession for diatoms (Sommer, 1985).

Despite the overall low phytoplankton biomass within Cardiff Bay, chlorophytes displayed seasonal succession. Chlorophyte biomass increased during spring until reaching its peak in summer, this was followed by a decline during autumn before chlorophyte biomass reached its lowest levels during winter. The seasonal succession of Cryptomonad biomass varied between 2003 and 2014 in Cardiff Bay, the majority of cryptomonad biomass peaks occurred during summer, but also occurred during spring and autumn.

Cyanobacteria biomass in Cardiff Bay was consistently low throughout the study period, therefore did not display any seasonal variation. Cyanobacteria usually display seasonal succession and often increase in abundance during summer when nutrients become limiting to other phytoplankton and temperatures are at their warmest (Kong & Fao, 2005). Vacuolate and colonial/filamentous cyanobacteria are buoyant species of phytoplankton and their growth is optimised in a stratified water column (Wilhelm & Adrian, 2008), therefore the polymictic conditions within Cardiff Bay may be limiting cyanobacteria growth. Conversely (Reynolds & Walsby, 1975) argue that the lower light requirements of some cyanobacteria means that a moderate level of mixing can result in a competitive advantage for this phytoplankton group, increasing HAB risk. This emphasises the need to integrate nutrient data and hydrology in

understanding risk modelling of the Bay with respect to future changes in summer river flows predicted under IPCC scenario.

4.6.5 The Relationship between Environmental Variables & Phytoplankton in Cardiff Bay

Lake flushing rate negatively influenced cryptomonad biomass during autumn and summer in Cardiff Bay. When lake flushing quickened, cryptomonad biomass decreased. Cyanobacteria biomass also had the same relationship with lake flushing rate during autumn. Phytoplankton require time to grow, with fast lake flushing rates in Cardiff Bay, phytoplankton are not given enough time to increase in number before the lake water is flushed out (Rennella & Quirós, 2006). Therefore, the flushing rate of Cardiff Bay is a controlling factor of cryptomonad and cyanobacteria biomass during autumn and cryptomonad biomass during summer.

Phytoplankton biomass within Cardiff Bay were not controlled by air or water temperatures, which is unusual because an increase in temperature typically increases phytoplankton productivity (Butterwick et al., 2005). Precipitation and river discharge usually influence phytoplankton biomass in a lake through the external nutrient supply from river waters and runoff. When river discharge increases, and nutrient inputs rise, phytoplankton biomass often increases (Cloern et al., 1983). However, this relationship was not apparent in Cardiff Bay, likely as a result of the quick flushing rate of the lake preventing exposure of phytoplankton to the nutrients supplied by the rivers.

The absence of a relationship between DO and phytoplankton biomass indicates that the phytoplankton community of Cardiff Bay is not responsible for any variations in DO within the lake. It is reasonable to hypothesise that this may be a result of the aeration system preventing any relationship being formed, or that the phytoplankton biomass of the lake being too low to significantly contribute to the DO concentrations.

Photic depth within Cardiff Bay displayed no relationship with the phytoplankton biomass. This indicates that the phytoplankton were not light limited, their productivity was healthy therefore light was not a limiting factor. Light limitation usually comes into play within

stratified lakes, the polymictic nature of Cardiff Bay may prevent light limitation from occurring (Berger et al., 2006).

The phytoplankton biomass within Cardiff Bay is very low, making it difficult for statistically significant relationships to be formed between phytoplankton biomass and environmental factors within the lake. This study has highlighted that there are relationships between phytoplankton biomass and environmental variables within Cardiff Bay that could strengthen in the future, potentially changing the dynamic of the lake and impact management.

4.6.6 Other Factors Potentially Influencing the Phytoplankton of Cardiff Bay

Optimal phytoplankton growing conditions in a lake are warm with a stratified water column and sufficient light. The water temperature of Cardiff Bay has been predicted to increase (Welsh Assembly Government, 2009a) and light has been found to be sufficient within the lake. This makes water column mixing potentially the most influential environmental factor for phytoplankton growth. Few strong relationships have been witnessed between environmental factors and phytoplankton within Cardiff Bay, suggesting that other factors are influencing the phytoplankton biomass. The nutrient concentrations within the lake, and whether phytoplankton are limited by them are discussed in Chapter 5.

4.7 Conclusions

The phytoplankton biomass of Cardiff Bay was analysed between January 2003 and April 2014, with the phytoplankton productivity and assemblage determined for the period between April 2012 and April 2014. Effective lake management can only be implicated when the phytoplankton community and its seasonal succession is understood in relation to the environmental variables of a lake system, and the limitations and controls that effect the phytoplankton community.

Using chlorophyll *a* data collected by Cardiff Harbour Authority between January 2003 and April 2014, the phytoplankton biomass of the lake was found to be very low. Despite this, using

fluorescence data collected between April 2012 and April 2014 the phytoplankton population of Cardiff Bay was found to be healthy and productive and the identification of samples supported the chlorophyll *a* data taken between January 2003 and April 2014 that the dominant phytoplankton group within the lake were diatoms. Chlorophytes, and cryptophytes made up most of the rest of the phytoplankton population, with cyanobacteria biomass being insignificant throughout the study. The phytoplankton population of Cardiff Bay was not found to be light limited, or to influence the DO concentrations within the lake.

The polymictic nature of the lake prevented any spatial differences in phytoplankton biomass, productivity or assemblage from occurring, on the surface or at depth, within the lake. The low phytoplankton biomass prevented strong relationships from being formed with environmental variables, however, the phytoplankton biomass did vary seasonally. Variations followed the typical seasonal succession of a phytoplankton population, where biomass peaked in spring, declined through summer and autumn, before reaching its lowest levels during winter. Unusually, although chlorophytes demonstrated seasonal succession within the total biomass analysis, diatoms dominated during every season. Lake flushing rate was the only environmental variable that influenced the phytoplankton population of Cardiff Bay. During spring and summer when flushing rates quickened, the phytoplankton biomass decreased.

Although the phytoplankton biomass in the lake was low, a weak relationship between DO and phytoplankton was expected. It is possible that the aeration system within the Bay overshadows the effect of phytoplankton photosynthesis and respiration processes on DO concentrations within the lake. The only way to determine if there is a true relationship between them would be to switch off the aeration system for a period of time; something that was not able to take place during this study.

Understanding the phytoplankton community of this polymictic temperate lake and the environmental controls on the population will not only aid the formation of a long-term successful management plan for Cardiff Bay, it will help other lake management policies to be implemented in polymictic lakes. This study particularly provides an insight into how the phytoplankton community of an artificially controlled system with a constant lake level and the presence of an aeration system responds to changes in environmental factors, and the potential risks for future management.

Increasingly HABs are becoming a nuisance in lake systems, particularly as temperatures begin to rise. This study has highlighted how cyanobacteria biomass can be controlled using relatively natural processes; a fast lake flushing rate and an aeration system both ensuring that the water column is constantly mixed and regularly flushed to prevent cyanobacteria from dominating in the environmental conditions. At present the lake of Cardiff Bay is being suitably managed with regard to HAB prevention. However, the weak relationships found between phytoplankton biomass and temperature, precipitation, and river discharge, indicates that if the flushing rate of the lake slows, and the aeration system does not mix the water column effectively there is the possibility that an increase in temperature could increase phytoplankton biomass with the potential for cyanobacteria growth to be favoured. Therefore, future management recommendations will be required to ensure that the aeration system is used frequently enough to prevent the lake from stratifying.

THE BOTTOM-UP CONTROLS ON PHYTOPLANKTON WITHIN CARDIFF BAY

5.1 Summary

The bottom-up controls on the phytoplankton community of Cardiff Bay were analysed between January 2003 and April 2014. The phosphorus, nitrogen and silica concentrations within the main body of the lake were determined, along with the influence of inputs from the River Taff and River Ely. The potential for phosphorus release from the sediment was also explored, and the impact of phosphorus reduction management that took place during 2005 and 2006 was examined. Mescocosm experiments were used to determine if phytoplankton growth in the lake was nutrient limited. Although there was evidence of relative silica limitation within the lake, the phytoplankton population of Cardiff Bay was not found to be nutrient limited. Nutrient concentrations were sufficient for growth, and phosphorus release from lake sediments was low. It was determined that the hydrological control of lake flushing rate had more of an impact on the phytoplankton population of Cardiff Bay than bottom-up controls.

5.2 Introduction

The process of nutrient availability in a lake is the most significant bottom-up (BU) process that takes place within a lake. Nutrients are important for regulating freshwater food-webs as they have the ability to limit primary production and control the transfer of energy to herbivores (Taylor et al., 2015). Nutrient supply is crucial when determining the health and productivity of a freshwater lake food-web (Harper, 1992, Macintyre & Jellison, 2001, Musungu et al., 2014b), as the growth of phytoplankton is often limited seasonally by nitrogen or phosphorus (Rhee, 1978, Harper, 1992, Eaton et al., 2005, Carey et al., 2012). This causes the anthropogenic input of phosphorus and nitrogen to a lake system to often have a major impact on lake ecosystem structure by affecting water quality and causing eutrophication (Nyenje et

al., 2010a, Xu & Zhang, 2012, Abell et al., 2013a, Musungu et al., 2014b). The natural fluctuation in nutrient concentrations caused by seasonal and long-term meteorological changes are escalated by the addition of nutrients from human activities (Anneville et al., 2004a).

5.2.1 Phosphorus

Phosphorus (P) enters a lake system through the weathering of rock, and soil erosion, therefore the P supply to a lake is determined by the rate of runoff and river discharge into the system (Sterner, 2009, Kumar & Sekaran, 2014). Phosphorus that is locked up in bedrock, soils and sediments is not directly available to organisms, however when converted to SRP it can be directly assimilated (Ruttenberg, 2003). This means that a lake's capacity to absorb an increase in phosphorus loading increases with lake volume, depth, flushing and sedimentation rates (Holdren et al., 2001).

Phytoplankton productivity requires the presence of P, often making it the principle limiting nutrient in a lake (Holdren et al., 2001, Ruttenberg, 2003). When phytoplankton grow under P-rich conditions they express high growth rates (They et al., 2017).

5.2.2 Nitrogen

Nitrogen (N) is an essential nutrient for lake flora and fauna (United States Environmental Protection Agency, 2014). It enters a lake system through precipitation and soil erosion (Kumar & Sekaran, 2014), therefore the residence time of nitrogen in a lake is dependent on lake flushing rates, however the presence of vegetation and the rate of nitrification and denitrification also affects its availability (They et al., 2017). Nitrate is converted to ammonium by bacteria and cyanobacteria through nitrogen fixation. Bacteria then convert this ammonium to nitrate and nitrite through the process of nitrification. Denitrification also occurs where nitrate is reduced to dinitrogen gas (Arnell, 2014).

Nitrate, nitrite, and phosphorus can be loaded externally into a lake from anthropogenic sources. Precipitation can wash excess nutrients from residential areas and agricultural land

and sewerage treatment works often discharge effluent into water bodies (Harper, 1992, Perkins & Underwood, 2001a, Anneville et al., 2004a, Eaton & Franson, 2005, Chen & Taylor, 2011, Skwierawski, 2012). Therefore anthropogenic loading can increase the ratio of nitrogen to phosphorus, as well as silicate, with the ability to intensify phosphorus limitation (Hessen et al., 2009).

5.2.3 Silica

Silica enters a lake system through the weathering of rock (Kumar & Sekaran, 2014) and is often considered as a non-essential trace element in lakes, although it is often present in low concentrations in freshwaters, concentrations are usually well above limiting values during the seasons of spring and summer where optimum phytoplankton growth takes place (Conley et al., 1989, Harper, 1992, Tallberg, 2000, Eaton et al., 2005). However, silica is vital for diatom communities and under certain conditions can become a limiting (Conley et al., 1989, Harper, 1992, Tallberg, 2000, Eaton et al., 2005).

5.2.4 Nutrient Input and Accumulation within a Lake

Rivers are a dominant external input of nutrients to a lake, particularly if they drain agricultural and industrial land (He et al., 2012). A combination of high nutrient loading and a low lake flushing rate can cause nutrient loading problems as water within the system is not replenished often enough to prevent the accumulation of nutrient-rich run-off in the lake. This forms a negative relationship between phosphorus concentrations and the flushing rate of a lake (Brett & Benjamin, 2008). Crucially though, the flushing rate of a lake if fast enough, also has the ability to dilute and remove nutrients from a lake system (Interagency Freshwater Group, 2015).

If the external loading of nutrients in a lake is reduced (for example, through precipitation and river discharge reduction), it can cause the internal loading, particularly of phosphorus, from lake sediments. Phosphorus release from sediments originates from a pool accumulated when nutrient levels in the water column were high. If this sediment accumulation of phosphorus is high, the internal input of phosphorus to a lake from the sediment has the ability to last for

decades (Sondergaard et al., 2001). The level of dissolved oxygen in a lake has been found to be the most influential factor of P release from lake sediments; under anaerobic conditions, phosphorus release is much higher than under aerobic conditions, making the DO concentration of a lake a controlling factor of nutrient availability when DO concentrations are low (Wu et al., 2014, Zhang et al., 2015).

High levels of phosphorus, from internal or external inputs can result in eutrophication within a lake. Eutrophication is the response of a lake to high nutrient levels. Phytoplankton blooms form, which can result in an increased turbidity and anoxia of the water column, ultimately affecting higher trophic organisms such as fish (Schindler et al., 2008). It can also lead to the formation of harmful algal blooms (HABs), commonly composed of cyanobacteria. Eutrophication has been established as the most important problem affecting 44% of lakes in the United States (Holdren et al., 2001) and is one of the most pervasive threats to lake surface waters (Wu et al., 2014).

Freshwater systems are generally stoichiometrically variable as a result of fast flushing rates and high contributions of total organic matter from the watershed (Sterner, 2009, They et al., 2017). However, in many countries, the water quality of lakes has been improved by reducing total phosphorus (TP) loading. This often results in a decrease in chlorophyll *a* concentrations and more specifically cyanobacteria (World Health Organization, 2003, Jeppesen et al., 2005, Jeppesen et al., 2009, Allinger & Reavie, 2013).

5.3 Hypotheses

To determine the bottom-up controls on phytoplankton biomass, productivity and community structure within Cardiff Bay the following thesis objectives will be addressed within this chapter:

The bottom-up control on phytoplankton biomass, productivity and community structure will be investigated using phosphate, nitrate, nitrite, ammonium and silica concentrations, alongside the potential of nutrient release from the lake sediment.

Using data collected between January 2003 and April 2014 in Cardiff Bay, the following hypotheses have been tested:

The concentration of phosphorus, nitrogen and silica within the lake do not vary spatially or with depth.

There is a seasonal variation in phosphorus, nitrogen and silica concentrations within the lake.

Compared to nitrogen and silica, phosphorus is the most limiting nutrient in Cardiff Bay, causing phosphorus concentrations to determine phytoplankton biomass.

Phosphorus exchange between the lake bed and the water column is relatively low due to the water column being well-mixed.

5.4 Materials and Methods

In 2002, the 200 ha lake of Cardiff Bay was designated as a sensitive water. This resulted in the removal of phosphorus at five main sewerage treatment works that discharge into the River Taff and River Ely catchments. Phosphorus removal began between March 2005 and March 2006 with the aim of reducing excessive phytoplankton growth in the lake.

5.4.1 Data Processing

Cardiff Harbour Authority (CHA) collected nutrient data once a month between January 2003 and April 2014 at sites 4, 5, 6, 9, 10 and 17 (site locations can be found in Chapter 2). Surface (water depth c1 m) concentrations were determined for ammonium ($n = 115$), nitrate ($n = 114$), nitrite ($n = 118$), soluble reactive phosphorus (SRP) ($n = 118$), total phosphorus (TP) ($n = 118$), and silica ($n = 118$). Total nitrogen (TDN) values ($n = 118$) were calculated as the sum of ammonium, nitrate and nitrite.

In more recent years the nutrient data collection by CHA was restricted to once every three months. Missing data was not interpolated as the timescale between data was too large. Some

data supplied were labelled as ‘less than’ values (below the current level of detection at the time of analysis), these were converted to the highest possible concentration to allow for statistical analysis to take place. Outliers were removed from nitrite data in March 2008, SRP data in April 2007 and TP data in April 2005. All of these values were significantly higher than any other data collected, therefore were replaced with interpolated values. Error bars were calculated using standard error.

Individual site means for ammonium, nitrite, nitrite, TDN, SRP, TP and silica were calculated for sites 4, 5, 6, 9, 10 and 17 using all data collected at 1 m on an annual, seasonal and monthly basis from January 2003 until April 2014. Additionally, mean annual and seasonal whole-Bay data were calculated using all data collected at 1 m depth from sites 4, 5, 6, 9, 10 and 17 between January 2003 and April 2014.

Between April 2012 and April 2014 for the purpose of this study, at sites 4, 5, 9, 10, A, B, C and D concentrations of TP, SRP, silica, nitrate, nitrite, ammonium and TDN once every two weeks. Concentrations of TP, SRP, silica, nitrate, nitrite, ammonium and TDN were determined from the lake surface, the middle of the water column and c1 m from the lake bed. Mean site values were calculated using data collected throughout the water column at sites 4, 5, 9, 10, A, B, C and D. Mean whole-Bay nutrient data were calculated using all of the data collected at all 8 sites and from the top, middle and bottom of the water column.

Labile, calcium bound (Ca) and iron-aluminium (Fe-Al) bound phosphorus found within sediment cores was converted from mg L⁻¹ to mg P g⁻¹ dry weight. Data refers to summer and winter only (see chapter 2 for details of site locations and time of data collection).

The loading of total phosphorus was calculated using the following equation, adapted from the Vollenweider (1976) and Co-Operation & Development (1982):

$$L_c \text{ (mg/m}^2\text{.y)} = P_c^{\text{SP}} (z/T_w + 10)$$

Where:

L_c is the critical loading of phosphorus

P_c^{SP} is the critical concentration of total phosphorus (mg/m³) for simplicity taken at spring overturn – Cardiff Bay does not have a spring overturn but used spring 2013 data.

z is the mean depth

T_w is the water residence time

To calculate phosphorus storage in the lake sediment the following calculation was used (Perkins, 2000):

$$[P] \text{ in the lake} \times \text{lake volume} = (\text{input volume} \times \text{input } [P]) - (\text{output volume} \times \text{output } [P]) - \text{sediment P}$$

Annual and seasonal surface water temperature, precipitation, dissolved oxygen, River Taff and River Ely discharge and lake flushing rates were all calculated as described in chapter 3. A description of the phytoplankton biomass and productivity data used can be found in chapter 4.

5.4.2 Statistical Analysis

Mean monthly values for all data sets were used individually to conduct the decomposition of additive time series in 'Relevance' (R) (R Core Team, 2013), the script can be found in Chapter 3.

Pearson's correlation was undertaken on seasonal data sets using R to determine the relationships between nutrients and phytoplankton.

Nested Analysis of Variance (ANOVA) was carried out using MINITAB statistical software to determine if there was any variation in data on a seasonal basis or throughout the water column. Single-factor ANOVA was used on data sets shown to be significant in nested ANOVA to identify more specifically where differences lie using *post hoc* Tukey test ($\alpha = 0.05$) in MINITAB.

Two sample data sets when comparing water parameters, pre and post 2006, were initially tested using ANOVA. Where data showed no significant differences, data were pooled for analysis of pre and post 2006 as two samples. As sample size was small $n = 5$ and 9 , and data were not normally distributed a Mann Whitney U test was applied using MiniTab.

Seasonal data were used for linear regression analysis using R to determine if there was a “cause and effect” relationship between the meteorological and hydrological variables in Cardiff Bay. Following significant linear regression results, multiple regressions using R were undertaken using backward elimination to refine the “cause and effect” relationships.

Principle Component Analysis (PCA) variables (sites) were correlated against principle components to determine significant strength in separation between sites, with significance reported as $P < 0.05$.

5.5 Results

5.5.1 The Vertical & Spatial Distribution of Nutrients within Cardiff Bay

Mean monthly surface nutrient concentrations recorded from January 2003 until April 2014 found that site 17 in the River Ely had significantly higher concentrations of nutrients than at other sites. Ammonium at site 17 was significantly higher than the other sites studied ($F_{5,794} = 8.6$, $P < 0.01$) (Figures 5.1a and 5.1b). PCA analysis also found that mean monthly ammonium concentrations at sites 9 and 17 were spatially significantly different with site 9 showing a positive correlation with positive with PC2, and site 17 showing a negative correlation with PC2 (Figure 5.2). The significantly higher concentrations of nitrite at site 17 ($F_{5,794} = 32.98$, $P < 0.01$), was weak. There was only a weak significantly higher concentration at site 17 of nitrate ($P < 0.05$). PCA analysis indicated that nitrate at site 4 and site 17 separated spatially from other sites. Site 4 showed positive correlation with PC1 and negative correlation with PC2. Site 17 showed a positive correlation with PC2 (Figure 5.3). TDN concentrations were found to be significantly higher at site 17 than the other study sites ($F_{5,794} = 31.82$, $P < 0.01$), (Figures 5.4a and 5.4b). PCA analysis for TDN showed the same pattern as nitrate (Figure 5.5). At site 17 TP concentrations were found to be significantly higher ($F_{5,794} = 8.98$, $P < 0.01$) (Figure 5.6a and 5.6b), with PCA analysis indicated that site 1, 4 and 7 were significantly spatially separated. Site 1 showed a positive correlation with PC1 and negative correlation with PC2; sites 4 and 17 showed positive correlation with both PC1 and PC2 (Figure 5.7). There was no statistical difference in SRP concentrations between sites. PCA analysis indicated that significant spatial separation of sites 6 and 17 from other sample sites. Site 17 showed positive

correlation with PC1 and negative correlation with PC2. Site 6 showed positive correlation with PC1 and PC2 (Figure 5.8). Silica concentrations were significantly higher at site 17 than the other sites studied ($F_{5,794} = 9.56, P < 0.01$), (Figures 5.9a and 5.9b) & PCA analysis showed that only site 17 was spatially separated with a positive correlation with PC2 (Figure 5.10).

The sites surveyed between April 2012 and April 2014 found that at site A, situated at the mouth of the River Ely, fortnightly surface nitrate concentrations were significantly higher ($F_{7,385} = 3.2, P < 0.01$) than the concentrations at site C in the marina ($1.72 \pm 0.75 \text{ mg L}^{-1}$), and site D adjacent to the wetlands ($1.86 \pm 0.75 \text{ mg L}^{-1}$). The fortnightly surface concentrations of TDN at site A were also significantly higher ($F_{7,388} = 3.5, P < 0.01$) than at site B to the East of the lake ($1.65 \pm 0.79 \text{ mg L}^{-1}$), site C ($1.84 \pm 0.75 \text{ mg L}^{-1}$) and site D ($1.96 \pm 0.75 \text{ mg L}^{-1}$).

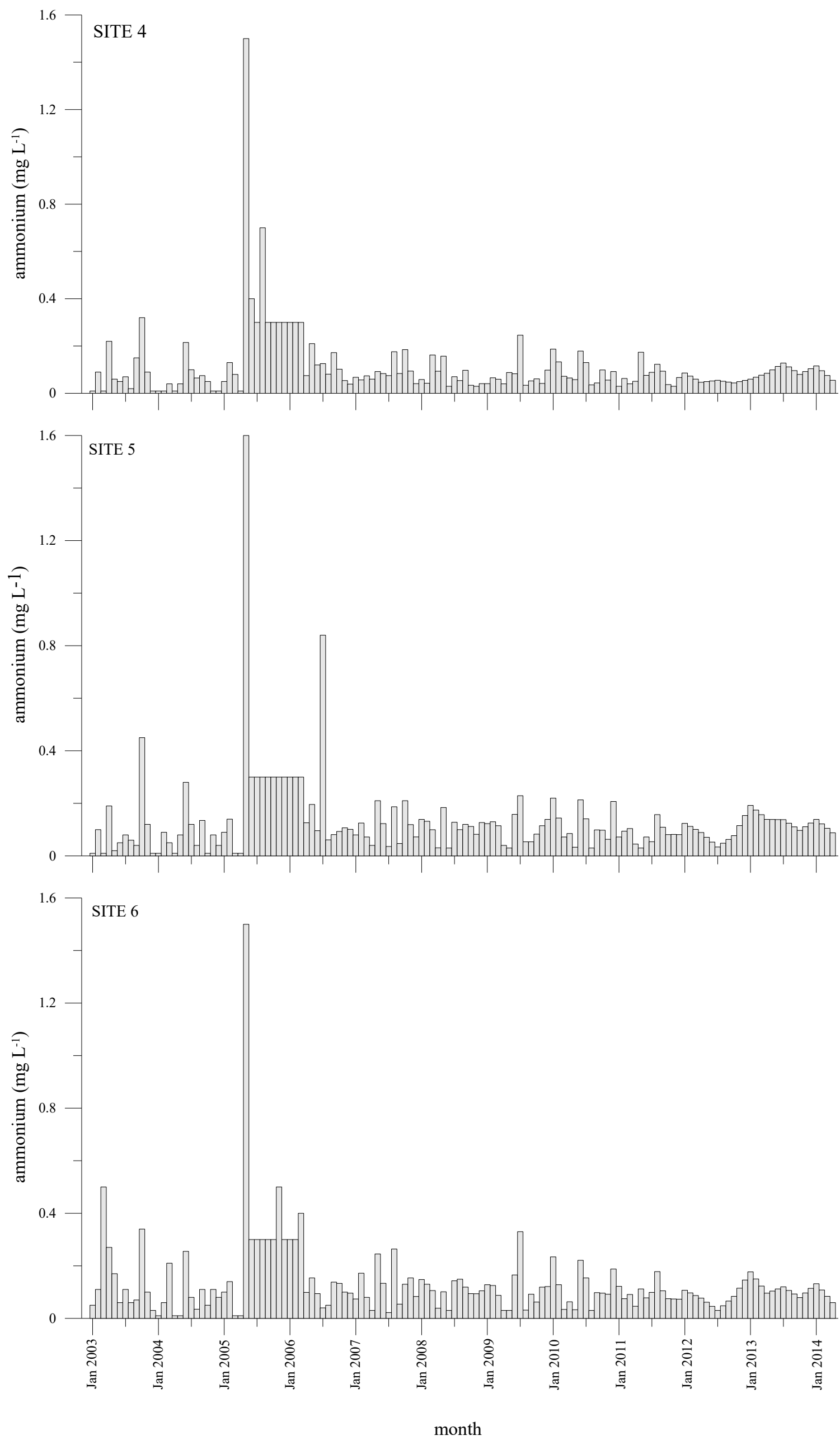


Figure 5.1a: Monthly surface ammonium concentration at site 4, 5, and 6 in Cardiff Bay between January 2003 until April 2014.

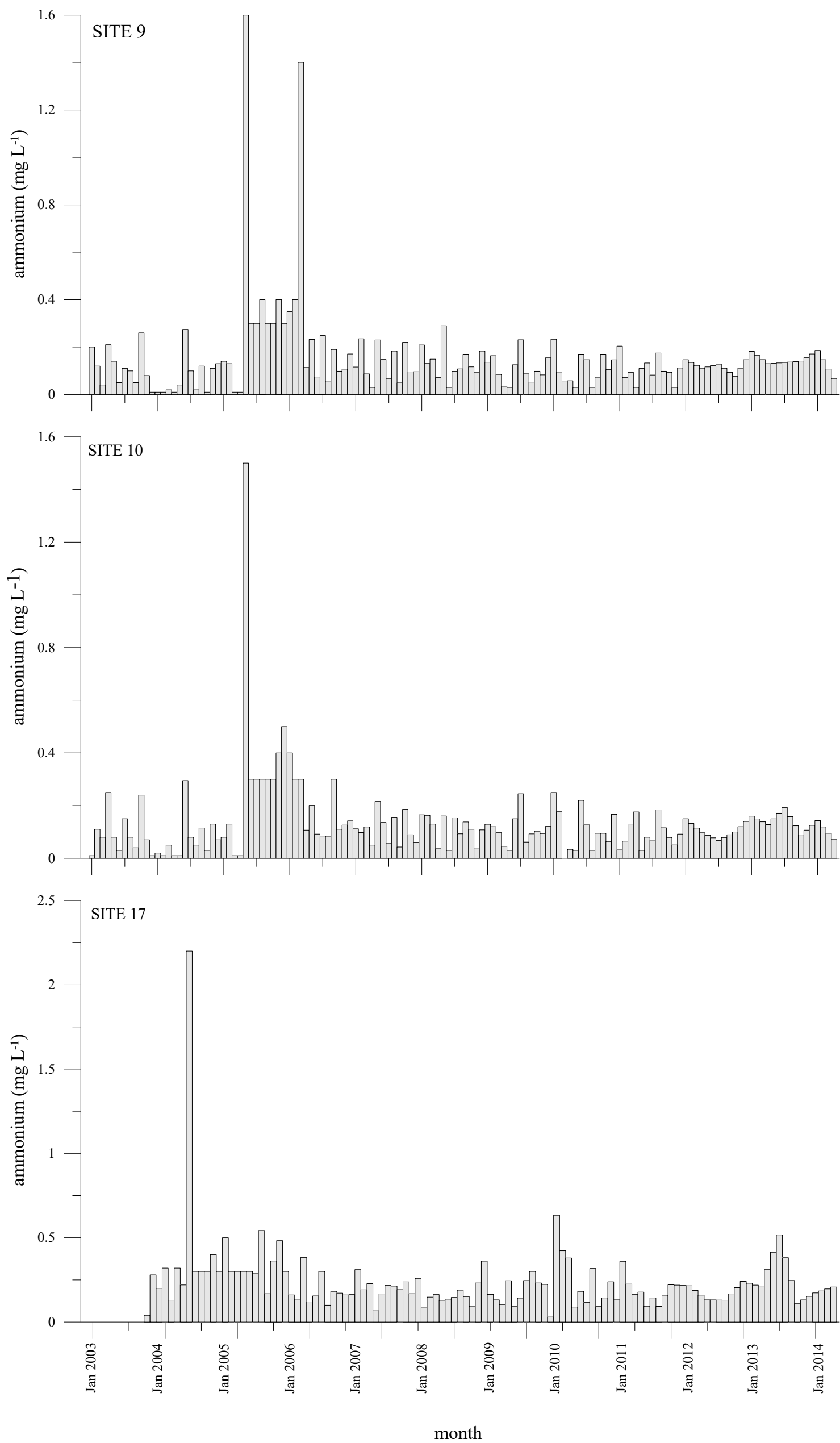


Figure 5.1b: Monthly surface ammonium concentration at site 9, 10 and 17 in Cardiff Bay between January 2003 until April 2014.

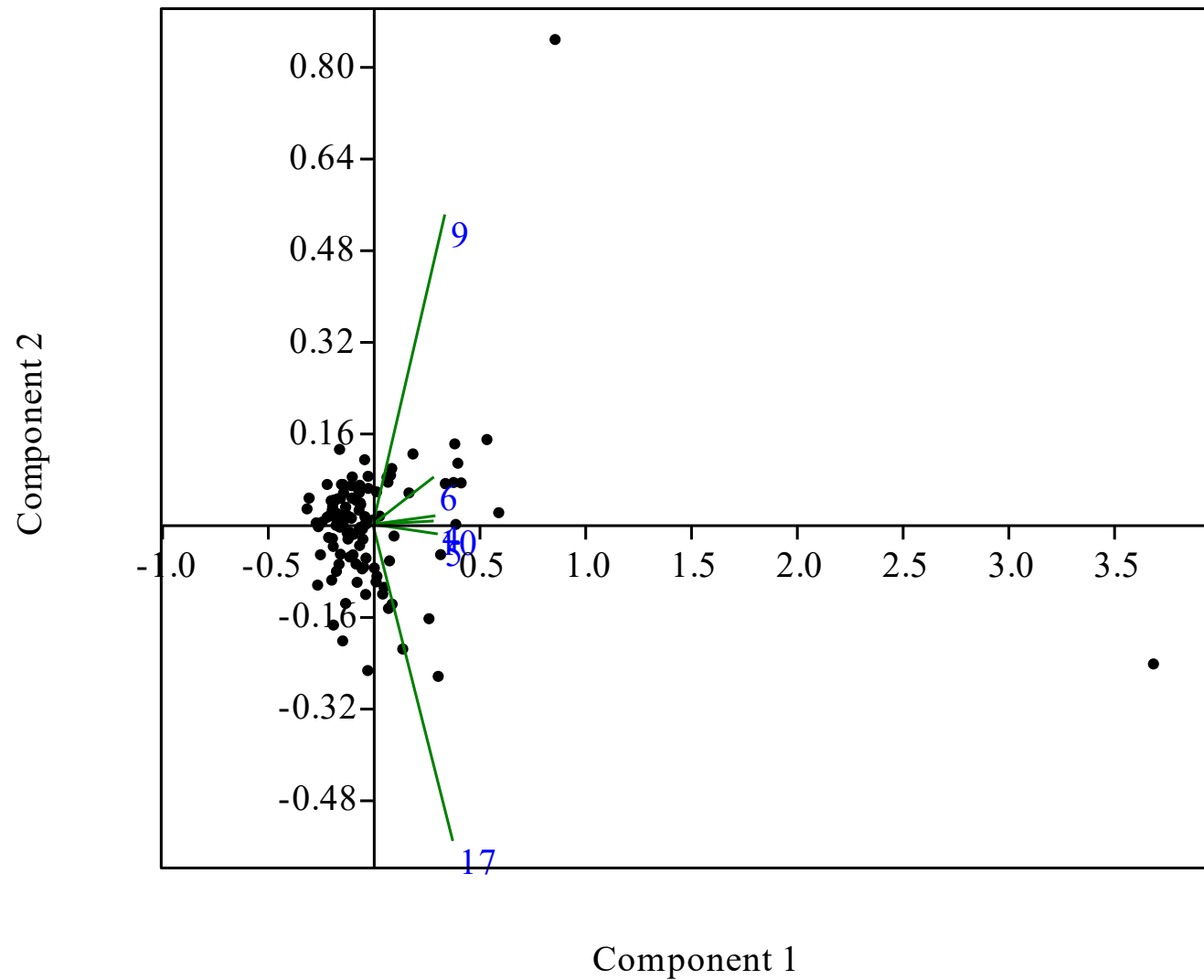


Figure 5.2: Principle Component Analysis of mean monthly ammonium between January 2003 and April 2014 in Cardiff Bay.

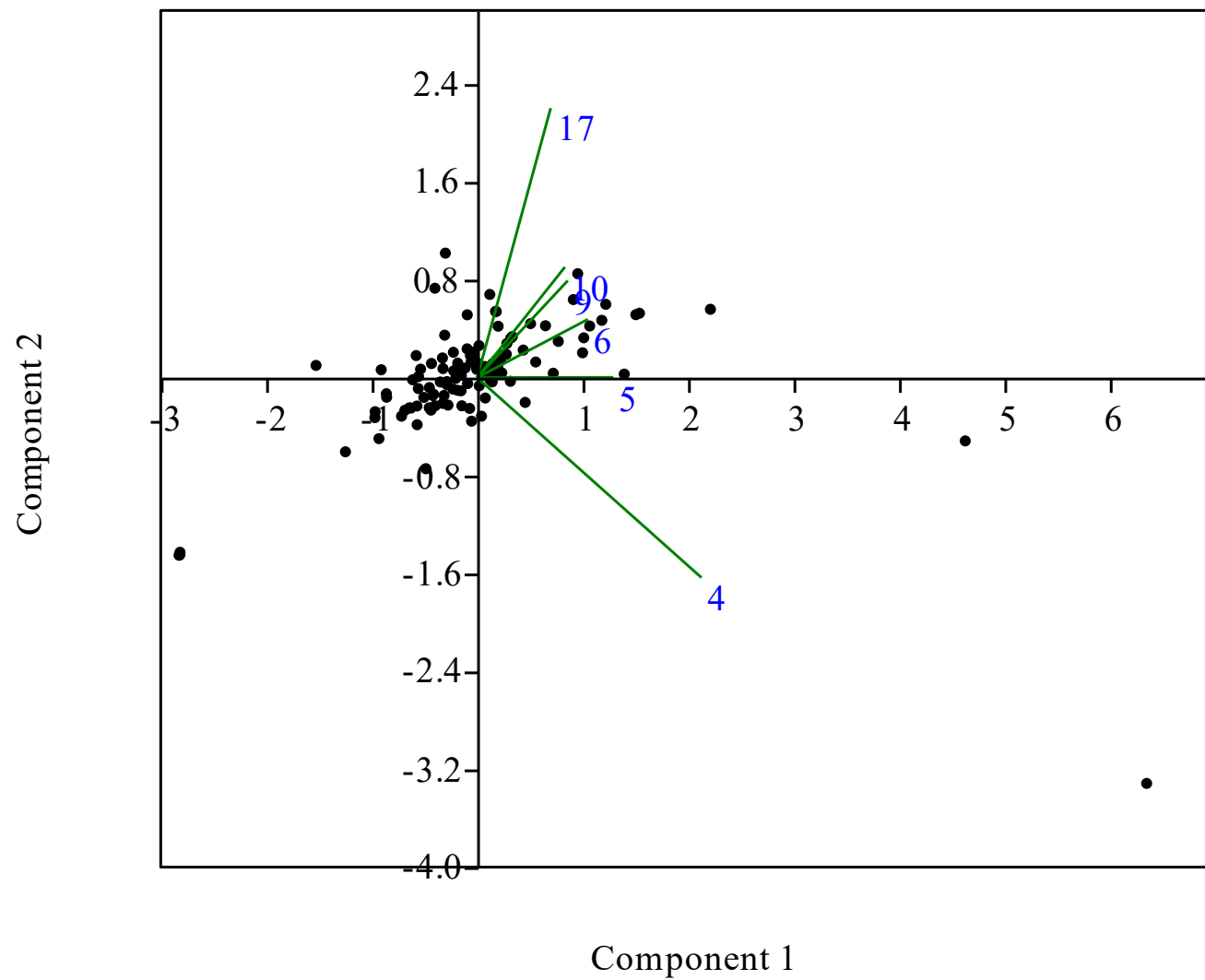


Figure 5.3: Principle Component Analysis of mean monthly nitrate between January 2003 and April 2014 in Cardiff Bay.

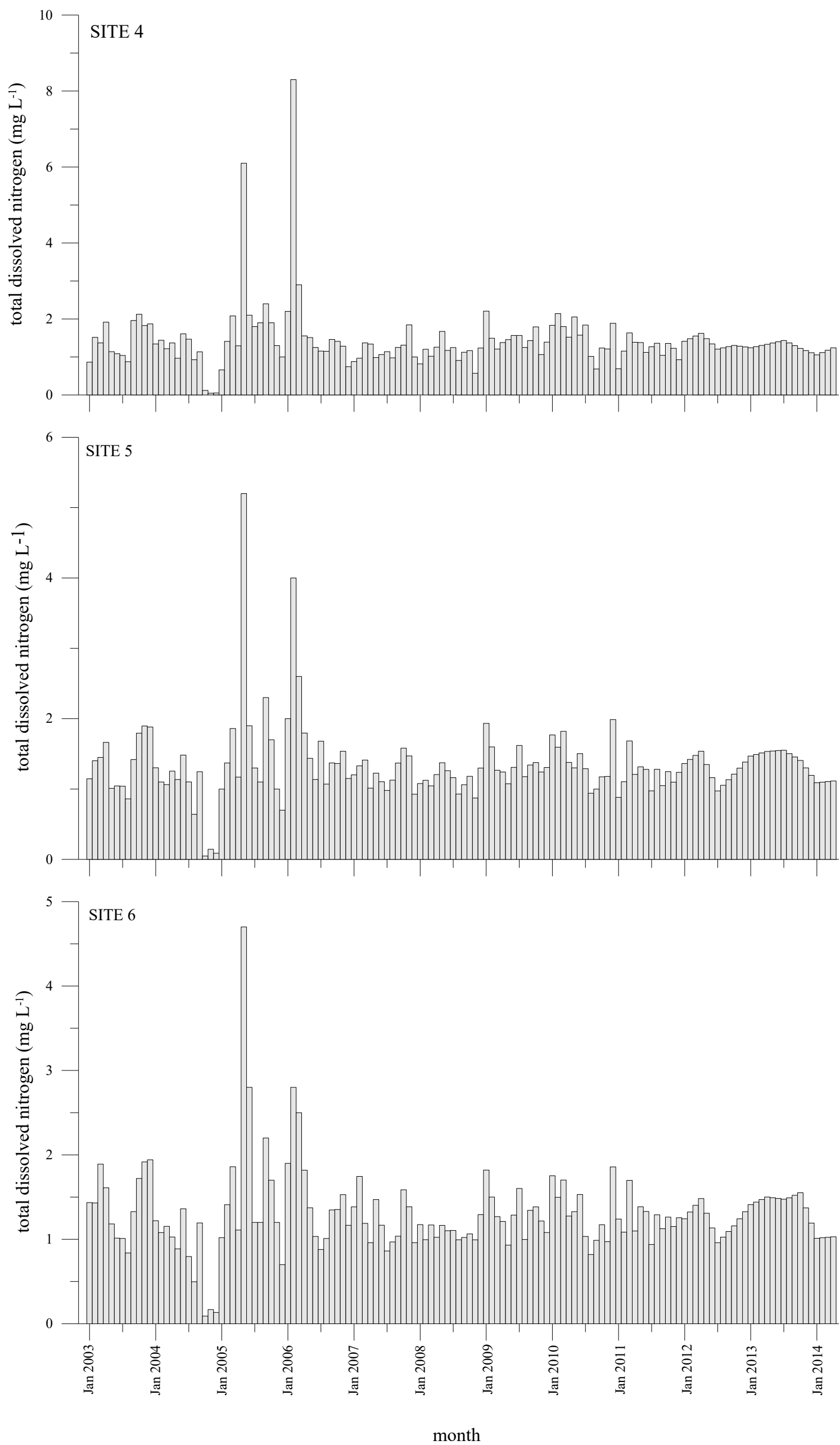


Figure 5.4a: Monthly surface total dissolved nitrogen (TDN) concentration at site 4, 5 and 6 in Cardiff Bay between January 2003 until April 2014.

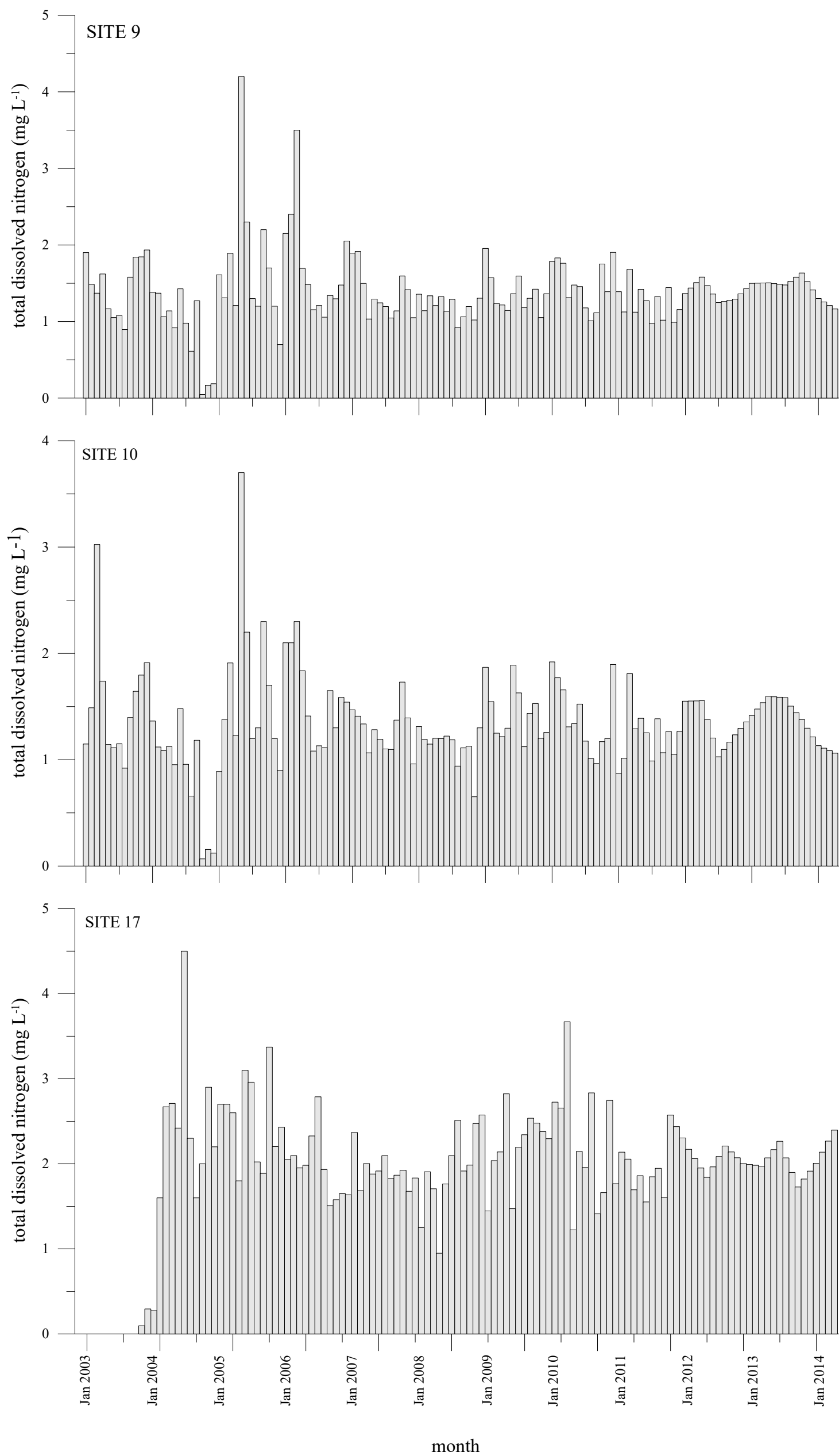


Figure 54b: Monthly surface total dissolved nitrogen (IDN) concentration at site 9, 10 and 17 in Cardiff Bay between January 2003 until April 2014.

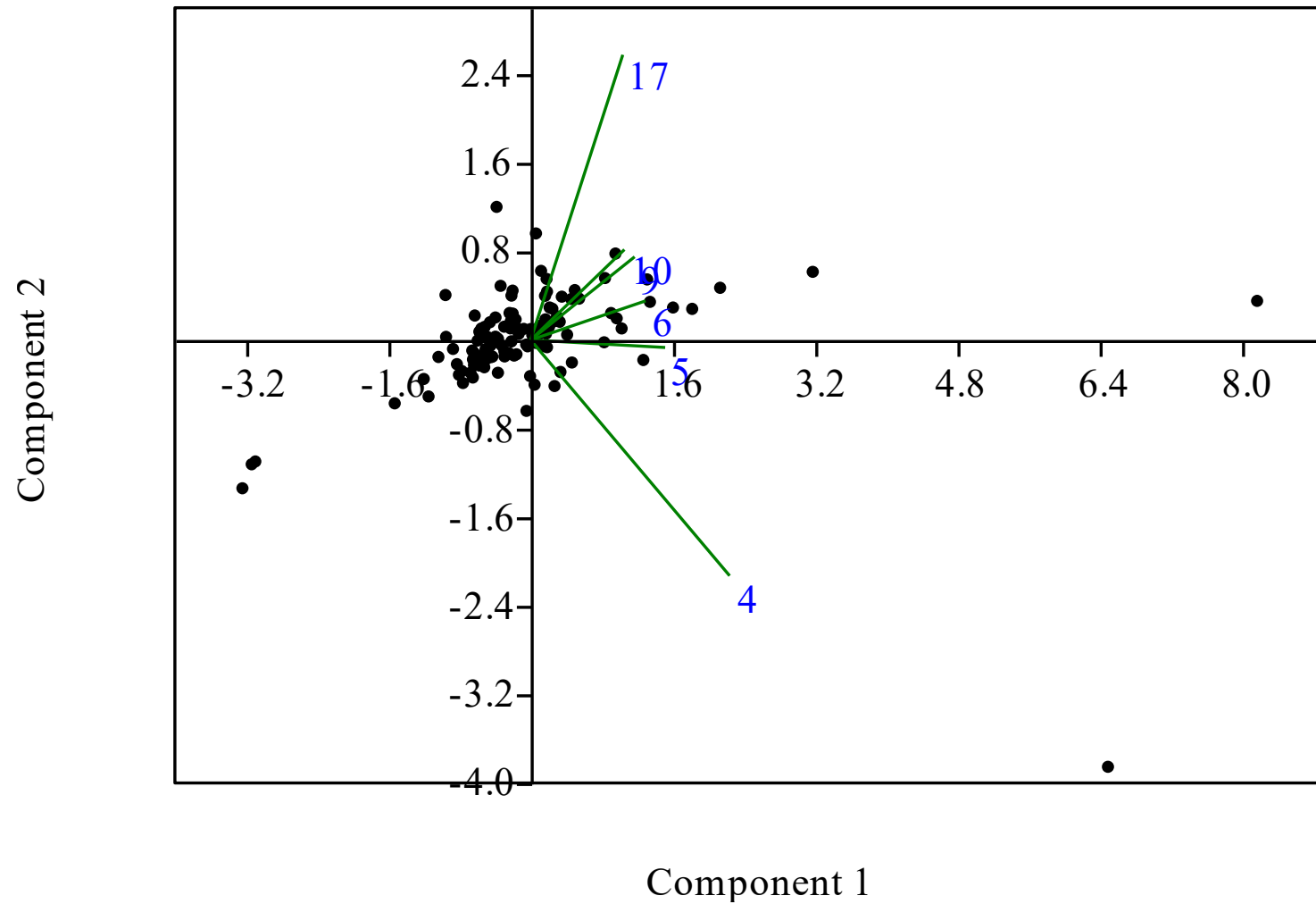


Figure 5.5: Principle Component Analysis of mean monthly total dissolved nitrogen between October 2004 and April 2014 in Cardiff Bay.

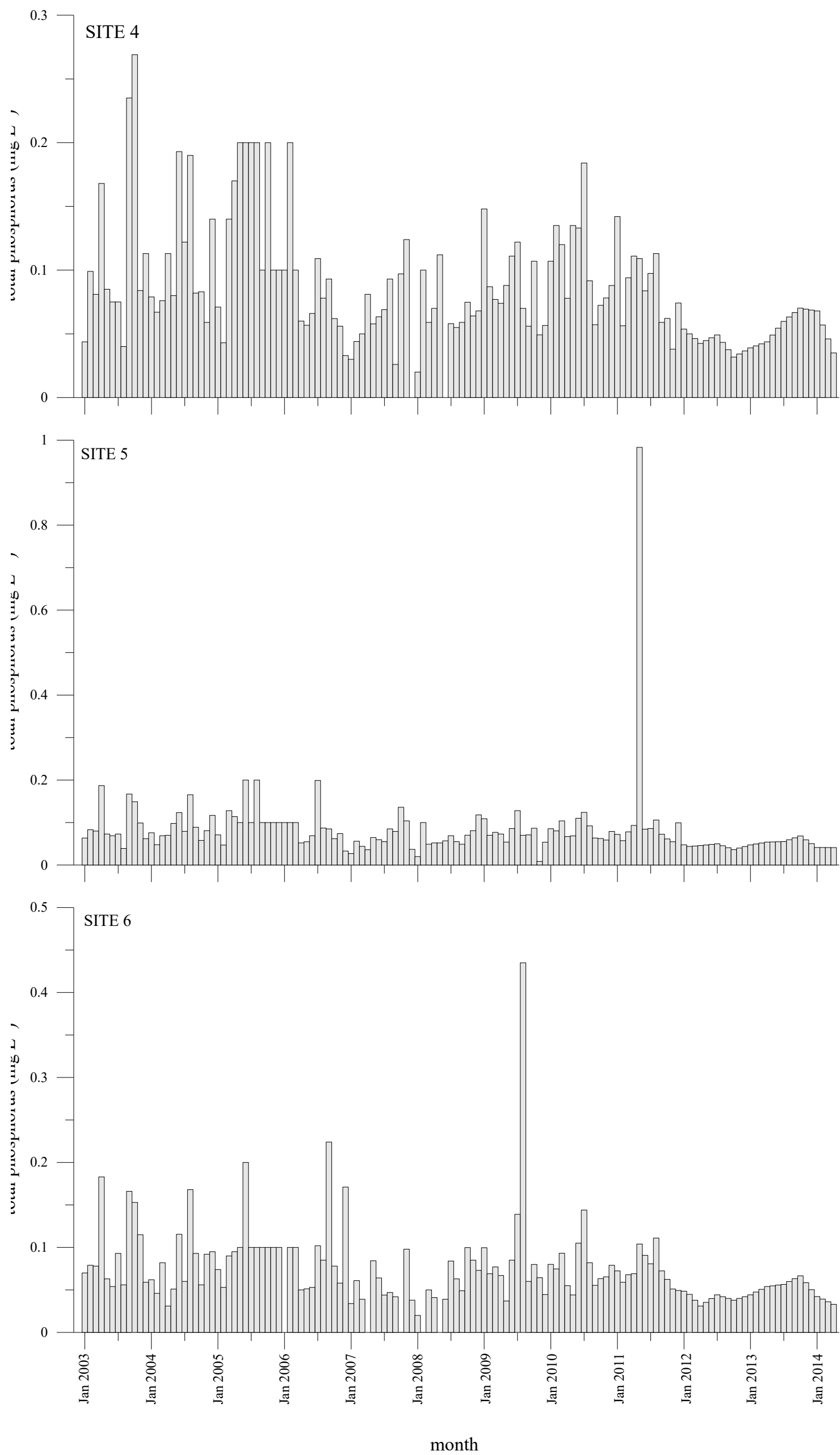


Figure 5.6a: Monthly surface total phosphorus (TP) concentration at site 4, 5 and 6 in Cardiff Bay between January 2003 until April 2014.

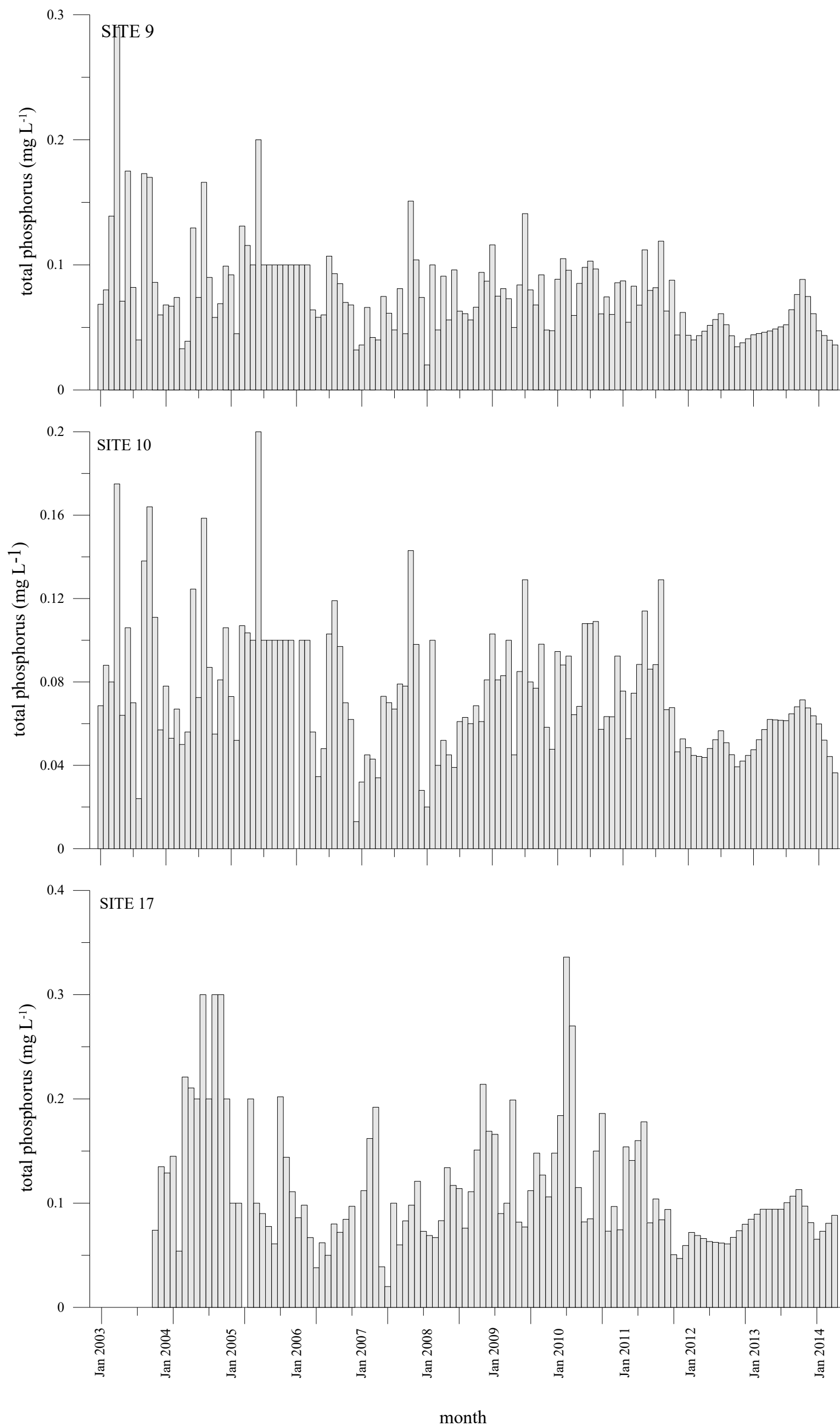


Figure 5.6b: Monthly surface total phosphorus (TP) concentration at site 9, 10 and 17 in Cardiff Bay between January 2003 until April 2014.

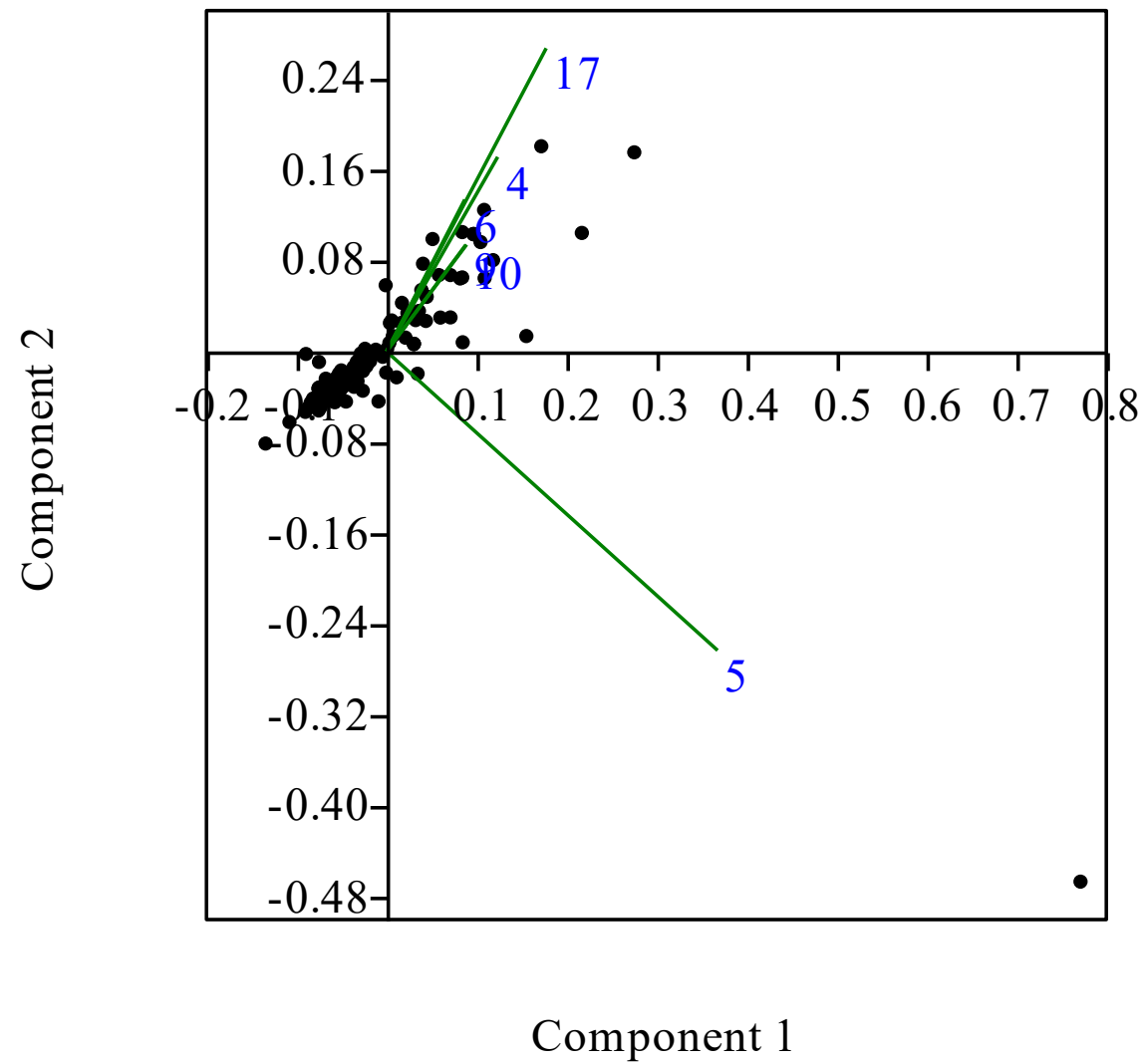


Figure 5.7: Principle Component Analysis of mean monthly TP between January 2003 and April 2014 in Cardiff Bay.

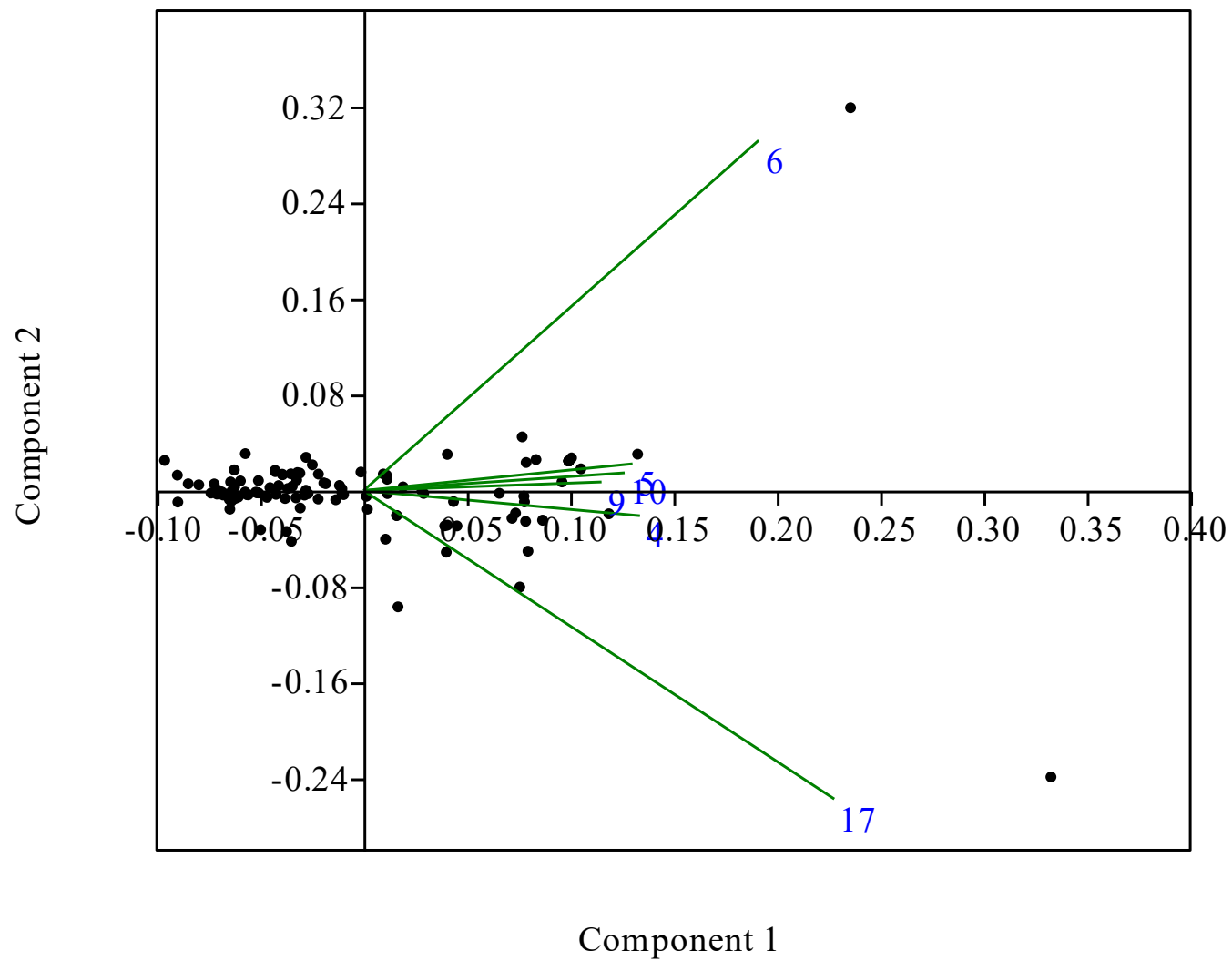


Figure 5.8: Principle Component Analysis of mean monthly SRP between January 2003 and April 2014 in Cardiff Bay.

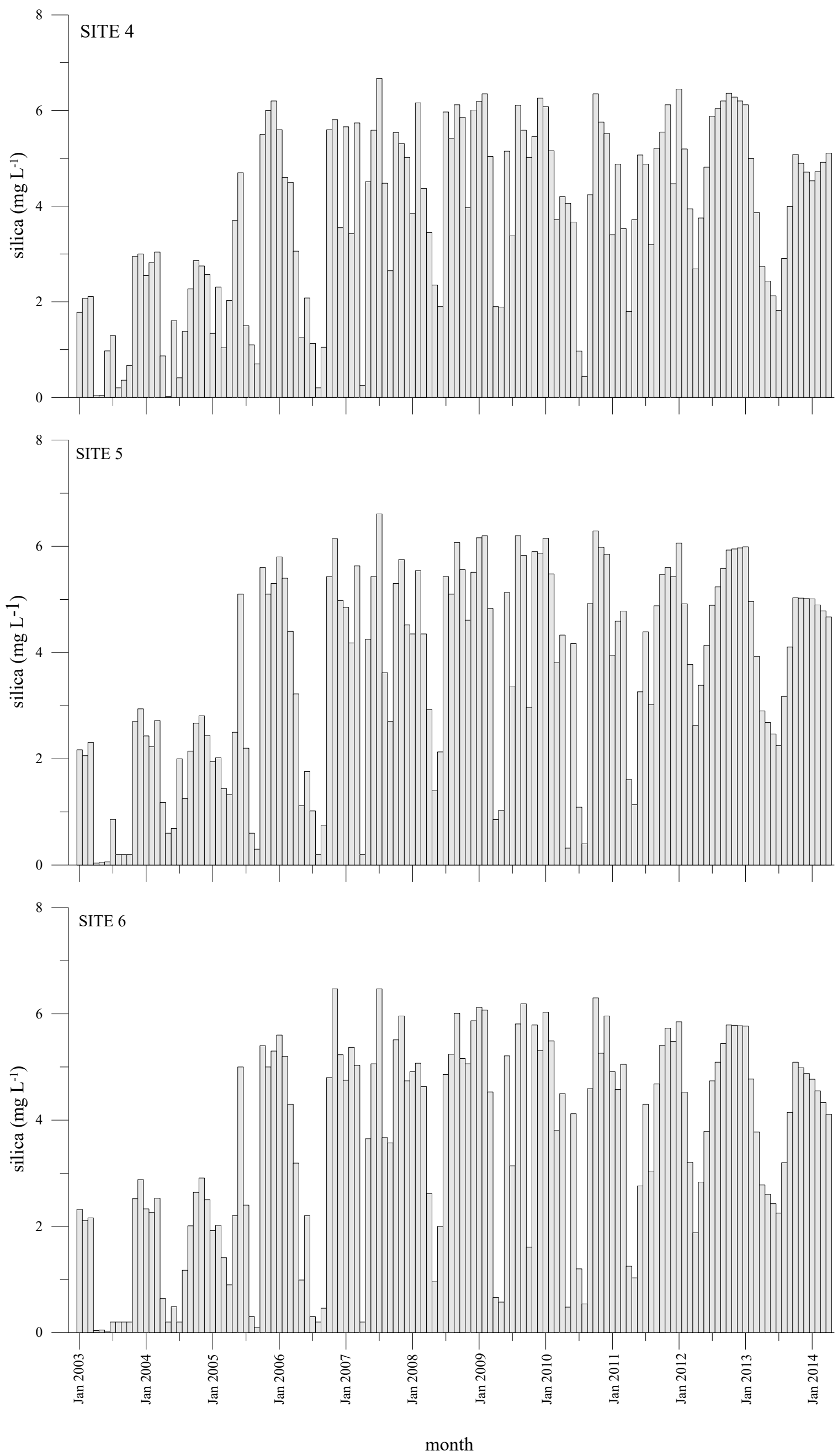


Figure 5.9a: Monthly surface silica concentration at site 4, 5 and 6 in Cardiff Bay between January 2003 until April 2014.

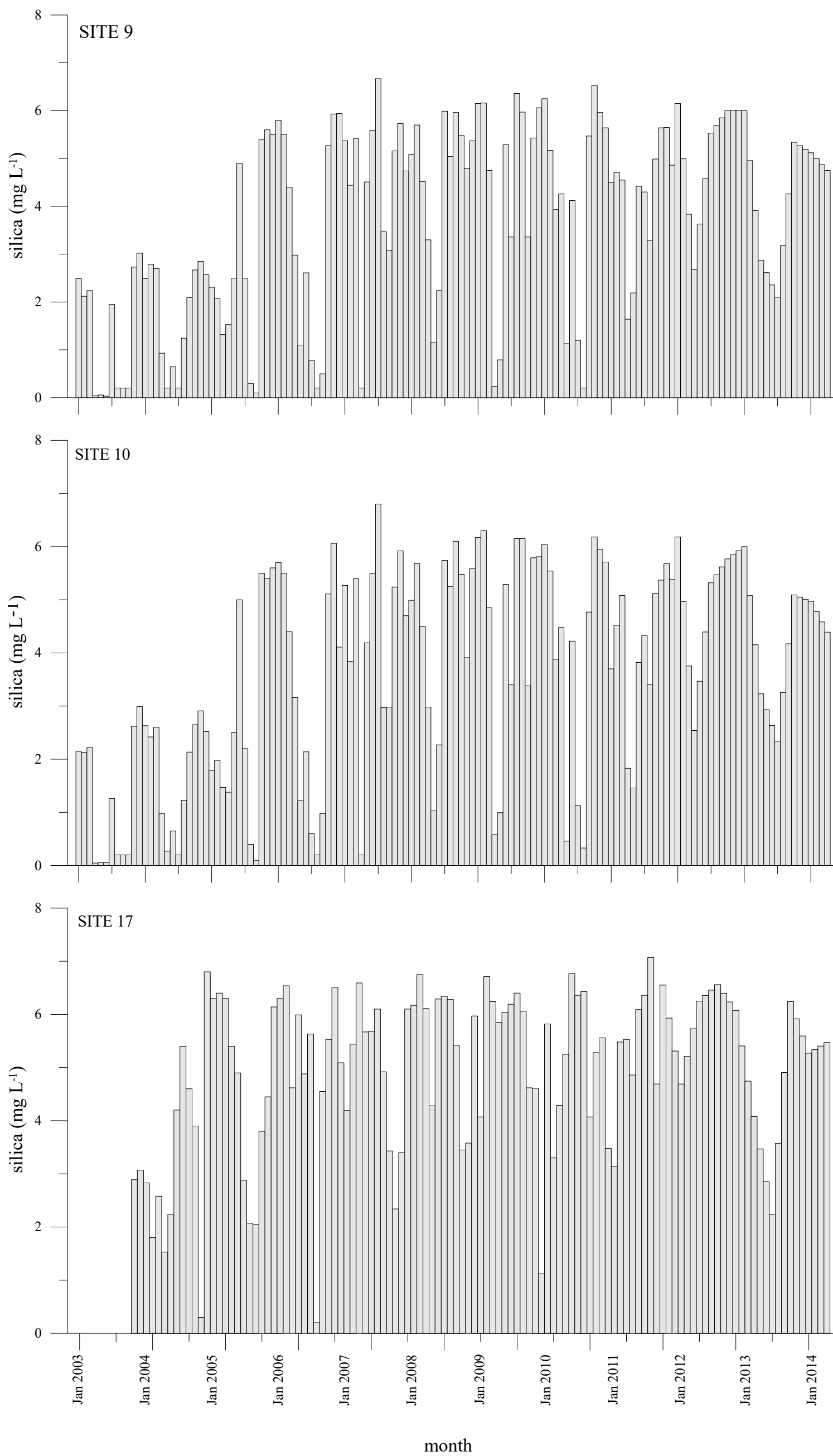


Figure 59b: Monthly surface silica concentration at site 9, 10 and 17 in Cardiff Bay between January 2003 until April 2014.

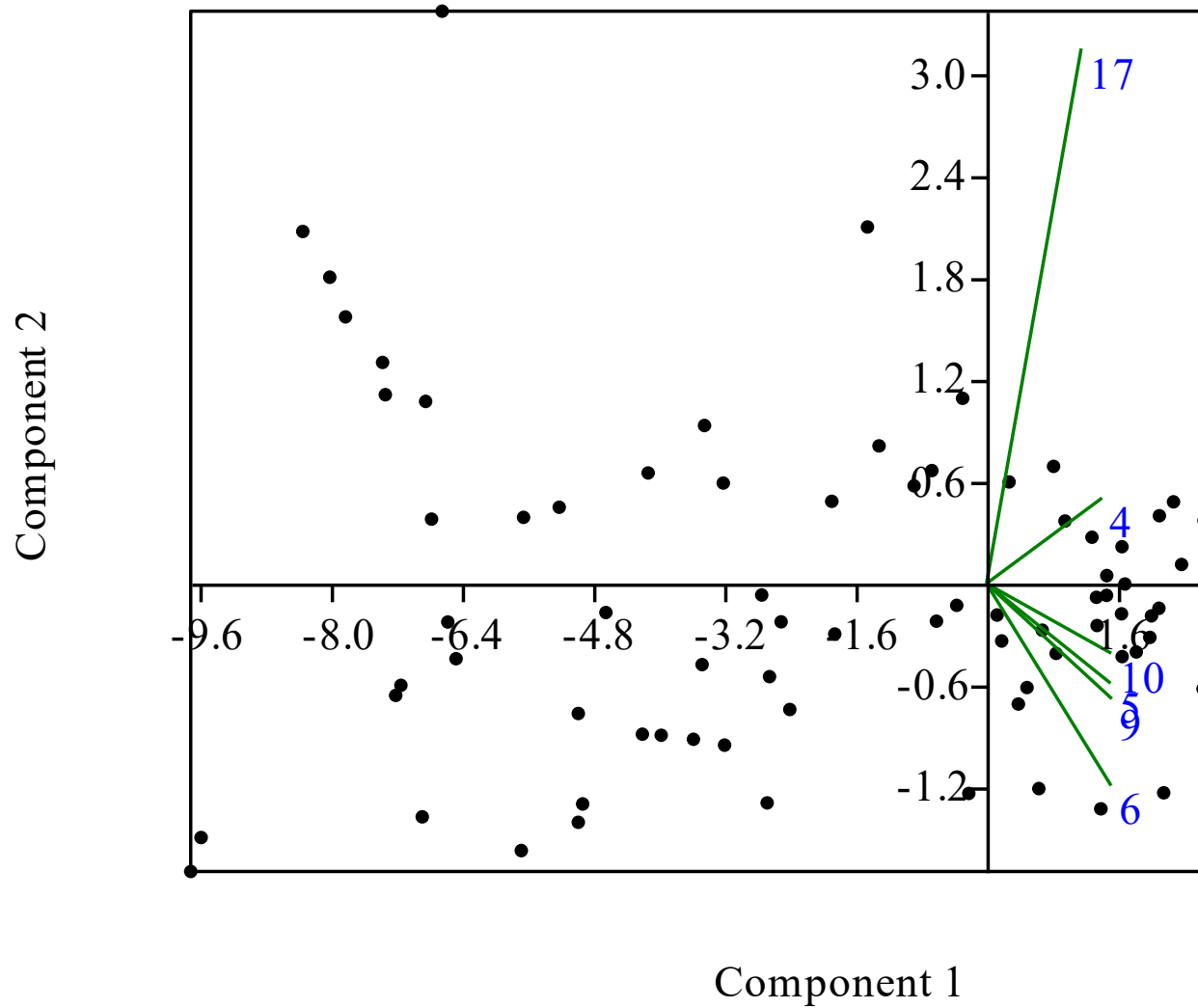


Figure 5.10: Principle Component Analysis of mean monthly silica between January 2003 and April 2004

At the bottom of the water column, between April 2012 and April 2014, fortnightly nitrate concentrations were significantly ($F_{7,381} = 6.4$, $P < 0.01$) different between sites; the nitrate concentrations at site A were higher than those at site 5 situated in the middle of the lake ($2.02 \pm 0.88 \text{ mg L}^{-1}$), site B ($2.61 \pm 0.82 \text{ mg L}^{-1}$), site C ($2.72 \pm 0.81 \text{ mg L}^{-1}$) and site D ($3.04 \pm 0.84 \text{ mg L}^{-1}$). The nitrate concentrations at the bottom of the water column at site D were lower than those at site 4 in the mouth of the River Taff ($1.79 \pm 0.77 \text{ mg L}^{-1}$) and site 9 to the West of the lake ($1.71 \pm 0.72 \text{ mg L}^{-1}$).

Between April 2012 and April 2014, at the bottom of the water column, fortnightly TDN concentrations were also significantly different ($F_{7,381} = 6.4$, $P < 0.01$) between sites; the TDN concentrations at site A were higher than those at site 5 ($2.03 \pm 0.88 \text{ mg L}^{-1}$), site B ($2.79 \pm 0.88 \text{ mg L}^{-1}$), site C ($2.81 \pm 0.81 \text{ mg L}^{-1}$) and site D ($3.12 \pm 0.84 \text{ mg L}^{-1}$). The TDN concentrations at the bottom of the water column at site D were lower than those at site 4 ($1.87 \pm 0.77 \text{ mg L}^{-1}$) and site 9 ($1.79 \pm 0.72 \text{ mg L}^{-1}$).

Using fortnightly surface, middle and bottom water data sampled between April 2012 and April 2014, there was no vertical difference in TP, SRP, silica, ammonium, nitrate, nitrite or TDN concentrations recorded at any site.

5.5.2 The Temporal Variation of Total Phosphorus in Cardiff Bay

The mean annual whole-Bay surface concentrations of TP in Cardiff Bay (Figure 5.11) varied significantly between years ($F_{10,131} = 5.19$, $P < 0.001$). Mean annual whole-Bay surface TP concentrations were significantly lower in 2012 than in 2003 ($0.057 \pm 0.048 \text{ mg L}^{-1}$). 2005 TP concentrations were significantly higher than those in 2007 ($0.058 \pm 0.011 \text{ mg L}^{-1}$), 2008 ($0.059 \pm 0.021 \text{ mg L}^{-1}$), 2012 ($0.080 \pm 0.038 \text{ mg L}^{-1}$) and 2013 ($0.064 \pm 0.033 \text{ mg L}^{-1}$). Concentrations in 2012 were significantly lower than in 2009 ($0.046 \pm 0.025 \text{ mg L}^{-1}$), 2010 ($0.051 \pm 0.025 \text{ mg L}^{-1}$) and 2011 ($0.051 \pm 0.052 \text{ mg L}^{-1}$). This annual variation can be seen in Figure 5.12 (trend panel).

Mean seasonal whole-Bay surface TP concentrations varied seasonally between 2003 and 2014 in Cardiff Bay (Figure 5.12, seasonal panel). The concentrations of TP were significantly different between seasons ($F_{3,135} = 2.89$, $P < 0.05$), where TP concentrations in winter were

lower than those in summer ($0.052 \pm 0.06 \text{ mg L}^{-1}$). Mean monthly whole-Bay surface TP concentrations did not vary significantly between months.

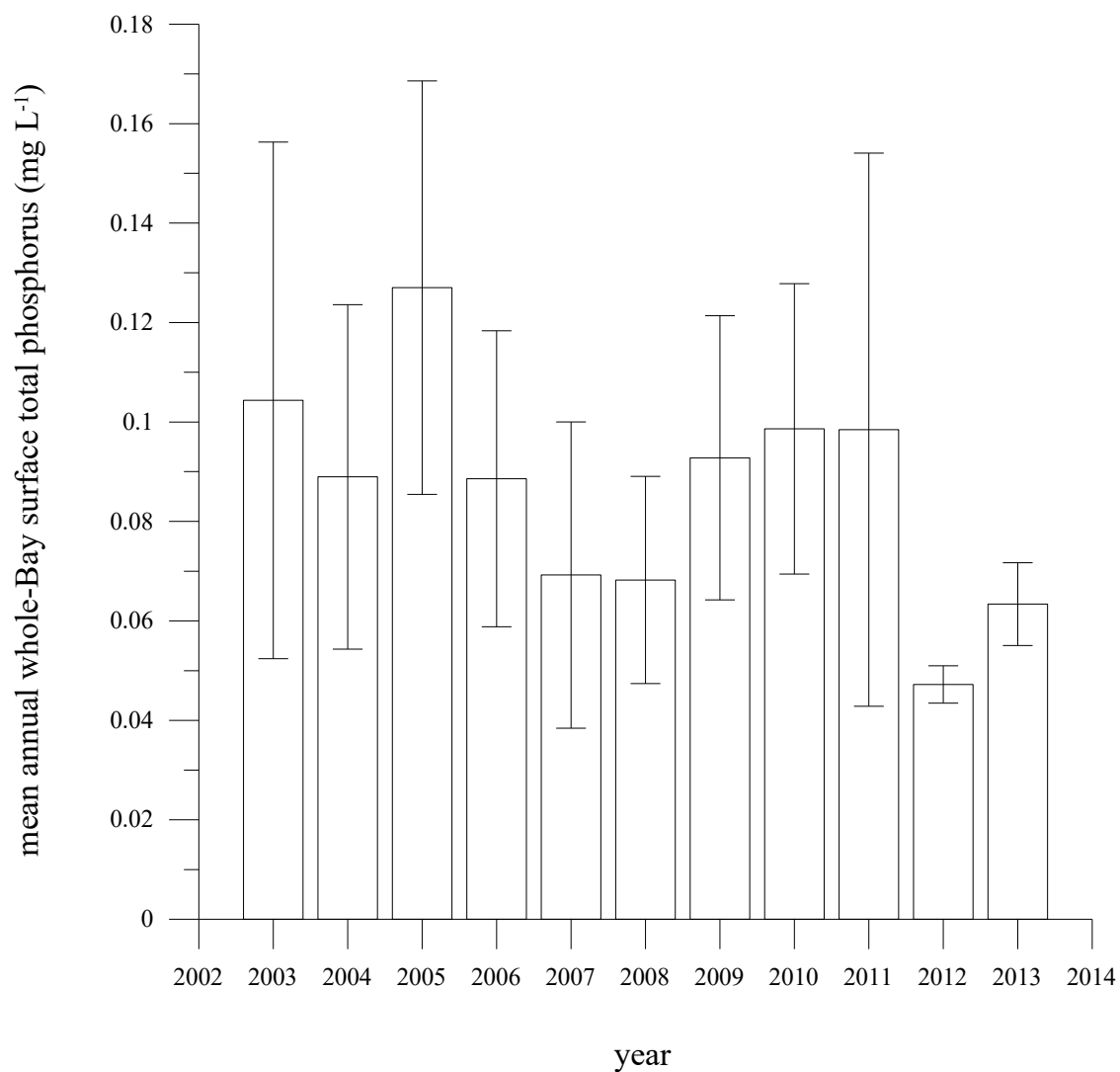


Figure 5.11: Mean annual whole-Bay surface total phosphorus concentrations in Cardiff Bay recorded between 2003 and 2013 (standard error bars).

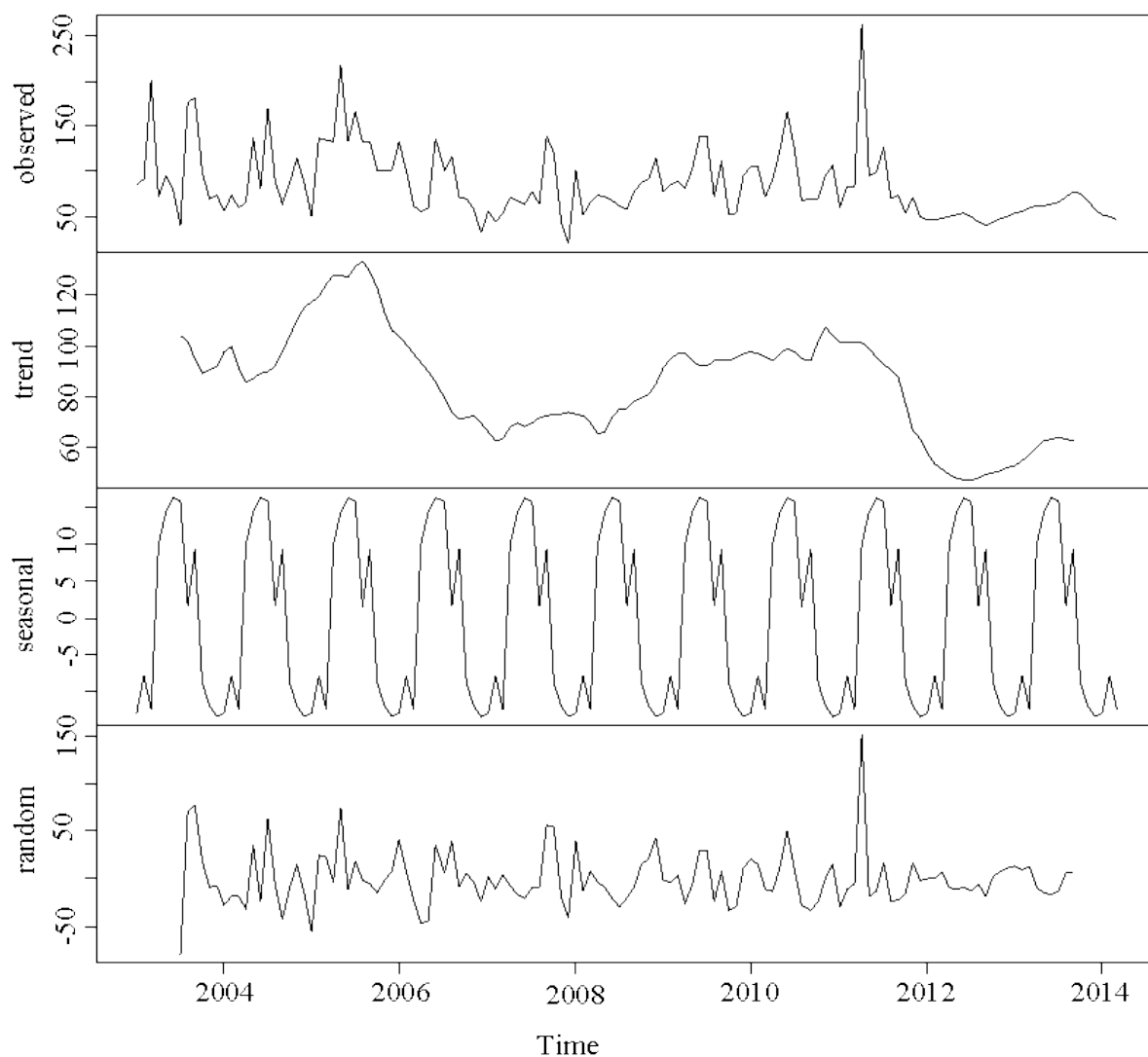


Figure 5.12: Time Series Analysis of mean monthly whole-Bay surface Total Phosphorus (TP) concentrations in Cardiff Bay, using monthly data collected from January 2003 until April 2014.

5.5.3 The Temporal Variation of Soluble Reactive Phosphorus in Cardiff Bay

There were significant differences between mean annual whole-Bay surface SRP concentrations in Cardiff Bay between 2003 and 2013 ($F_{10,131} = 5.31$, $P < 0.01$). SRP concentrations in 2005 were significantly higher than those in 2003 ($0.052 \pm 0.016 \text{ mg L}^{-1}$), 2007 ($0.037 \pm 0.005 \text{ mg L}^{-1}$), 2008 ($0.041 \pm 0.017 \text{ mg L}^{-1}$), 2011 ($0.037 \pm 0.012 \text{ mg L}^{-1}$), 2012

($0.061 \pm 0.029 \text{ mg L}^{-1}$) and 2013 ($0.054 \pm 0.027 \text{ mg L}^{-1}$). In 2010 concentrations were significantly higher than those in 2012 ($0.037 \pm 0.017 \text{ mg L}^{-1}$). This annual change over time can be seen in Figure 5.13 (trend panel). Despite Figure 5.13 (seasonal panel) indicating that there is a seasonal variation of SRP in Cardiff Bay, these differences are not statistically significant. There were no significant differences found between months using mean monthly whole-Bay surface SRP concentrations.

Mean monthly whole-Bay surface TP concentrations in Cardiff Bay were significantly correlated with mean monthly whole-Bay surface SRP concentrations ($r = 0.57$, $n = 134$, $P < 0.01$), however $r^2 = 0.32$, therefore the relationship was weak (Appendix Figure 5.1).

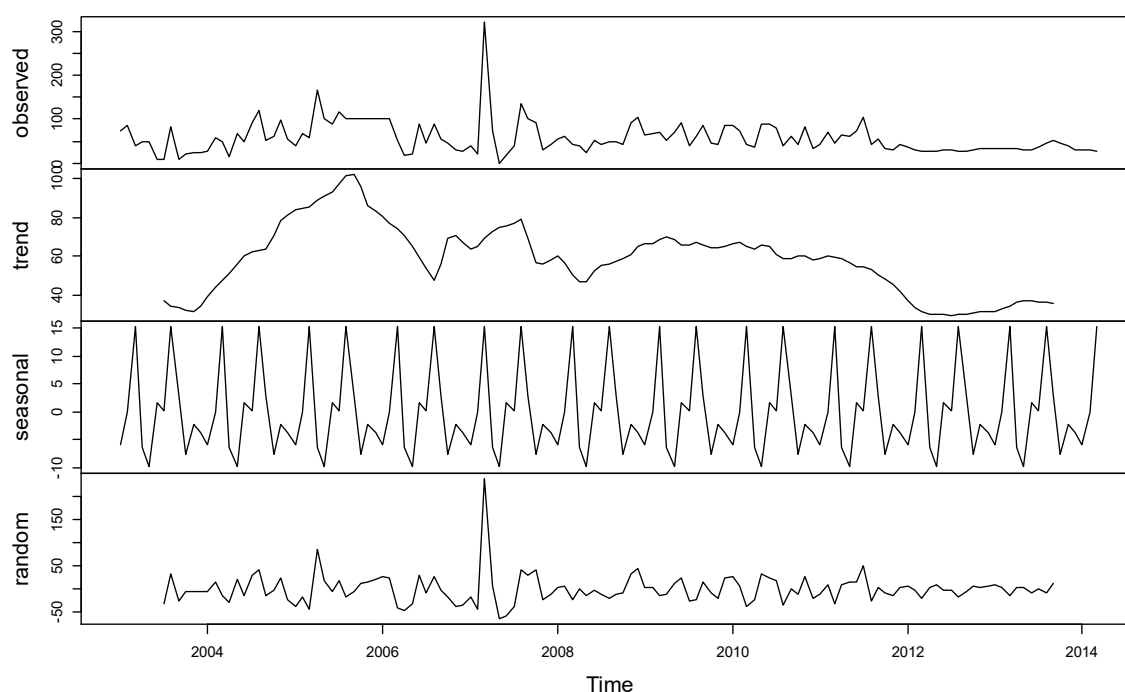


Figure 5.13: Time series analysis of mean monthly whole-Bay surface soluble reactive phosphorus (SRP) recorded in Cardiff Bay between January 2003 and April 2014.

5.5.5 The Temporal Variation of Total Dissolved Nitrogen in Cardiff Bay

The mean annual whole-Bay surface TDN concentration in Cardiff Bay varied significantly between years ($F_{10,131} = 5.96$, $P < 0.01$). Mean annual whole-Bay surface TDN concentrations in 2004 were significantly lower than the other years studied. TDN concentrations then rose

and were significantly higher in 2005 and 2006 than in 2008 and 2011. Figure 5.14 (trend panel) shows this annual variation between 2003 and 2013.

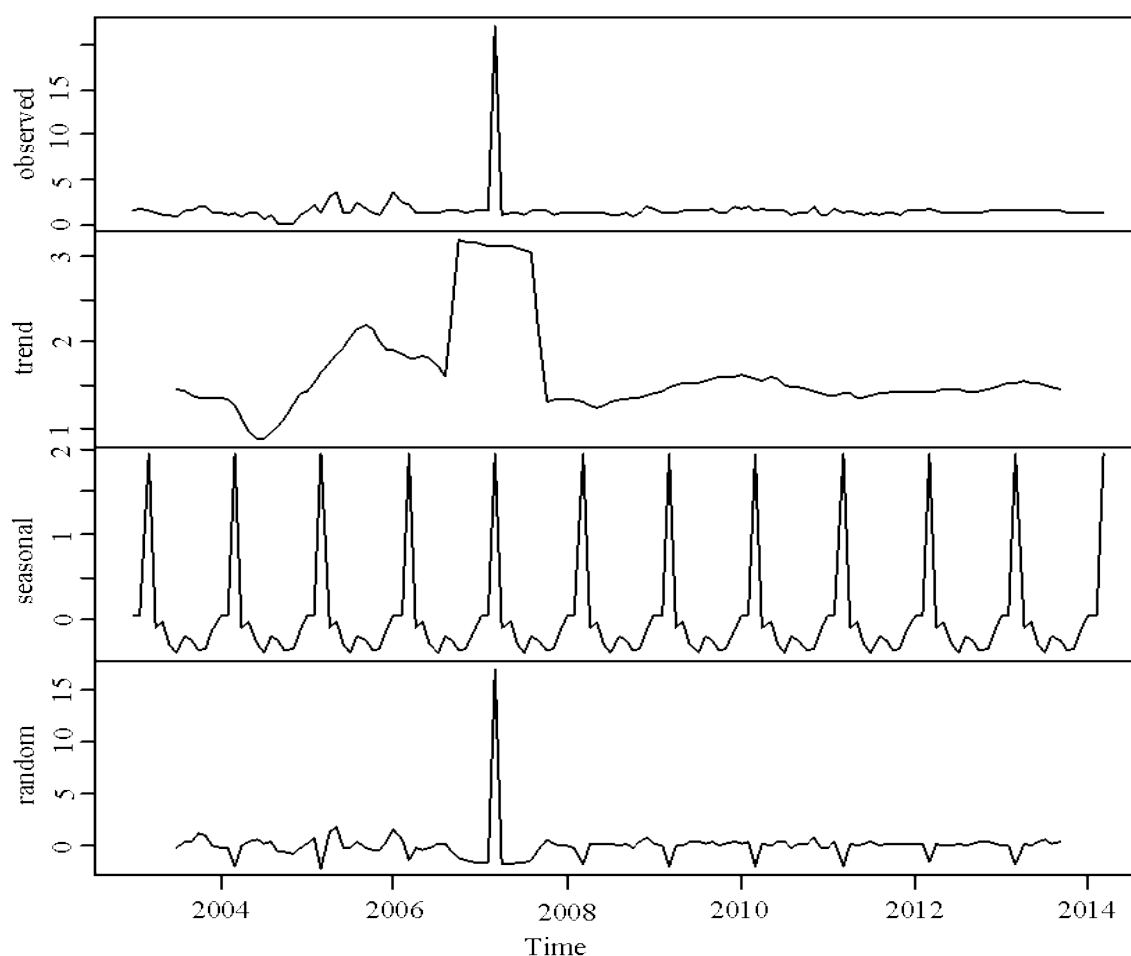


Figure 5.14: Time series analysis of mean monthly whole-Bay surface total dissolved nitrogen (TDN) recorded in Cardiff Bay between January 2003 and April 2014.

Seasonal variations in mean whole-Bay surface TDN concentrations were observed in Cardiff Bay (Figure 5.14, seasonal panel), where TDN concentrations peaked during summer and were at their lowest during winter. However, statistically there was no difference between seasons of TDN. There was also no statistical difference on a monthly basis between mean monthly whole-Bay surface TDN. The mean monthly whole-Bay surface TDN:TP ratio only went above 16:1 during 2 months between January 2003 and April 2014 (Figure 5.15). Between April 2012 and April 2014, the balance of TDN:TP was higher than 16:1 on 2 out of the 50 days it was studied (Figure 5.16).

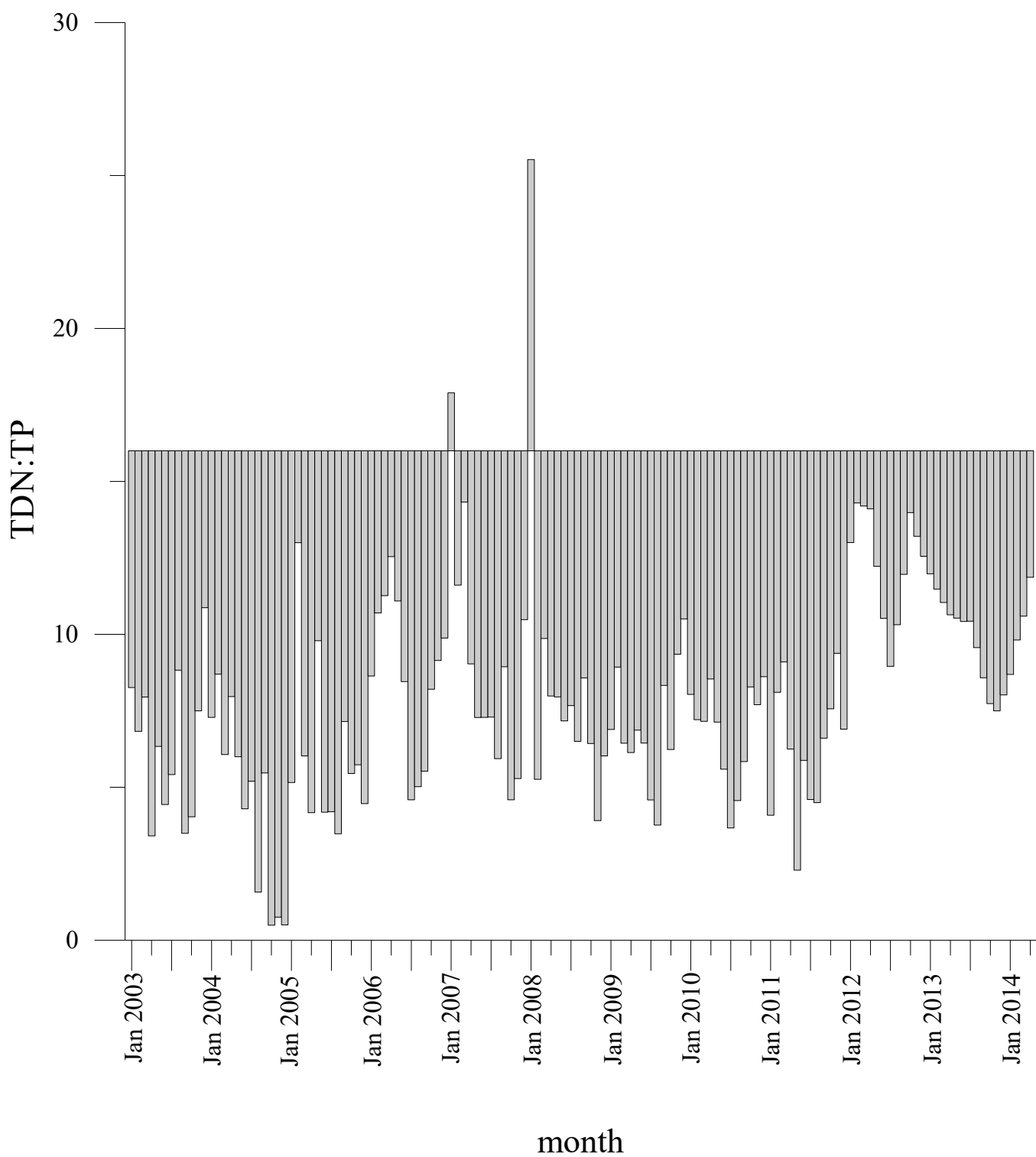


Figure 5.15: Mean monthly whole-Bay surface total dissolved nitrogen : total phosphorus in Cardiff Bay from January 2003 until April 2014. Base line at 16:1 TDN:TP.

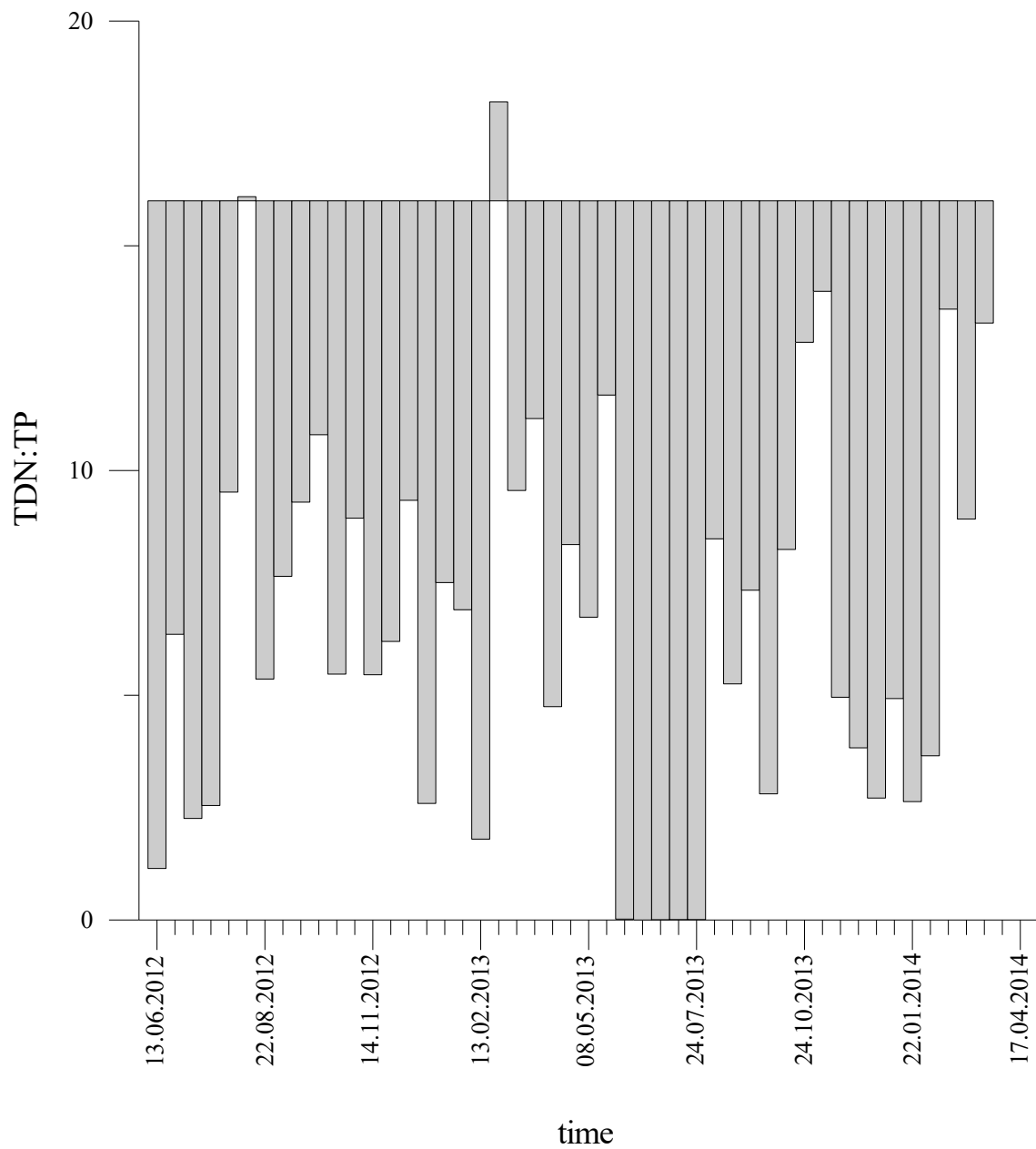


Figure 5.16: Mean whole-Bay surface total dissolved nitrogen : total phosphorus in Cardiff Bay, recorded fortnightly from 13th June 2012 until 30th April 2014.

5.5.5 The Temporal Variation of Ammonium in Cardiff Bay

The concentration of mean annual whole-Bay surface ammonium in Cardiff Bay varied between 2003 and 2013 ($F_{10,131} = 6.14$, $P < 0.01$). The mean annual whole-Bay surface ammonium concentrations in Cardiff Bay rose with time; concentrations in 2003 and 2004 were significantly lower than the concentrations experienced in 2005 and 2006. The concentration of ammonium in 2007, 2008, 2009, 2010, 2011 and 2012 were significantly higher than those in 2005 and 2006. The increase in ammonium concentration in Cardiff Bay over time can be seen in Figure 5.17 (trend panel).

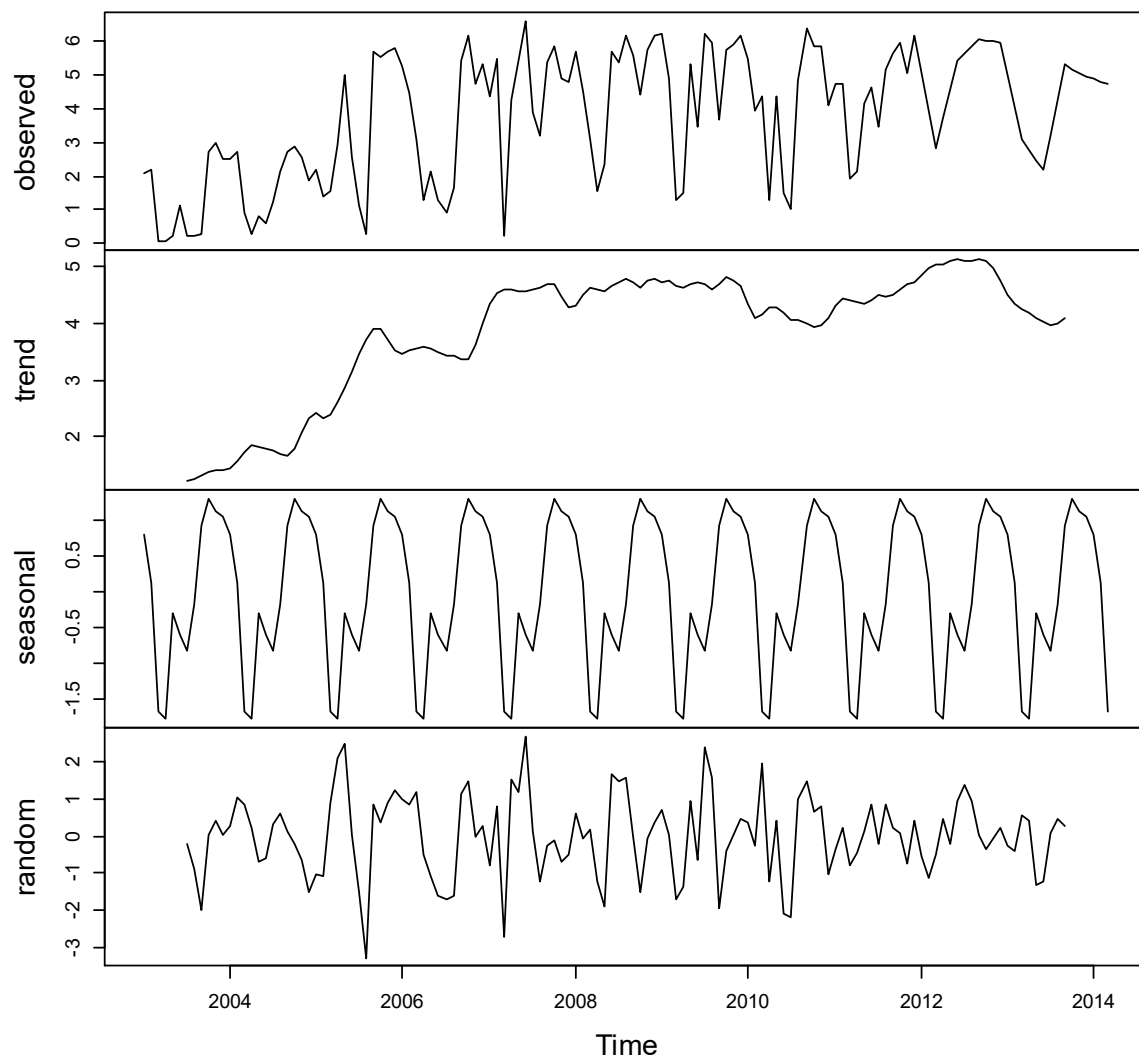


Figure 5.17: Time series analysis of mean whole-Bay monthly surface ammonium recorded in Cardiff Bay between January 2003 and April 2014.

Visually, ammonium concentrations varied seasonally in Cardiff Bay (Figure 5.17, seasonal panel), however this variation was not statistically significant. There was also no statistical difference on a monthly basis between mean monthly whole-Bay surface ammonium.

Mean monthly whole-Bay surface ammonium concentrations in Cardiff Bay were significantly correlated with mean monthly whole-Bay surface TDN concentrations ($r = 0.5$, $n = 134$, $P < 0.01$) but $r^2 = 0.25$, therefore the relationship was weak (Appendix Figure 5.2).

5.5.3.3 The Temporal Variation of Nitrate in Cardiff Bay

There were significant differences between years of mean annual whole-Bay surface nitrate concentrations in Cardiff Bay ($F_{10,131} = 4.56$, $P < 0.01$), between 2003 and 2013. The mean annual whole-Bay surface nitrate concentration in 2003 was significantly higher than in 2004. The concentration in 2004 was significantly lower than in 2005, 2006, 2009, 2010, 2012 and 2013, and the nitrate concentration in 2006 was significantly higher than 2008 (Figure 5.18, trend panel).

Visually, there were significant differences between seasons of mean seasonal whole-Bay surface nitrate concentrations in Cardiff Bay (Figure 5.18, seasonal panel), where spring nitrate concentrations were higher than those in summer. However statistically the differences were weak ($F_{3,135} = 3.54$, $P < 0.05$). There was no statistical difference between months using mean monthly whole-Bay surface nitrate concentrations.

Mean monthly whole-Bay surface nitrate concentrations in Cardiff Bay were significantly correlated with mean monthly whole-Bay surface TDN concentrations (Figure 5.19).

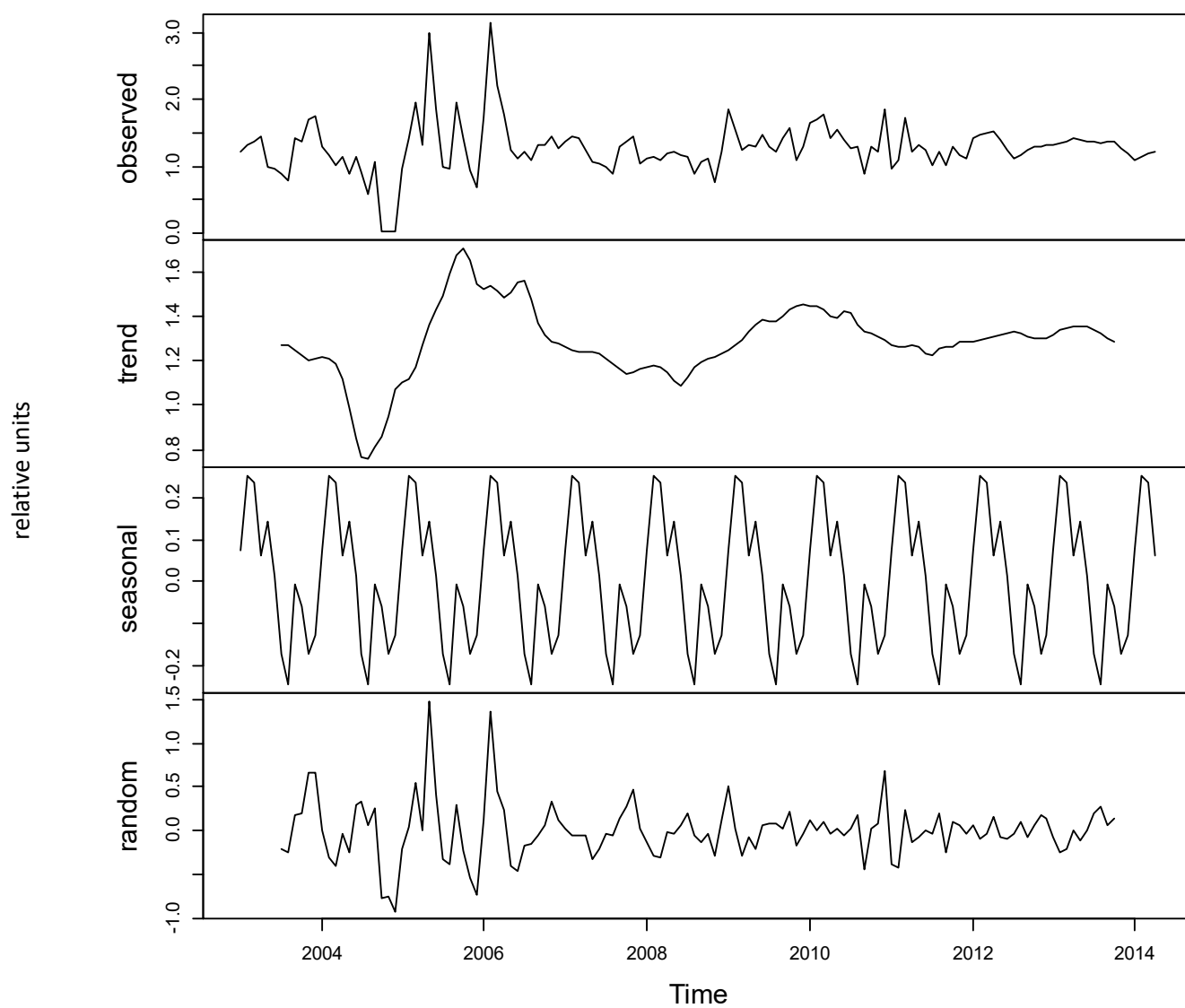


Figure 5.18: Time series analysis of mean monthly whole-Bay surface nitrate recorded in Cardiff Bay between January 2003 and April 2014.

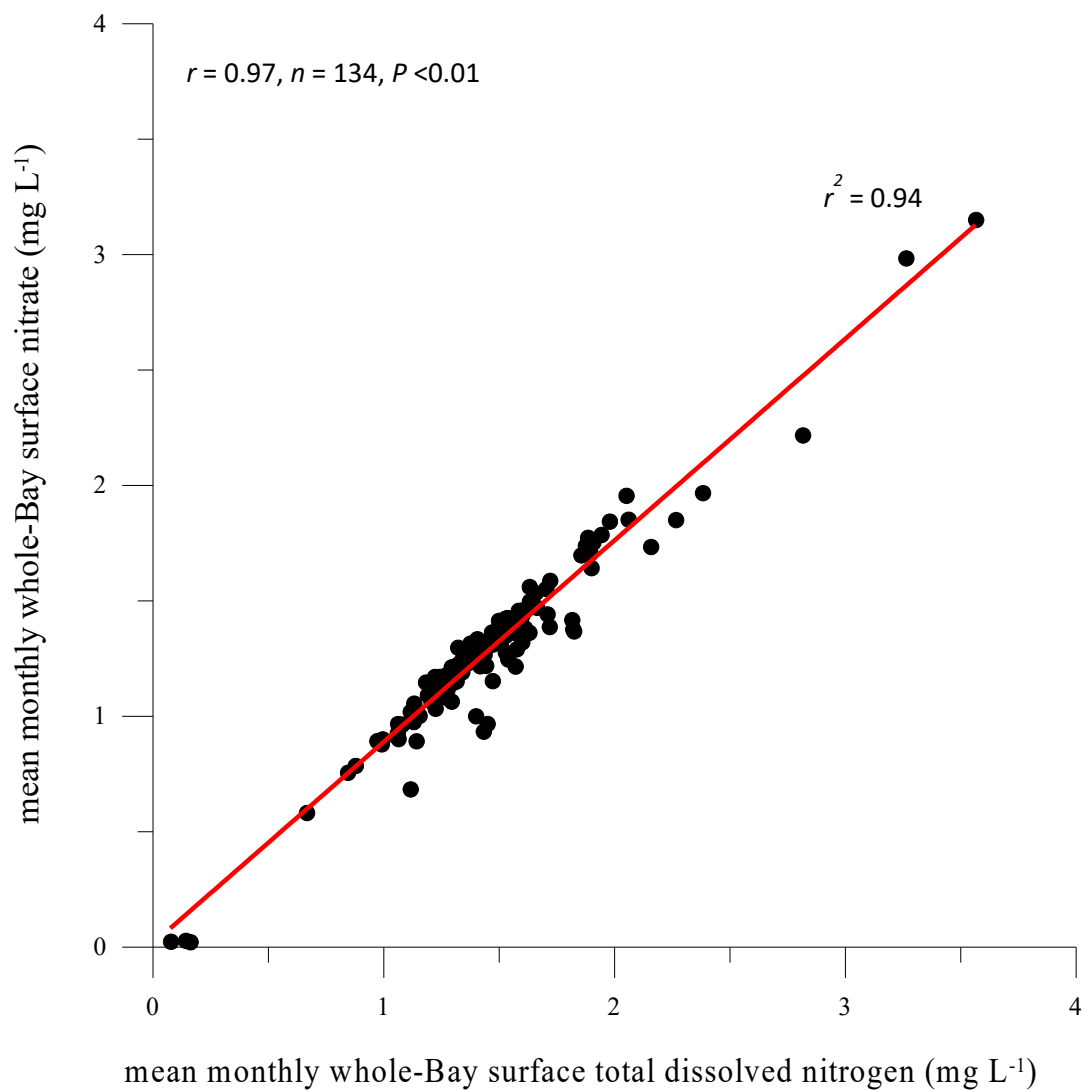


Figure 5.19: Mean monthly whole-Bay surface total dissolved nitrogen concentrations versus mean monthly whole-Bay surface nitrate concentrations in Cardiff Bay, taken from January 2003 until April 2014.

5.5.7 The Temporal Variation of Nitrite in Cardiff Bay

The nitrite concentration of Cardiff Bay was significantly different between 2003 and 2013 ($F_{10,131} = 6.11$, $P < 0.01$), using mean annual whole-Bay surface nitrite concentrations. 2003 nitrite concentrations were significantly higher than those in 2004, 2007, 2008, 2009, 2010, 2011, 2012 and 2013. The concentrations in 2004 were significantly lower than those in 2005. In 2005 nitrite concentrations were significantly higher than in 2008, 2009, 2010, 2011, 2012 and 2013 (Figure 5.20, trend panel).

Despite the visual seasonal trend shown in Figure 5.20 (seasonal panel), statistically there was no difference in mean seasonal whole-Bay surface nitrite concentrations. There was no statistically significant difference between months using mean monthly whole-Bay surface nitrite concentrations.

Mean monthly whole-Bay surface nitrite concentrations in Cardiff Bay were significantly correlated with mean monthly whole-Bay surface TDN concentrations ($r = 0.37$, $n = 134$, $P < 0.01$), however $r^2 = 0.14$, therefore the relationship was weak.

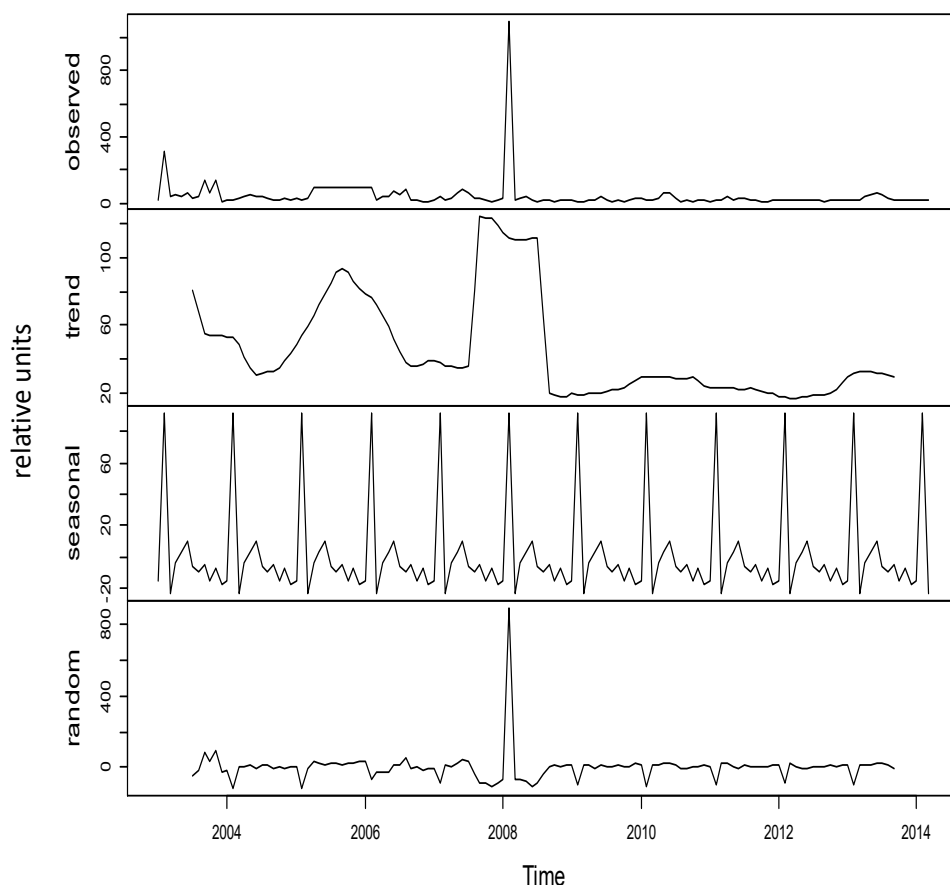


Figure 5.20: Time series analysis of mean monthly whole-Bay surface nitrite recorded in Cardiff Bay between January 2003 and April 2014.

5.5.8 The Temporal Variation of Silica in Cardiff Bay

The concentration of mean annual whole-Bay surface silica in Cardiff Bay significantly varied between years ($F_{10,131} = 7.76$, $P < 0.001$). Concentrations in 2003 were significantly lower than those from 2006 to 2013, 2004 had significantly lower concentrations than 2007 to 2013, and concentrations in 2005 were significantly lower than 2012 (Figure 5.21, trend panel).

Between 2003 and 2014, mean seasonal whole-Bay surface silica concentrations significantly varied seasonally ($F_{3,135} = 10.78$, $P < 0.01$), (Figure 5.21, seasonal panel). Silica concentrations peaked in winter (averaging $4.62 \pm 0.53 \text{ mg L}^{-1}$) and concentrations were significantly higher during winter than those in spring and summer. On average, winter silica concentrations were

$1.99 \pm 0.68 \text{ mg L}^{-1}$ higher than those in spring and $1.47 \pm 0.94 \text{ mg L}^{-1}$ higher than summer. Spring and summer concentrations were significantly lower than those in autumn, with concentrations reaching their lowest levels during spring (averaging $2.63 \pm 0.27 \text{ mg L}^{-1}$). Concentrations experienced a rise and a fall during summer.

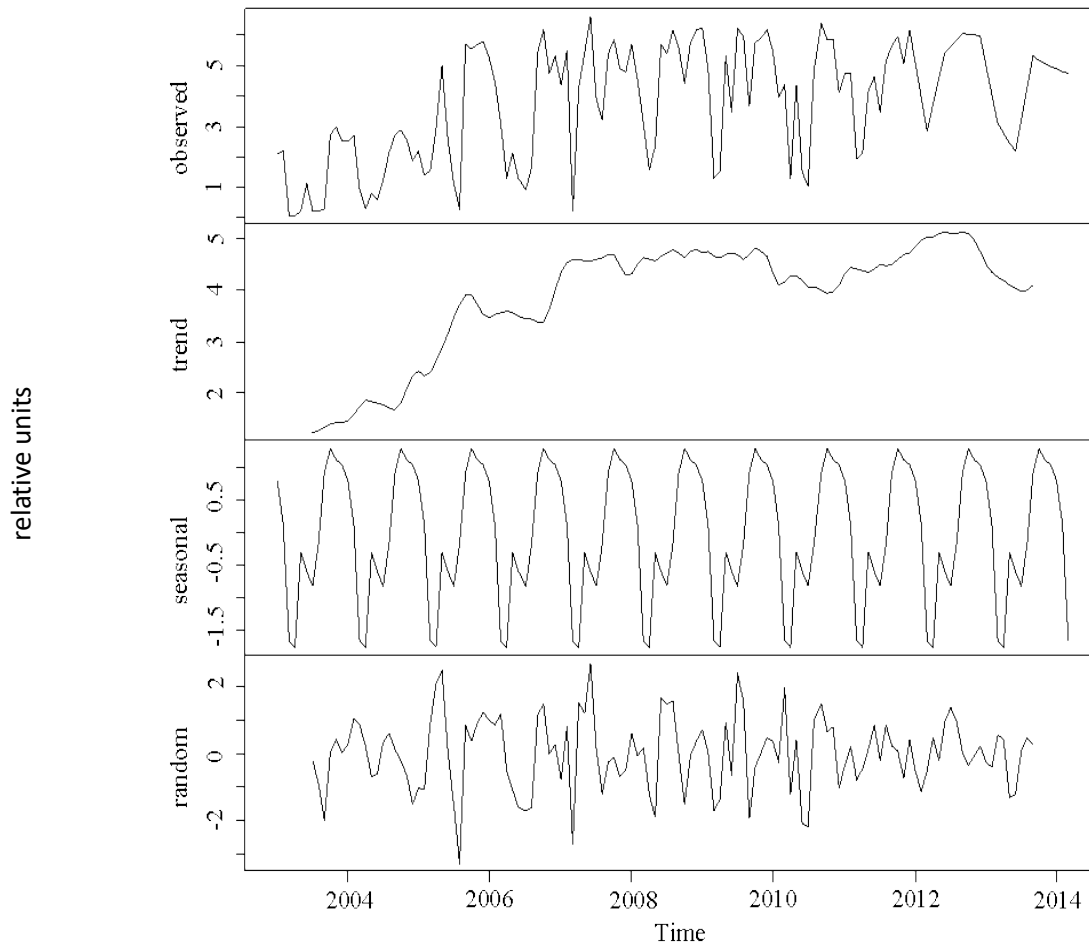


Figure 5.21: Time series analysis of mean monthly whole-Bay surface silica in Cardiff Bay between January 2003 and April 2014.

Mean monthly whole-Bay surface silica concentrations were significantly different ($F_{11,135} = 4.46$, $P < 0.01$). Mean monthly whole-Bay surface silica concentrations were higher in January than April and May. February silica concentrations were higher than those experienced in May. May concentrations were lower than those in October, November and December.

Between January 2003 and April 2014 TP:Si was above 1:1 for 28 out of the 136 months. This occurred most commonly during the month of May (Figure 5.22). Data from April 2012 to

April 2014 indicated that TP:Si did not fall below 1:1 at any time (Figure 5.23). Between January 2003 and April 2014 TDN:Si only reached 16:1 and above during April 2003 (Figure 5.24). Between April 2012 and April 2014, TDN:Si was above 16:1 the majority of the time. The only dates that TDN:Si was less than 16:1 was on 11th July 2012 and from 17th July 2013 until 4th September 2013 (Figure 5.25).

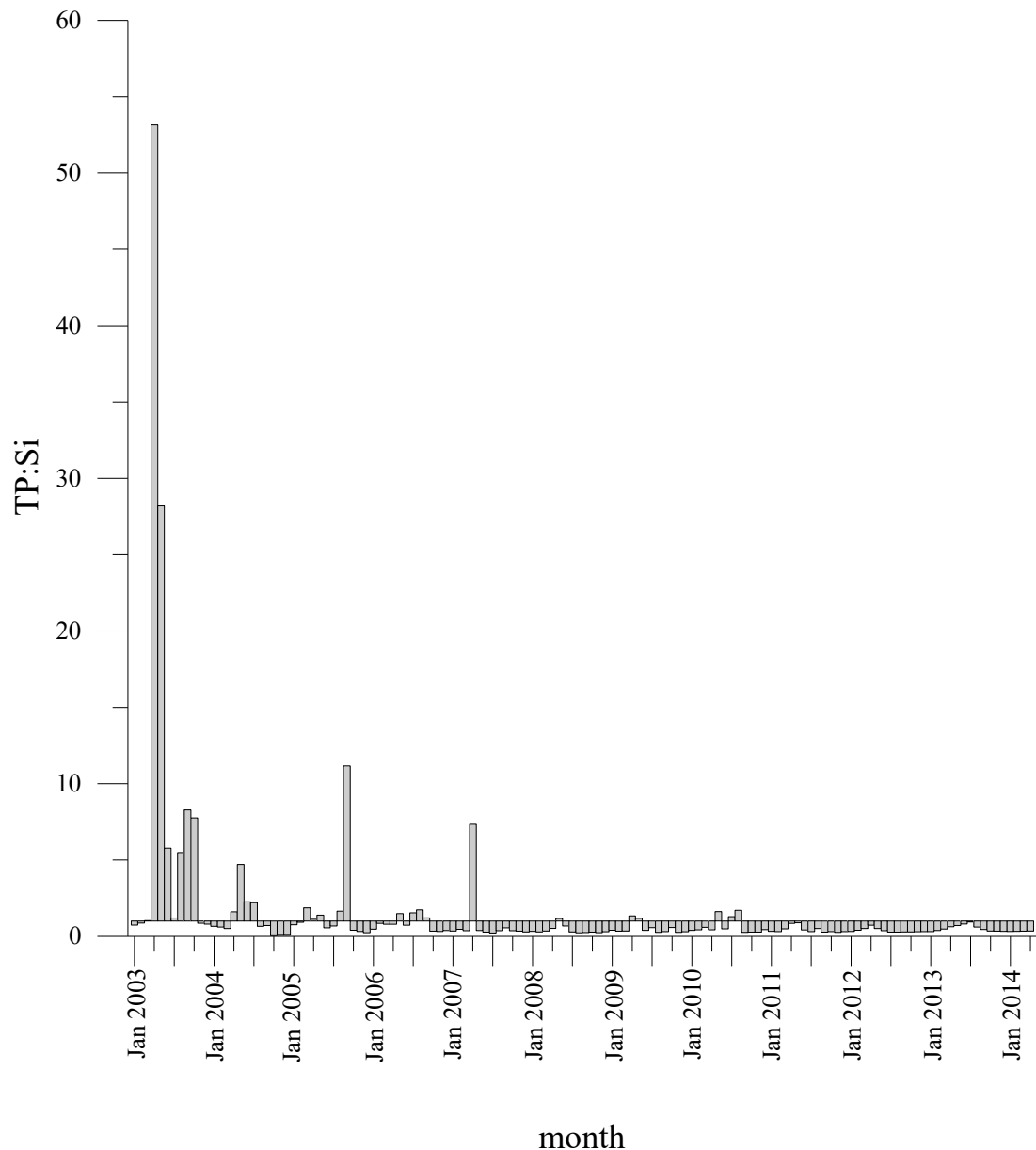


Figure 5.22: Mean monthly whole-Bay surface total phosphorus : silica in Cardiff Bay from January 2003 until April 2014. Base line at 1:1 TP:Si.

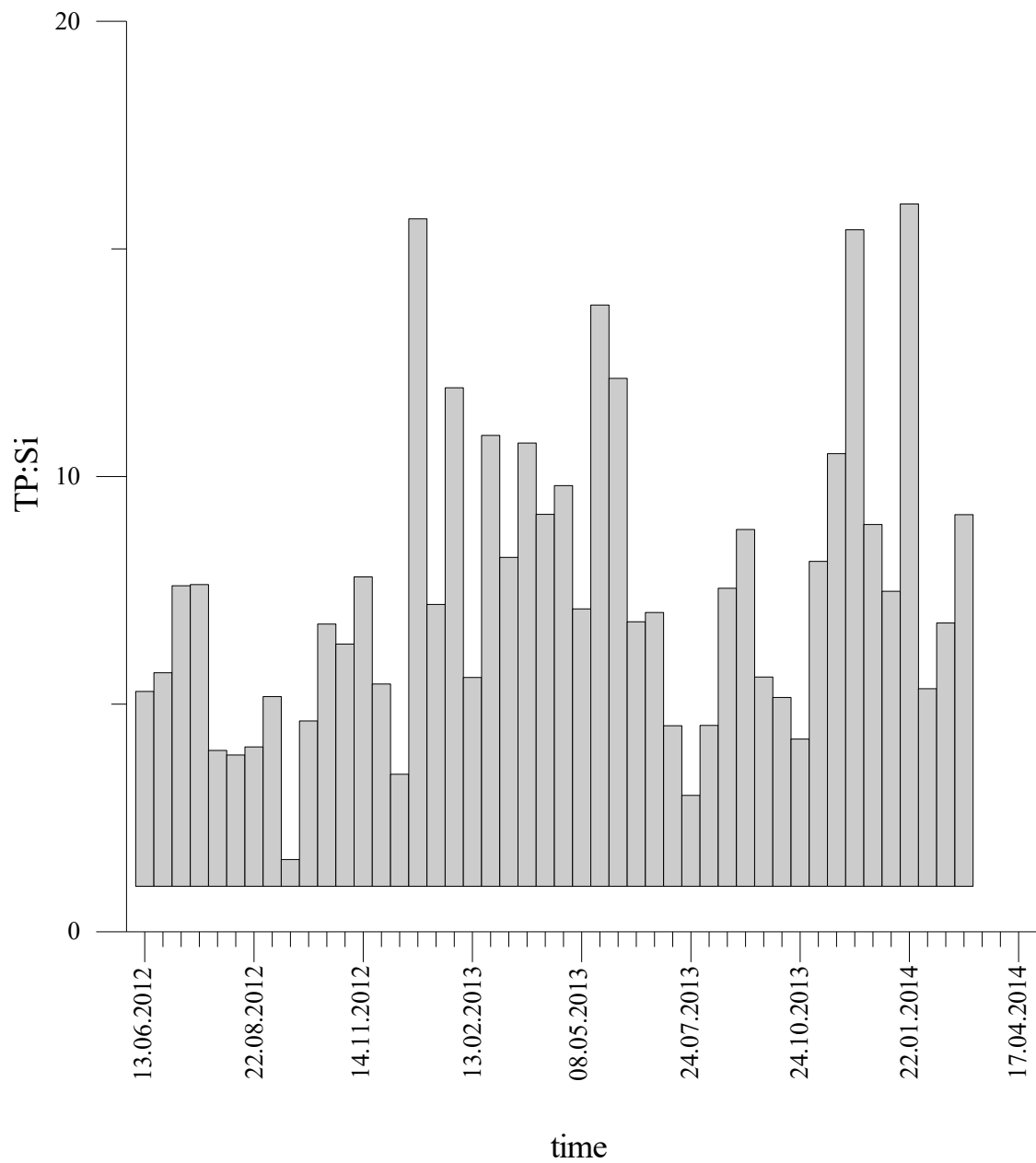


Figure 5.23: Mean monthly whole-Bay surface total phosphorus : silica in Cardiff Bay, recorded fortnightly from 13th June 2012 until 30th April 2014. Baseline at 1:1 TP:Si.

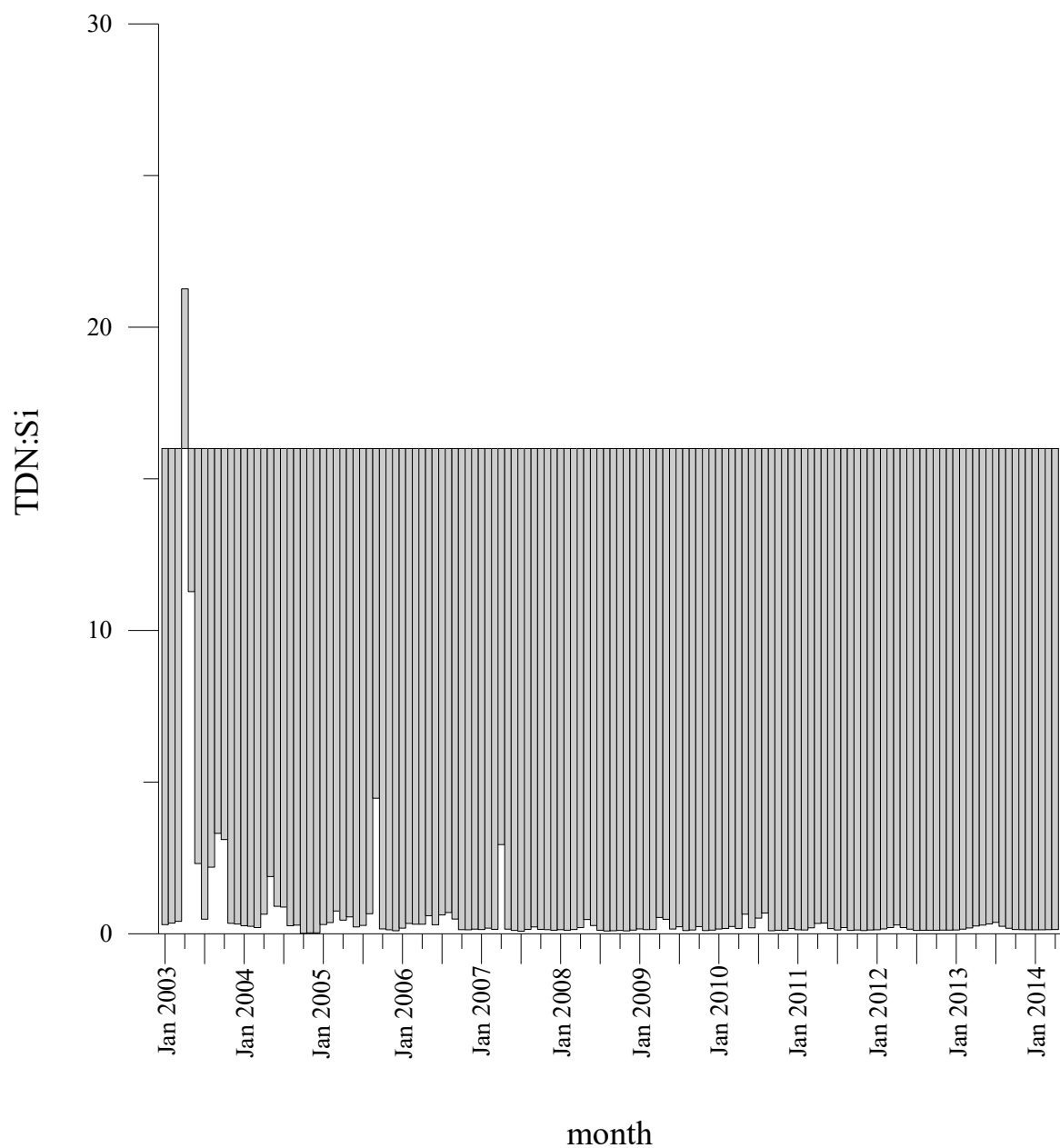


Figure 5.24: Mean monthly whole-Bay surface total dissolved nitrogen: silica in Cardiff Bay from January 2003 until April 2014. Base line at 16:1 TDN:Si.

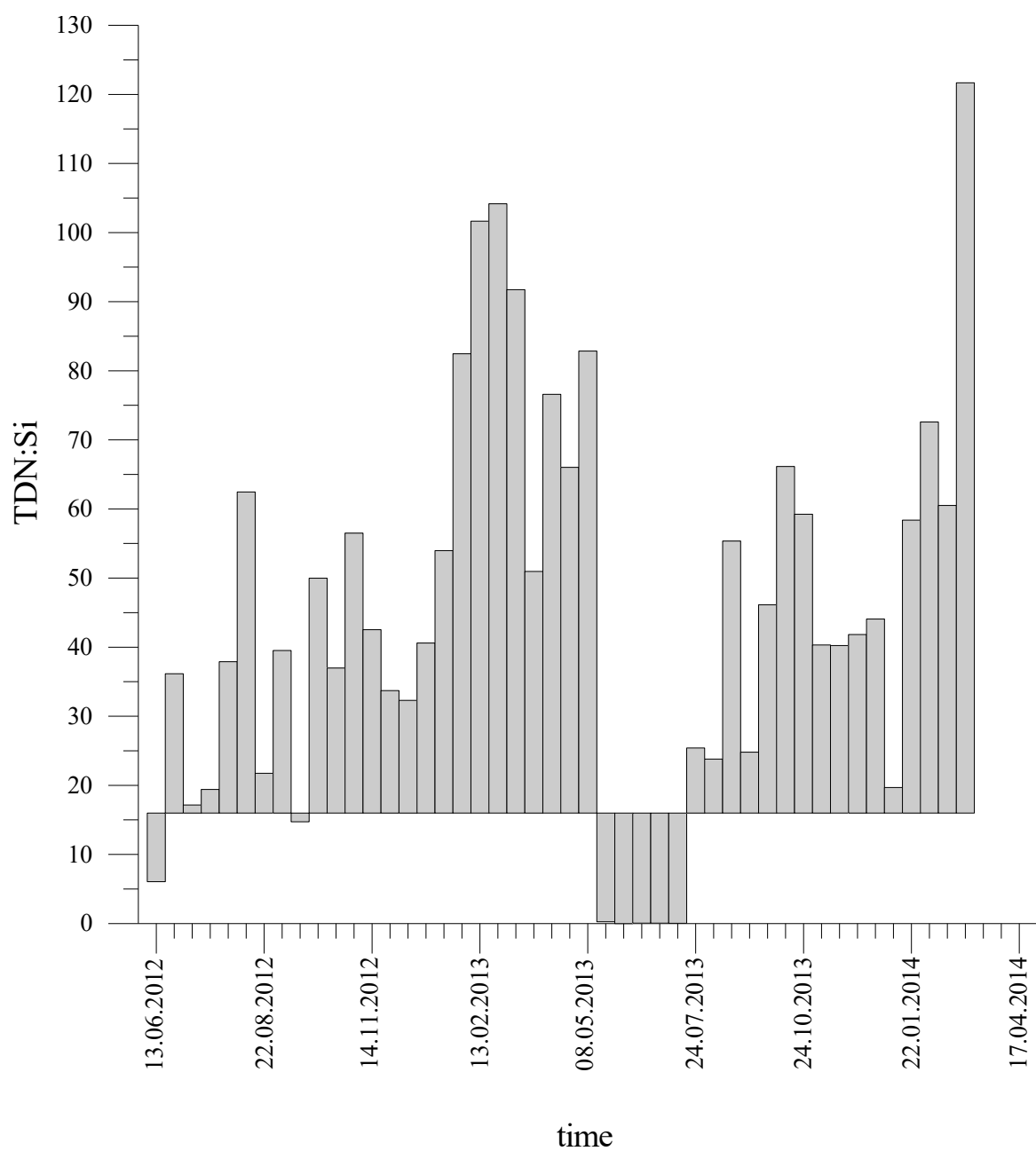


Figure 5.25: Mean monthly whole-Bay surface total dissolved nitrogen : silica in Cardiff Bay, recorded fortnightly from 13th June 2012 until 30th April 2014. Baseline at 16:1 TDN:Si.

5.5.9 The Relationship of Phosphorus, Nitrogen & Silica with the Environmental Factors of Cardiff Bay

The environmental factors used in this section were analysed independently in Chapter 3.

5.5.9.1 Significant Relationships between Nutrients & Environmental Variables in Cardiff Bay

Mean monthly precipitation had a significant positive relationship with mean monthly whole-Bay surface ammonium ($r = 0.75$, $n = 136$, $P < 0.01$), nitrate ($r = 0.90$, $n = 136$, $P < 0.01$), TDN ($r = 0.93$, $n = 136$, $P < 0.01$), and SRP ($r = 0.93$, $n = 136$, $P < 0.01$) concentrations.

Annually, mean combined river discharge significantly negatively correlated with mean annual whole-Bay surface TP ($r = 0.77$, $n = 9$, $P < 0.01$). During summer, mean whole-Bay surface silica was significantly negatively correlated with mean summer combined river discharge ($r = 0.74$, $n = 9$, $P < 0.01$).

The mean annual flushing rate of Cardiff Bay had a significant relationship with mean annual whole-Bay surface nitrite, negatively ($r = 0.74$, $n = 9$, $P < 0.01$). During summer, mean seasonal lake flushing rate had a negative significant relationship with mean summer whole-Bay surface silica ($r = 0.85$, $n = 19$, $P < 0.01$).

Multiple regression was carried out to determine if summer combined river discharge or summer lake flushing rate was the controlling variable for mean summer whole-Bay surface silica. Mean summer lake flushing rate influenced 71% of the variation in summer silica concentrations (Equation A). Combined river discharge was insignificant.

Equation A: (*silica*) = $8.6 + 4.7$ (*flushing rate*); $F_{1,9} = 22.5$, $P < 0.01$, $r^2 = 0.71$.

5.5.9.2 Weak Relationships between Nutrients & Environmental Variables in Cardiff Bay

Mean monthly air temperature significantly positively correlated with mean monthly whole-Bay surface ammonium concentrations but the relationship was weak (Appendix Figure 5.3).

Mean annual surface water temperature had a significant positive relationship with mean annual whole-Bay surface SRP ($r = 0.63$, $n = 9$, $P < 0.05$) and mean annual whole-Bay surface TP ($r = 0.67$, $n = 9$, $P < 0.05$) concentrations. Seasonally, mean winter whole-Bay surface water temperature and mean winter whole-Bay surface SRP had a significantly negative relationship ($r = 0.6$, $n = 10$, $P < 0.05$). Mean winter whole-Bay surface water temperature also had a significantly negative relationship with mean winter whole-Bay surface TP ($r = 0.65$, $n = 10$, $P < 0.05$).

Mean annual precipitation has a significantly negative relationship with mean annual whole-Bay surface TP ($r = 0.71$, $n = 9$, $P < 0.05$). Mean winter precipitation has a weak negative relationship with mean winter whole-Bay surface TP ($r = 0.69$, $n = 10$, $P < 0.05$) and mean winter whole-Bay surface SRP ($r = 0.7$, $n = 10$, $P < 0.05$). 49% of the changes experienced in SRP concentrations during winter could be explained by precipitation ($F_{1,10} = 9.74$, $P < 0.05$); $SRP = 329.28 + 2420.24 \times \text{Precipitation}$; $r^2 = 0.49$.

Mean annual combined river discharge was weakly negatively significantly correlated with mean annual whole-Bay surface nitrite ($r = 0.69$, $n = 9$, $P < 0.01$) and silicate ($r = 0.61$, $n = 9$, $P < 0.01$). During summer, mean whole-Bay surface nitrite was also weakly significantly negatively correlated with mean summer combined river discharge ($r = 0.63$, $n = 9$, $P < 0.01$).

The mean annual flushing rate of Cardiff Bay had a weak significant relationship with mean annual whole-Bay surface TP, negatively ($r = 0.68$, $n = 9$, $P < 0.05$) and silica, positively ($r = 0.65$, $n = 10$, $P < 0.05$). Mean autumn lake flushing rate had a weak positive significant relationship with mean autumn whole-Bay surface TP ($r = 0.68$, $n = 9$, $P < 0.05$). Mean monthly lake flushing rate weakly negatively significantly correlated with mean monthly whole-Bay surface silica ($r = 0.61$, $n = 136$, $P < 0.01$).

Air temperature did not significantly influence any surface nutrient concentrations within Cardiff Bay on an annual or seasonal scale. On a monthly basis there was no statistical relationship between mean whole-Bay surface water temperature and mean whole-Bay surface nutrient concentrations within Cardiff Bay. Mean monthly whole-Bay surface nutrients did not have a significant relationship with mean monthly combined river discharge.

5.5.10 The Relationship of Phosphorus, Nitrogen and Silica with the Phytoplankton Biomass of Cardiff Bay

The phytoplankton biomass data used in this section were analysed independently in Chapter 4.

5.5.10.1 Significant Relationships between Nutrients & Phytoplankton Biomass in Cardiff Bay

During spring, mean whole-Bay surface silica had a significant positive relationship with mean whole-Bay surface diatom chlorophyll *a* ($r = 0.77$, $n = 9$, $P < 0.01$). Diatom chlorophyll *a* also had a significant positive relationship using mean monthly whole-Bay surface data with nitrate ($r = 0.86$, $n = 136$, $P < 0.01$) and TDN ($r = 0.99$, $n = 136$, $P < 0.01$).

Mean monthly whole-Bay surface chorophytes had a significant positive relationship with mean monthly whole-Bay surface SRP ($r = 0.99$, $n = 136$, $P < 0.01$).

Seasonal whole-Bay surface cryptophytes had a significant positive relationship with mean seasonal whole-Bay surface silica during summer ($r = 0.89$, $n = 9$, $P = < 0.01$). Mean monthly whole-Bay surface cryptophytes had a significant positive relationship with mean monthly whole-Bay surface ammonium ($r = 0.98$, $n = 136$, $P < 0.01$); nitrate ($r = 0.97$, $n = 136$, $P < 0.01$); SRP ($r = 0.84$, $n = 136$, $P < 0.01$); and TDN ($r = 0.96$, $n = 136$, $P < 0.01$). However, multiple regression determined all of the monthly relationships between cryptophytes and nutrients as insignificant.

Mean annual whole-Bay surface cyanobacteria was negatively correlated with mean annual whole-Bay surface silica ($r = 0.81$, $n = 9$, $P = <0.01$). Mean seasonal whole-Bay surface cyanobacteria had a significant relationship with mean seasonal whole-Bay surface TP ($r = 0.79$, $n = 9$, $P = <0.01$) and silica ($r = 0.8$, $n = 9$, $P = <0.01$) during winter. Multiple regression determined that there was an insignificant relationship between winter cyanobacteria with both TP and silica concentrations.

5.5.10.2 Weak Relationships between Nutrients & Phytoplankton Biomass in Cardiff Bay

Mean annual whole-Bay surface total chlorophyll *a* was negatively correlated with mean annual whole-Bay surface silica ($r = 0.67$, $n = 9$, $P = <0.05$). During spring, mean whole-Bay surface chlorophyll *a* has a significant positive relationship with silica ($r = 0.73$, $n = 9$, $P = <0.05$). Mean winter whole-Bay surface chlorophyll *a* had a significant positive relationship with mean winter whole-Bay surface SRP ($r = 0.68$, $n = 10$, $P = <0.05$), TP ($r = 0.7$, $n = 10$, $P = <0.05$). Autumn mean whole-Bay surface chlorophyll *a* had a significant positive relationship with TP ($r = 0.66$, $n = 9$, $P = <0.05$) and silica ($r = 0.65$, $n = 9$, $P = <0.05$).

Mean seasonal whole-Bay surface diatoms had a significant relationship with mean seasonal whole-Bay surface TP during winter ($r = 0.69$, $n = 10$, $P = <0.05$) and nitrate during autumn ($r = 0.61$, $n = 9$, $P = <0.05$). Mean annual whole-Bay surface chlorophytes was negatively correlated with mean annual whole-Bay surface silica ($r = 0.66$, $n = 9$, $P = <0.05$). Mean seasonal whole-Bay surface chlorophytes had a significant relationship with mean seasonal whole-Bay surface SRP during winter ($r = 0.62$, $n = 10$, $P = <0.05$), nitrite ($r = 0.66$, $n = 9$, $P = <0.05$), silica ($r = 0.73$, $n = 9$, $P = <0.05$), and TP ($r = 0.65$, $n = 9$, $P = <0.05$) during autumn. Mean annual whole-Bay surface cyanobacteria was negatively correlated with mean annual whole-Bay surface nitrate ($r = 0.7$, $n = 9$, $P = <0.05$) and TDN ($r = 0.63$, $n = 9$, $P = <0.05$).

There was no significant relationship between mean annual whole-Bay surface diatoms or cryptophytes and mean annual whole-Bay surface nutrients in Cardiff Bay.

Mean monthly whole-Bay surface total chlorophyll *a* did not have any relationship with mean monthly whole-Bay surface phosphorus, nitrogen or silica in Cardiff Bay between January

2003 and April 2014. Mean monthly whole-Bay surface cyanobacteria did not have a significant relationship with any of the mean monthly whole-Bay surface nutrients surveyed.

5.5.11 Determining Nutrient Limitation of Phytoplankton in Cardiff Bay

During spring, diatom chlorophyll *a* was slightly influenced by silica; 57% of the changes in the mean spring whole-Bay surface diatom chlorophyll *a* (Equation A) were caused by mean whole-Bay surface silica between 2003 and 2013. No other phytoplankton chlorophyll *a* group were significantly influenced by silica during spring and summer.

Equation A: (*diatom chlorophyll a*) = $22.2 + 5.2 (\text{silica})$; $F_{1,9} = 11.89$, $P < 0.01$, $r^2 = 0.57$.

The mean whole-Bay surface total chlorophyll *a* concentration was influenced by mean whole-Bay surface nitrate, silica and SRP concentrations during autumn (Equation B).

Equation B: (*total chlorophyll a*) = $6.3 + 5.2 (\text{nitrate}) + 1.4 (\text{silica}) + 43.6 (\text{soluble reactive phosphorus})$; $F_{3,7} = 41.9$, $P < 0.01$, $r^2 = 0.95$. Nitrate ($P < 0.01$). Silica ($P < 0.01$). SRP ($P < 0.01$).

Mean winter whole-Bay surface cyanobacteria chlorophyll *a* concentration had a weak negative relationship with SRP and a weak positive relationship with TP (Equation E). There were no other significant relationships during winter between nutrient concentrations and chlorophyll *a*.

Equation C: (*cyanobacteria chlorophyll a*) = $0.1 - 3.8 (\text{soluble reactive phosphorus}) + 4.8 (\text{total phosphorus})$; $F_{2,9} = 14.12$, $P < 0.05$, $r^2 = 0.49$. SRP & TP ($P < 0.05$).

5.5.11.1 Nutrient Addition Experiments

Analysis of the mesocosm study determined that after 7 days of incubation, there was no statistical difference between the chlorophyll-*a*, *b*, or *c* concentrations within the control, phosphorus or nitrogen addition mesocosm bags.

Post nutrient removal from input sources on the River Taff and River Ely in March 2005 and March 2006, there was the visual indication that TP and chlorophyll *a* reduced in concentration

(Figure 5.26, trend panel). However, ANOVA indicated that there was no significant difference within years. This data was pooled to compare pre and post 2006 (Mann Whitney U test), which did not indicate any differences between years despite a visual reduction in TP in the most recent years.

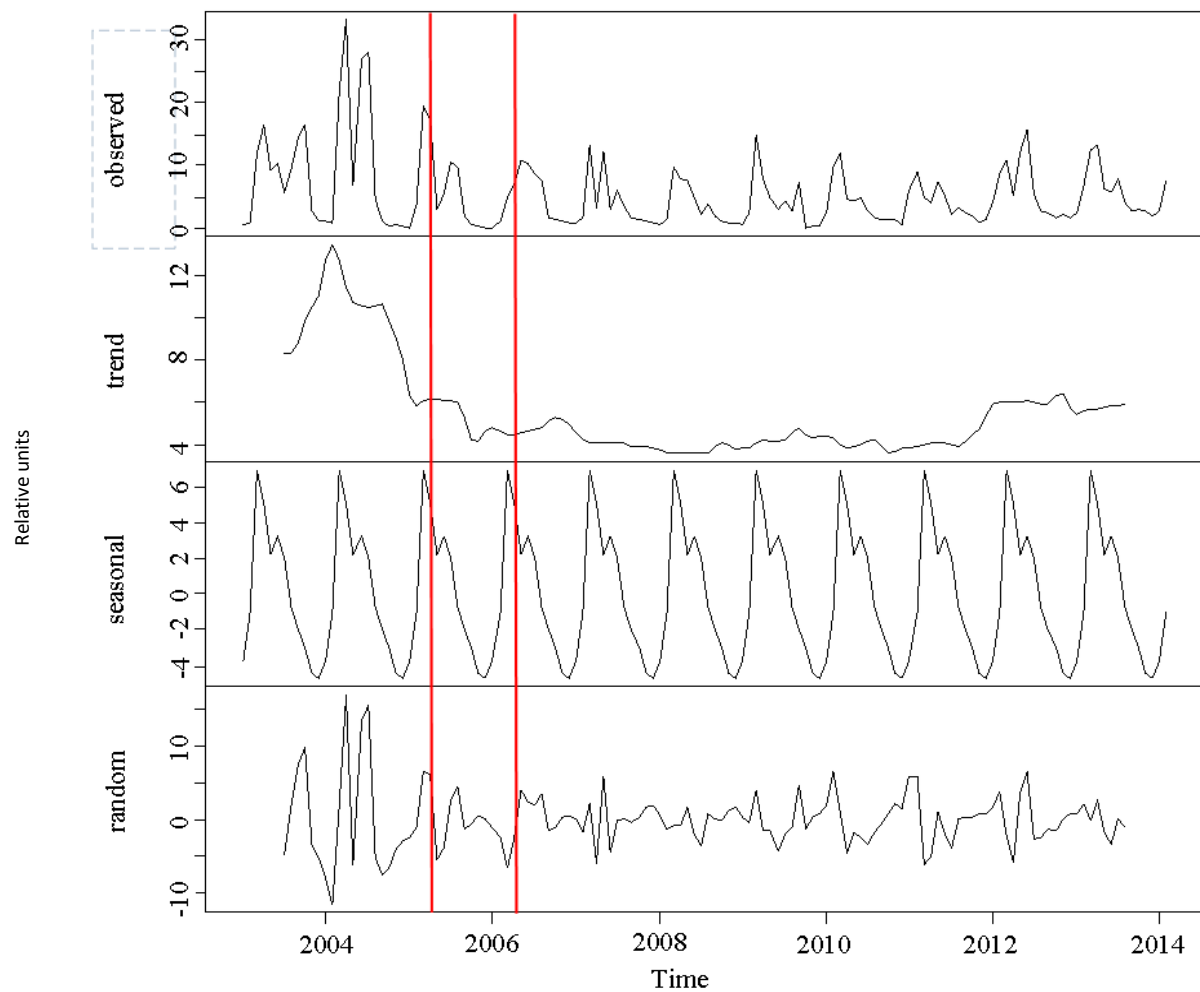


Figure 5.26: Time series analysis of mean monthly total lake chlorophyll-*a* concentrations within Cardiff Bay between January 2003 and April 2014. Red lines indicate the initiation of phosphorus removal from wastewaters.

5.5.12 The Sediment-Water Exchange of Phosphorus in Cardiff Bay

There was no seasonal variation of total phosphorus loading in Cardiff Bay.

The Ca bound (Table 5.1), Fe-Al bound (Table 5.2) and labile bound (Table 5.3) phosphorus per gram⁻¹ dry weight (Pgdw) did not vary statistically between depths in the lake sediment during winter or summer (Appendix Figure 5.4).

The concentrations of phosphorus within the lake sediment varied between winter and summer. Ca bound phosphorus was higher in summer than in winter ($F_{1,142} = 25.16$, $P < 0.01$). Labile Pgdw was higher in winter than in summer ($t = 110.6$, $n = 101$, $P < 0.001$). There was no difference observed between winter and summer concentrations of Fe-Al bound phosphorus.

Ca-bound phosphorus in winter and summer was statistically higher (winter: $r_{160} = 122.5$, $P < 0.01$; summer: $r_{160} = 338.1$, $P < 0.01$) than the concentrations of labile and Fe-Al bound P.

There was no difference in Ca or Fe-Al bound P between sites during summer, but during winter the Ca bound phosphorus at site A was statistically higher ($F_{5,75} = 10.5$, $P < 0.01$) than sites 9, 10 and B. Fe-Al bound P was statistically higher ($r_{28} = 7.83$, $P < 0.001$) at site A than at sites 9, 10, B and C.

During winter, labile phosphorus concentrations were statistically higher ($r_{28} = 6.2$, $P < 0.001$) at site A than at sites 9, 10, B and C. During summer there was also a difference in labile Pgdw between sites. Concentrations at site 10 were statistically higher ($r_{23} = 5.07$, $P < 0.01$) than those at sites C and D.

Table 5.1: Ca bound phosphorus per gram⁻¹ dry weight (Pgdw) in the sediment of Cardiff Bay during summer and winter 2014.

Site	Depth (cm)	Summer (Pgdw)	Winter (Pgdw)
4	0-3	0.0645	
5	0-3	0.0924	
9	0-3	0.1327	0.0610
9	3-6		0.0503
9	6-9		0.0407
9	9-12		0.0599
9	12-15		0.0514
10	0-3	0.1189	0.0221
10	3-6	0.1325	0.0570
10	6-9	0.1246	0.0769
10	9-12	0.1164	0.0790
10	12-15	0.1117	
A	0-3	0.1491	0.0767
A	3-6		0.1239
A	6-9		0.1247
A	9-12		0.1502
A	12-15		0.1165
B	0-3	0.0496	0.0613
B	3-6	0.00645	0.0448
B	6-9	0.1213	0.0693
B	9-12	0.1226	
B	12-15	0.1208	0.0649
C	0-3	0.1013	0.0845
C	3-6	0.1158	0.0941
C	6-9	0.1105	0.1301
C	9-12	0.0917	0.0906
C	12-15	0.1092	0.0801
D	0-3	0.1427	0.0563
D	3-6	0.1487	0.1163
D	6-9	0.1434	0.1167
D	9-12	0.0872	0.1466
D	12-15	0.0964	0.086

Table 5.2: Fe-Al bound phosphorus per gram⁻¹ dry weight (Pgdw) in the sediment of Cardiff Bay during summer and winter 2014.

Site	Depth (cm)	Summer (Pgdw)	Winter (Pgdw)
4	0-3	0.0228	
5	0-3	0.0245	
9	0-3	0.03387	0.0293
9	3-6		0.0115
9	6-9		0.0043
9	9-12		0.0052
9	12-15		0.0065
10	0-3	0.0297	0.0140
10	3-6	0.0318	0.0272
10	6-9	0.0399	0.0362
10	9-12	0.0288	0.0235
10	12-15	0.0371	
A	0-3	0.0396	0.0259
A	3-6		0.0439
A	6-9		0.0411
A	9-12		0.0607
A	12-15		0.0485
B	0-3	0.0196	0.0236
B	3-6	0.0192	0.0103
B	6-9	0.0377	0.0263
B	9-12	0.0296	
B	12-15	0.0236	0.0146
C	0-3	0.0339	0.0218
C	3-6	0.0269	0.0286
C	6-9	0.0206	0.0248
C	9-12	0.0194	0.0285
C	12-15	0.0161	0.0169
D	0-3	0.0370	0.0276
D	3-6	0.0359	0.0454
D	6-9	0.0246	0.0240
D	9-12	0.0200	0.0276
D	12-15	0.0106	0.0165

Table 5.3: Labile bound phosphorus per gram⁻¹ dry weight (Pgdw) in the sediment of Cardiff Bay during summer and winter 2014.

Site	Depth (cm)	Summer (Pgdw)	Winter (Pgdw)
4	0-3	0.0063	
5	0-3	0.0089	
9	0-3	0.0068	0.0150
9	3-6		0.0132
9	6-9		0.0101
9	9-12		0.0133
9	12-15		0.0132
10	0-3	0.0132	0.0214
10	3-6	0.0084	0.0227
10	6-9	0.0088	0.0225
10	9-12	0.0075	0.0133
10	12-15	0.0074	
A	0-3	0.0069	0.0273
A	3-6		0.0276
A	6-9		0.0294
A	9-12		0.0259
A	12-15		0.0250
B	0-3	0.0047	0.0153
B	3-6	0.0048	0.0128
B	6-9	0.0065	0.0131
B	9-12	0.0058	
B	12-15	0.0063	0.0113
C	0-3	0.0042	0.0187
C	3-6	0.0042	0.0171
C	6-9	0.0042	0.0184
C	9-12	0.0047	0.0152
C	12-15	0.0029	0.0105
D	0-3	0.0059	0.0211
D	3-6	0.0058	0.0217
D	6-9	0.0048	0.0206
D	9-12	0.0061	0.0164
D	12-15	0.0042	0.0192

5.6 Discussion

There was no evidence that the phytoplankton population of Cardiff Bay were nutrient limited. Silica was limiting relative to phosphorus and nitrogen but did not limit phytoplankton growth. There was little evidence for phosphorus release from the lake sediments, and the phosphorus reduction from river inputs in 2005 and 2006 did not impact the phytoplankton biomass of the lake. There was no spatial or vertical variation of nutrients within the majority of the lake. With the exception of site D, situated close to the wetlands. Nitrogen concentrations were lower here, possibly as a result of nitrification by macrophytes removing nitrogen from the surrounding water column. The River Ely experienced higher nitrogen concentrations than any other sites studied; the river acts an external source of nitrogen and its slow-moving nature and the relatively deep water-column could contribute to nitrogen build-up. Lake flushing rate directly and indirectly, through precipitation and river discharge, influenced nutrient concentrations within the lake. Fast flushing rates caused nutrient concentrations to be diluted or washed out faster than they could build-up.

5.6.1 The Phytoplankton & Environmental Variables of Cardiff Bay

Cardiff Bay has been found to be a polymictic lake which is subject to natural seasonal variation. Despite the variation in temperature, precipitation, river discharge and lake flushing rate, the phytoplankton biomass concentrations were so low that no strong relationship were discovered between phytoplankton biomass and the environmental variables of the lake. Understanding if nutrient concentrations within Cardiff Bay vary seasonally will enable the variations in the phytoplankton population to be further understood.

The flushing rate of Cardiff Bay is quick enough that the lake acts more like a river system, therefore, despite a healthy productive phytoplankton community being present, biomass is flushed out faster than it can increase, resulting in low phytoplankton biomass throughout the year. It is essential to acknowledge if nutrient concentrations are influenced in the same way by flushing rate to determine how nutrient concentrations are controlled in the system.

The low phytoplankton biomass was dominated by diatoms, as they can withstand and flourish within a highly mixed water column such as Cardiff Bay. Chlorophytes were present in higher concentrations than cryptomonads. Cyanobacteria, arguably the most crucial phytoplankton group for lake management were present in insignificant concentrations throughout the study. Both diatoms and chlorophytes displayed seasonal variation in biomass, causing total phytoplankton biomass to peak during spring, and decline through summer and autumn until it reached its lowest concentrations in winter. Determining if the phytoplankton community of Cardiff Bay is limited by nutrient concentrations is essential to understand if nutrient inputs will be required to be reduced in the future to ensure that lake water quality is maintained.

It has been predicted that Cardiff Bay will be subject to changes in the future as a result of an increase in temperature and intense precipitation events. These changes in climate could have the ability to cause the lake to stratify, resulting in an increase in phytoplankton biomass and a shift in species composition. Studying the relationship of nutrients with the environmental variables within Cardiff Bay is essential to understand and predict if changes in climate will change the nutrient dynamics within the lake, and to understand the consequences this could have on lake water quality and management.

5.6.2 Spatial & Vertical Distribution of Nutrients within Cardiff Bay

Site 17, within the River Ely, displayed higher nutrient concentrations than any other site studied between 2003 and 2014. Site A, at the mouth of the River Ely also experienced higher surface nitrate and TDN concentrations than the other sites studied between April 2012 and April 2014. Nitrate and TDN concentrations were also higher between April 2012 and April 2014 in the bottom waters at site A. Nitrogen is very water soluble and can be transported into a lake system through river input (Kumar & Sekaran, 2014). The River Ely is a slower moving river compared to the Taff, therefore it not only transports nitrogen to the lake, but because it is slower moving, it has the ability for nitrogen concentrations to build up. At c8m, site A is also the deepest site studied and it has been documented that N removal can decrease with depth (Reisinger et al., 2016). In comparison to the River Ely, the conditions within the main body of the lake are so well mixed that nutrient concentrations are diluted.

Site D, adjacent to the wetlands, experienced lower nitrate and TDN concentrations at the bottom of the water column between April 2012 and April 2014. The higher density of macrophytes present within the wetland area has the potential to aid the removal of nitrogen from the surrounding water column faster than at other areas within the lake (Weisner et al., 1994, Saunders & Kalff, 2001). This removal generally results from nitrification in the root zone (Weisner et al., 1994), explaining why nitrate and TDN concentrations were found to be lower near the lake bed in this area.

The concentrations of phosphorus and silica did not vary spatially between the lake or river sites, and aside from site D, the nitrogen concentrations did not vary spatially within the main body of the lake. This is likely to be a result of the polymictic nature of the lake, constantly mixing the water column. The absence of vertical variation in nutrient concentrations within the water column of the lake between April 2012 and April 2014 is also likely to be a result of the polymictic nature, and therefore the absence of stratification.

5.6.3 The Temporal Variation of Lake Nutrients

The surface nutrient concentration of Cardiff Bay has altered over time. There was a reduction in mean annual SRP and TP between 2003 and 2013, which became apparent annually from 2006 onwards; coinciding with the reduction in phosphorus input into the lake. However, the reduction in phosphorus input was not clear when analysing TP river concentrations, and the phytoplankton biomass of the lake did not decrease as intended. This indicates that the phytoplankton population of the lake are not nutrient limited, and this management method, under current lake conditions, is not effective for phytoplankton control.

During winter in Cardiff Bay, TP and SRP concentrations were at their lowest. In a typical lake system, you would expect TP concentrations to increase during winter when precipitation rates increase (Chen et al, 2010). High levels of precipitation experienced during winter diluted TP and SRP concentrations within the lake of Cardiff Bay and combined with the impact of a high lake flushing rate, TP and SRP were washed out of the lake faster during winter, preventing a build-up of nutrients in the lake from occurring (Jagtman et al., 1992).

Silica showed a significant seasonal variation. Concentrations peaked in winter and dropped in spring until reaching their lowest concentrations in summer. This is a result of utilisation by the phytoplankton population. The silica concentrations were at their lowest in May, which corresponded to the diatom population growth (Atkins, 2009). Silica concentrations were also influenced by lake flushing rate during summer,

5.6.4 Environmental Controls on Nutrients in Cardiff Bay

Annually, an increase in river discharge resulted in a reduction of TP, and during summer an increase in combined river discharge caused silica concentrations to decline. Both of these relationships are likely to be as a result of dilution and increasing lake flushing rates, stemming from the influence of river discharge on these processes. Increasing lake flushing rates directly caused nitrite concentrations to decrease as a result of increased outflow from the lake.

Weak relationships were found between the environmental variables and nutrients in Cardiff Bay, indicating that, with the exception of the direct and indirect influence of lake flushing rate, other factors are more significant in controlling the nutrient concentrations.

The increase of dissolved oxygen causing an increase in nitrogen concentrations within the lake is unusual, as when nitrogen concentrations increase DO is often utilised by organisms, therefore decreasing the DO concentration. This indicates that the relationship between DO and nitrogen was not a direct one.

5.6.5 Nutrient Limitations within Cardiff Bay

Phytoplankton growth within Cardiff Bay was not limited by nutrient concentrations. Although silica concentrations could explain the change in diatom biomass during spring and cryptomonad biomass during summer, the nutrient addition mesocosm bags did not demonstrate any changes in phytoplankton biomass with nutrient addition. This is supported by the lack of phytoplankton biomass response the reduction in phosphorus in river inputs to the lake as the reduced TP concentrations did not lower phytoplankton biomass.

Silica appeared to be limiting relative to phosphorus, and at times, TDN. However, there was variation in this relative limitation, particularly when comparing the ratios for the timescale between 2012 and 2014; the results did not always correspond. Between 2003 and 2014 TDN:Si only went above 16:1 once, suggesting that TDN is limiting in the lake relative to silica. However the fortnightly data taken between 2012 and 2014 indicates that the TDN:Si was above 16:1 the whole time. Nutrient limitation often fluctuates in lake system, particularly on a seasonal basis as lake characteristics alter and phytoplankton biomass changes as a result. It has also been stated that some nutrient limitation can only be observed short-term (Kolzau et al., 2014), therefore the variation in relative nutrient limitation experienced in Cardiff Bay is not uncommon. Most importantly however, nutrient concentrations did not show any limitation of the phytoplankton biomass.

5.6.6 Lake Sediment-Water Phosphorus Exchange

There was no significant difference in Fe/Al bound-P between summer and winter for any site indicating that there was no evidence for redox-linked internal loading to occur. Therefore, conversion of ferric to ferrous Fe due to lower sediment redox potential, with concomitant loss of bound phosphate into the labile P pool was unlikely (Boström et al., 1988, Perkins & Underwood, 2001b). This is supported by a lower labile-P pool in summer suggesting a lack of release of bound P fractions. This lower labile-P in sediments in summer may indicate a loss to the overlying water column and/or biological utilisation. Warmer summer temperatures may have increased biological utilisation of phosphate or alternatively a greater diffusional release to the water column may have occurred (Perkins & Underwood, 2001b, Meis et al., 2012). Internal release of phosphate would only be of importance regarding Bay management if there was evidence of P-limitation, which has been thoroughly refuted from the water chemistry data evidenced earlier in this chapter. Calcium bound P (Ca-P) was significantly higher than other fractions of sediment phosphorus, however this pool is largely refractory in the absence of significant change in pH (Boström et al., 1988).

Although Fe/Al-P appeared to have lower sediment concentrations in deeper depth sections, this was not significant (probably due to low replication, $n=3$ in summer and winter for each site). A lower Fe/Al-P concentration in deeper sediments would indicate redox-mediated loss

of Fe-P through reduction of ferric to ferrous Fe (Boström et al., 1988) and hence should be matched by concomitant increase in labile P, unless the latter is transferred through diffusion and mixing into the upper sediment layers and lost to the water column. Data from this study are therefore inconclusive but may suggest a low level of internal loading from the Fe-P pool through these mechanisms.

The measurement of significant pools of Fe/Al-P and labile P indicate that there is potential for internal sediment loading in the Bay and this should be considered as a potential risk in the future should reduction in external loading occur (see Chapter 7). Furthermore, there was little spatial heterogeneity in sediment P fractions, with Fe/Al-P and labile-P being higher at Site A in winter and labile-P was higher at site 10 in summer, but otherwise there was no other significant variation between sites.

5.6.5 The Potential for Top-Down Control of Phytoplankton in Cardiff Bay

Lake phytoplankton biomass is often determined by nutrient control (bottom-up) or grazing (top-down). The former of which is often most influential (Moss et al., 1994). However, the absence of nutrient limitation on the phytoplankton population within Cardiff Bay suggests that top-down control through zooplankton grazing, or filter feeding by zebra mussels may be contributing to the limitation of phytoplankton biomass in the Bay. The significance of top-down control in the Bay has been explored in Chapter 6.

5.7 Conclusions

The bottom-up controls on the phytoplankton community within Cardiff Bay were investigated between January 2003 and April 2014. Phosphorus, nitrogen and silica concentrations were determined, alongside the potential for phosphorus release from the lake sediment. Nutrient limitation of the phytoplankton population was investigated using nutrient addition mesocosms, pre and post phosphorus removal data, and the analysis of nutrient ratios within the lake.

There was no evidence that the phytoplankton community of Cardiff Bay was nutrient limited, flushing rates appear to have a stronger influence on phytoplankton biomass than the availability of nutrients. There was no increase in phytoplankton biomass as a result of nutrient addition, and the biomass did not decrease after phosphorus removal from the river inputs. The TDN:TP:Si ratios varied throughout the study, and in some cases, silica was relatively limiting in comparison to nitrogen and phosphorus. However, this was not significant enough to influence the phytoplankton biomass of the lake. The absence in vertical distribution of phosphorus in the sediment of the lake suggests that there is little upward migration of dissolved phosphorus from deeper sediment layers. There was also limited internal loading taking place through phosphorus release from the sediment, indicating that the lake sediments are not the primary source of phosphorus into the lake system; external inputs such as river discharge have more of influence on nutrient availability in the lake.

The concentration of nutrients within the lake were related to the phytoplankton biomass. The dominance of diatoms required higher levels of silica than any other nutrient. Therefore, despite phosphorus being readily available within the lake, cyanobacteria biomass did not increase, indicating that cyanobacteria growth is being controlled by other factors within the lake.

The concentration of nitrogen, phosphorus and silica were uniform within Cardiff Bay, both spatially and with depth as a result of high mixing levels. However, there was evidence of nitrogen uptake by macrophytes within the wetland area of the lake as the surrounding waters experienced lower nitrogen concentrations. This highlights the positive impact that macrophytes can have on reducing nitrogen availability within the water column. In addition, as found previously in other Chapters (3 and 4), conditions in the River Ely are different to those experienced in the main body of the lake, therefore nitrogen concentrations were higher at this site than the others surveyed.

The combination of increased precipitation, river discharge and lake flushing rates, caused the dilution of nutrients in the lake, and increased outflow rates. This indicates that if the frequency of intense precipitation events increases in the future as predicted, the nutrient concentration in the lake is unlikely to increase.

It is uncommon for the phytoplankton population of a lake not to experience nutrient limitation during any time of year. Top-down and bottom-up regulation are both prevalent in freshwater

food-webs and the two processes do not act independently (Taylor et al., 2015). As previously reported in Chapters 3 and 4, Cardiff Bay has an unusually high flushing rate, which has a significant impact on the variables that control the phytoplankton population. However, it is possible for top-down process in freshwater ecosystems to influence phytoplankton biomass through trophic cascade and grazing, therefore it is important to determine if zooplankton grazing or filter-feeding by the zebra mussel population in Cardiff Bay has an impact on the productive, but low, phytoplankton biomass in the lake. Understanding these interactions is essential for water quality management as the interactions that take place between top-down and bottom-up processes can help conservationists predict the consequences of changing species composition on ecosystem function (Taylor et al., 2015).

THE TOP-DOWN CONTROLS ON PHYTOPLANKTON IN CARDIFF BAY

6.1 Summary

Findings presented in previous chapters demonstrate a low biomass of phytoplankton in Cardiff Bay, although productivity measured using fluorescence indicated a healthy productive biomass. There was no evidence of nutrient limitation, and hence bottom-up control appears unlikely. This chapter investigated potential top-down control through zooplankton grazing and examines other potential biological control by zebra mussel filtration. The zooplankton community of Cardiff Bay displayed seasonal variation, with the highest abundance found in summer. Although the zooplankton biomass in Cardiff Bay appeared to be significant enough to control the phytoplankton community, grazing pressure was only likely to be small due to the absence of effective zooplankton grazer species. More significantly, the presence of a growing population of zebra mussels within the lake has the potential to effect phytoplankton and zooplankton populations. Although the impact of zebra mussel grazing was not studied here, they are an important factor to consider for the future management of the lake.

6.2 Introduction

As discussed earlier in this thesis, phytoplankton biomass and productivity, along with community structure, can be controlled by nutrient limitation, bottom-up resource dependent control, and/or top down control in the form of predation and grazing. In most natural and semi-natural lakes, a balance of these two processes interact, also mediated by environmental and hydrological controls (Stephen & Phillips, 1998). Previous sections of this thesis have investigated bottom-up control and environmental and hydrological factors. In this chapter the remaining area focussed upon is biological control.

6.2.1 Biological Control of Lakes

Phytoplankton growth in a lake can be mitigated by mechanisms such as grazing, a form of biological control (Carpenter et al., 1995). It is possible for grazing to effectively control total phytoplankton biomass over a wide range of phosphorus input rates, however it may not be able to suppress all harmful algal blooms (HABs) (Carpenter et al., 1995). When phytoplankton abundance is controlled through grazing, a variation in productivity can be caused which is often unexplained by nutrient loading or phytoplankton standing stock, and can be driven by zooplankton grazing pressure (Elser & Goldman, 1991).

6.2.2 The Presence of Zooplankton in Lakes

Zooplankton biomass often peaks between mid-spring and mid-summer and is positively related to phytoplankton biomass, e.g. zooplankton biomass peaks slightly lag phytoplankton peaks, with the latter often declining due to the grazing enforced control of the former (Mccauley & Kalff, 1981, Zervoudaki et al., 2009). Zooplankton grazing can influence the phytoplankton community structure, therefore grazing rates are most critical during spring when phytoplankton growth often peaks to control phytoplankton blooms (Mckinstry & Campbell, 2018).

Phytoplankton community structure is also controlled by zooplankton grazing. For example cyanobacteria-dominated phytoplankton populations can be resistant to zooplankton grazing (Elser & Goldman, 1991). This can often be the most important impact for lake managers as it can contribute to HABs.

Zooplankton are also important for organisms higher up the trophic level of a lake as they comprise the prey for predators such as fish, and can support ecologically and economically important species such as salmon (Mckinstry & Campbell, 2018).

6.2.3 The Impact of Zebra Mussels on a Lake System

A further form of top-down biological control in Cardiff Bay is filtration feeding by zebra mussels (*Dreissena polymorpha*). The presence of zebra mussels within Cardiff Bay can be considered a cause for concern as their filtration rate, between 1 and 2 litres of water per day, per mussel, has the potential to remove essential phytoplankton biomass (Maguire & Sykes, 2004), hence reducing productivity to support the food web and biodiversity of the Bay. Conversely, from the perspective of maintain a clear water system and reducing risk of HABs, this filtration can be regarded as a biological management tool. Zebra mussel reproduction rate is rapid and increases in warmer waters making them a threat to lake ecosystems in the future should water temperatures rise due to climate change (Adrian et al., 2009a). Climate change associated temperature rise therefore has the potential to enhance zebra mussel filtration but may also reduce zooplankton grazing by disruption of zooplankton community structure.

6.3 Hypotheses

To determine how the phytoplankton community within Cardiff Bay may be controlled by top down control, the following thesis objectives will be addressed in this chapter:

1. The abundance and diversity of zooplankton species in the Bay will be determined, alongside their relationship with the phytoplankton community.
2. The potential for zebra mussel filtration control of phytoplankton biomass will be assessed.

Using data collected between April 2012 and April 2014 in Cardiff Bay, the following hypotheses have been tested:

The zooplankton of Cardiff Bay varied seasonally but did not alter spatially due to the lake being well-mixed.

Zooplankton control the phytoplankton biomass within the lake through grazing.

The zebra mussel population of Cardiff Bay has the potential to contribute to top down control through phytoplankton filtration.

6.4 Materials and Methods

The 200 ha artificial freshwater lake of Cardiff Bay was the first site in Wales to record the presence of the non-native zebra mussel (*Dreissena polymorpha*) (Gb Non-Native Species Secretariat, 2016). The description of the study site and the location of the zebra mussel survey can be found in Chapter 2 of this thesis.

6.4.1 Sampling Methods

Zooplankton data were collected once a fortnight between April 2012 and April 2014 at sites 4, 5, 9, 10, A, B, C, and D. Vertical samples were taken using a weighted fine mesh conical zooplankton net from the bottom to the surface of the water column (full sampling methods and sample site locations can be found in Chapter 2. Site specific and seasonal mean zooplankton counts were calculated using all data collected at each of the 8 sites.

Zebra mussel counts were taken from grab samples at 20 sites within the lake in July 2014 to replicate an earlier study carried out by Alix (2010) in 2007. The age of the mussels were determined by size (the values of which can be found in Chapter 2.2.4) and raw data used for analysis. Twenty-five transects within the lake were taken using side scan sonar to map the bed of the Bay and the potential presence of zebra mussels (full methods can be found in Chapter 2).

6.4.2 Statistical Analysis

Pearson's correlation was undertaken on seasonal data sets using 'R' to determine the relationship that phytoplankton has with zooplankton. The relationships that zooplankton has with the physical and chemical characteristics of the lake were also assessed utilising data presented in Chapters 3, 4 and 5.

6.5 Results

6.5.1 The Zooplankton Community of Cardiff Bay

Mean monthly whole-Bay zooplankton counts in Cardiff Bay taken between April 2012 and April 2014 varied throughout the year, with the highest mean monthly counts were found during June, July, August and September (Figure 6.1), with mean monthly whole-Bay counts higher than 23,000 ind. L⁻¹. During the rest of the year (excluding December as counts were not taken during this month), mean monthly whole-Bay zooplankton counts did not reach above $1,809 \pm 1401$ ind. L⁻¹ (May) and dropped to 168 ± 251 ind. L⁻¹ (January). There was a statistically significant difference in zooplankton counts between months ($F_{10,98} = 28.35$, $P < 0.01$). Where June, July, August and September experienced significantly higher zooplankton counts than the rest of the months in the year. The month of June experienced the highest mean monthly whole-Bay zooplankton count ($48,267 \pm 24,698$ ind. L⁻¹), a count significantly higher than that experienced in August ($23,486 \pm 10,628$ ind. L⁻¹).

The seasonal zooplankton counts are significantly different between seasons ($F_{3,98} = 42.94$, $P < 0.01$), (Figure 6.2). Summer zooplankton counts ($35,557 \pm 19,062$ ind. L⁻¹) were significantly higher than those in all other seasons.

The main genus of zooplankton (nomenclature correct at time of identification) found within the lake were; Alonella, Alona, Bosmina, Calanoida, Chaoborus, Copepod nauplii, Cyclopidae, Cyclopoid Nauplius, Daphina, Daphniidae, Diaphanosoma, Latona, Leptodora, Plomida and Sida (Figure 6.3). The zooplankton with the highest mean whole-Bay zooplankton count in the lake were Allonella (112 ± 447 ind. L⁻¹), Calanoida (149 ± 1337 ind. L⁻¹), Cyclopidae (204 ± 819 ind. L⁻¹), and zebra mussel larvae (189 ± 736 ind. L⁻¹), (Figure 6.3, over 100 ind.L⁻¹).

There were no significant differences in zooplankton biomass between sample sites using the total number of individuals counted (all species pooled).

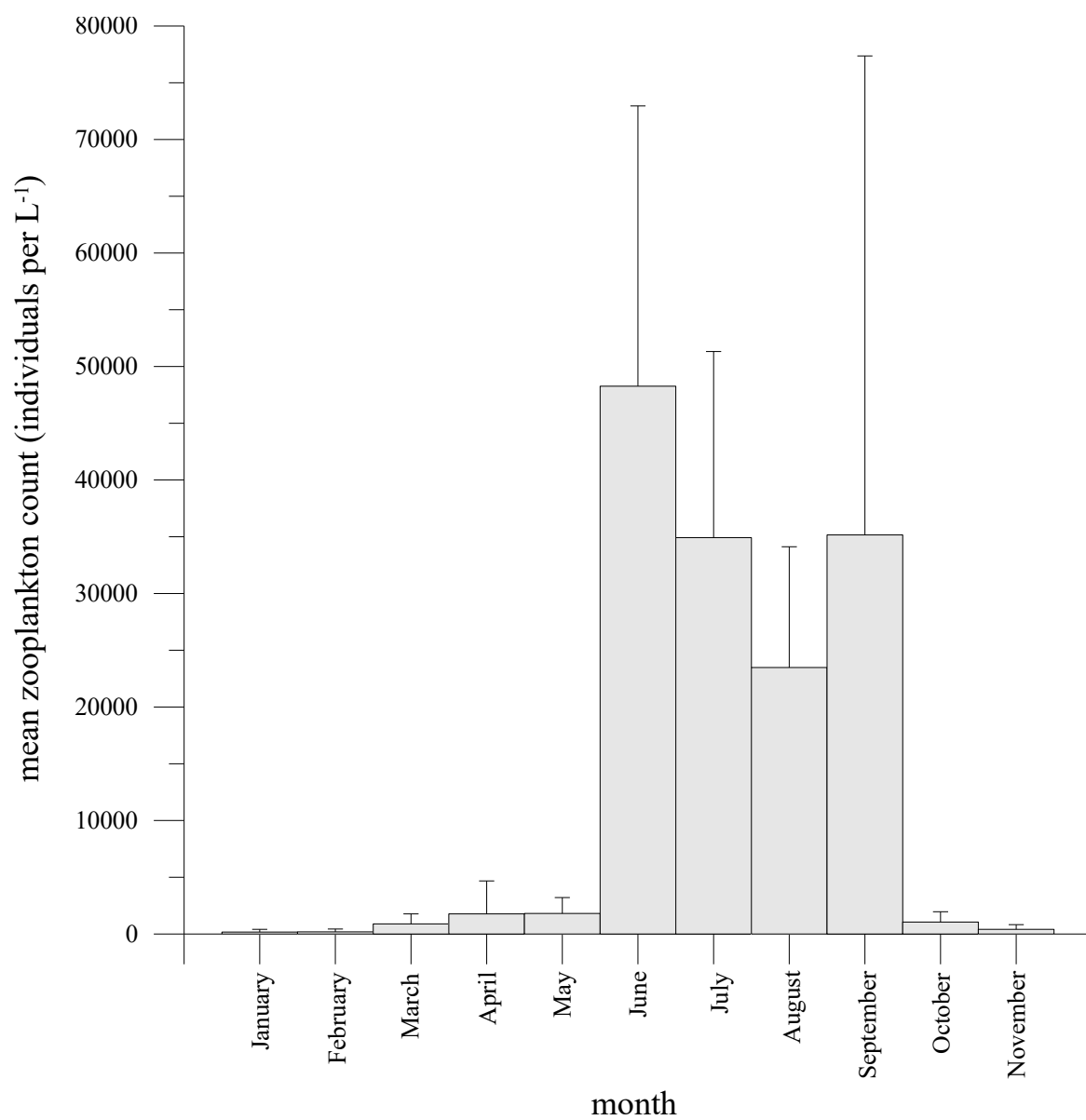


Figure 6.1: Mean monthly whole-Bay zooplankton counts from Cardiff Bay, taken between April 2012 and April 2014 (standard error bars).

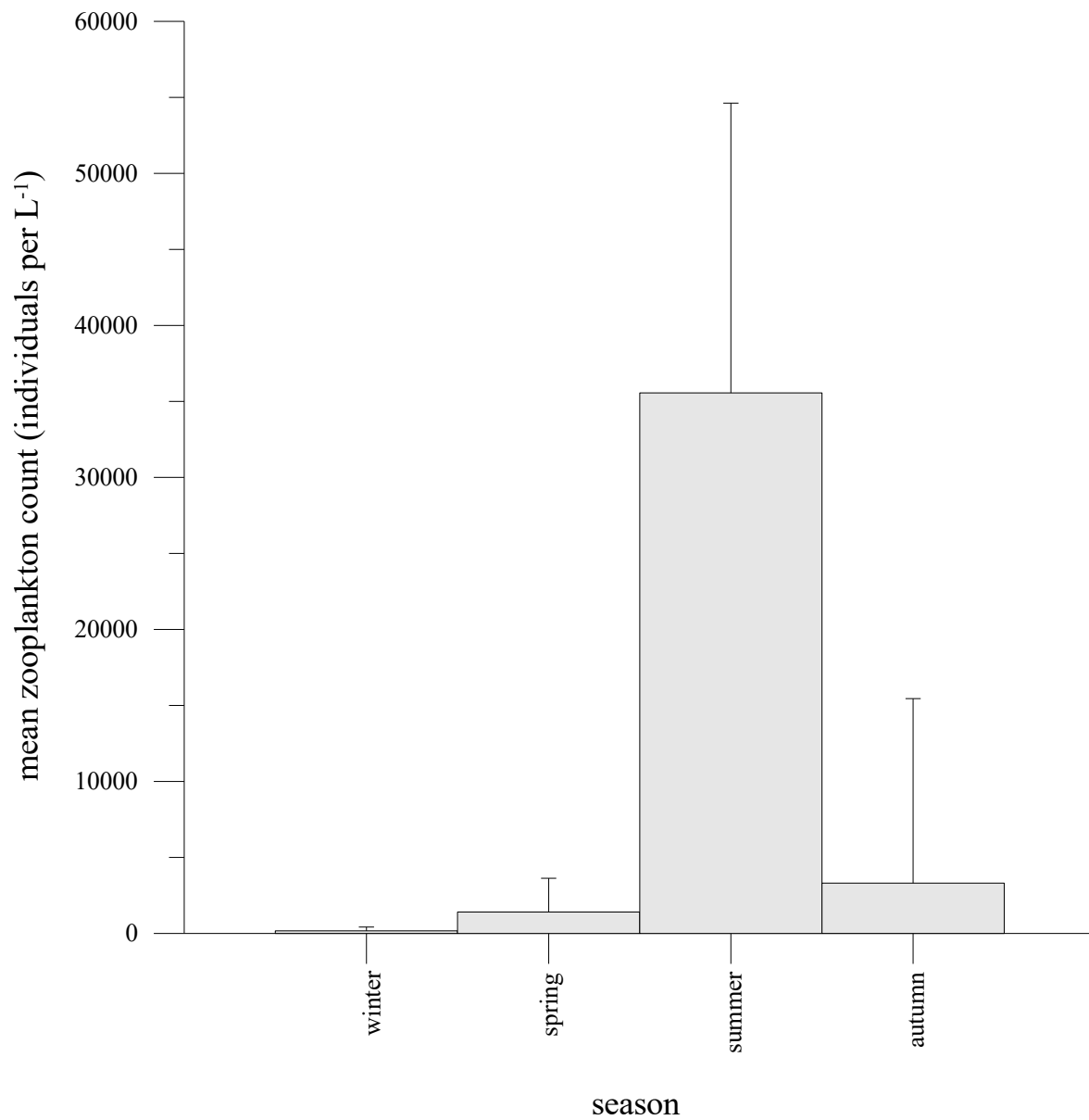


Figure 6.2: Mean seasonal whole-Bay zooplankton counts from Cardiff Bay, taken between April 2012 and April 2014 (standard error bars).

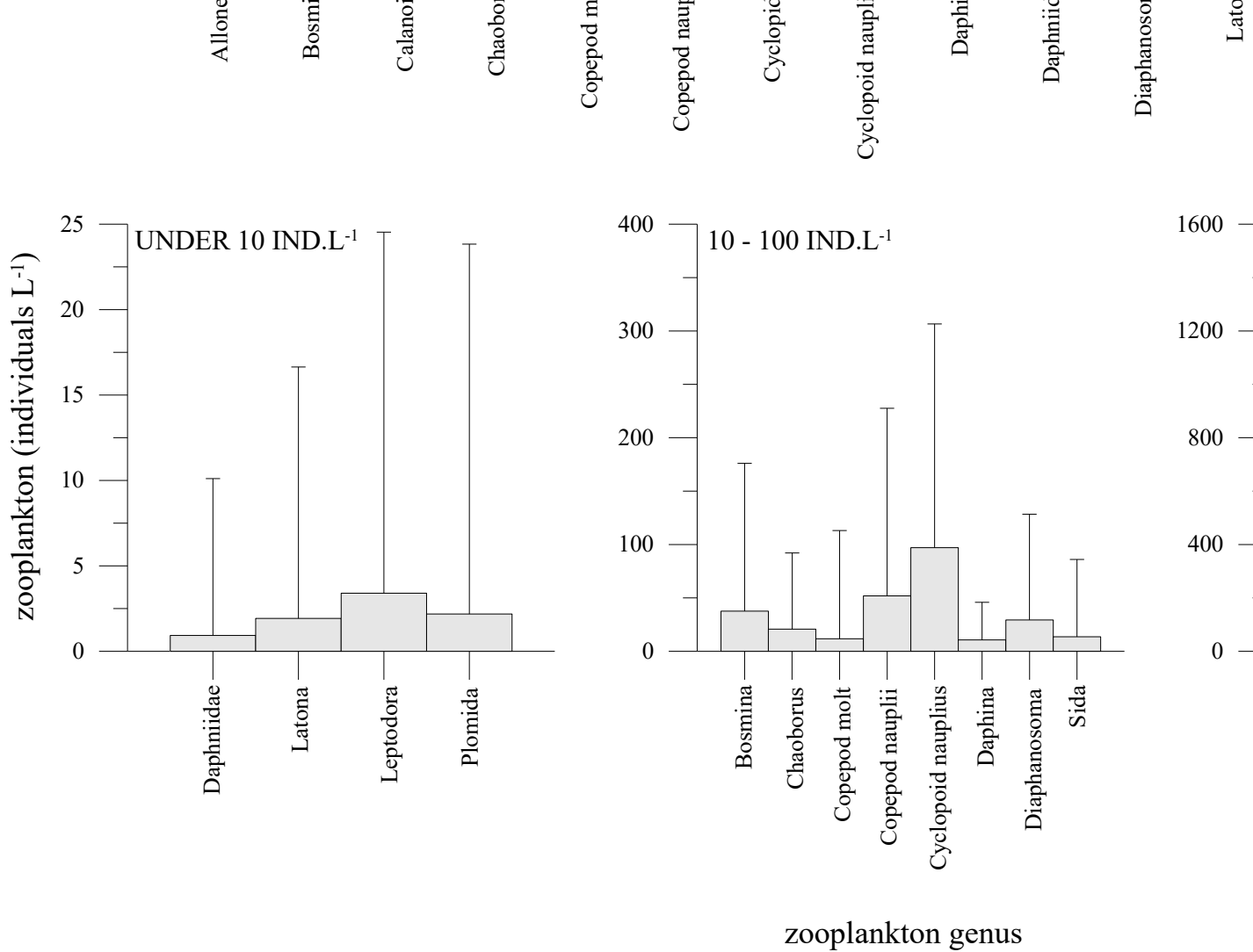


Figure 6.3: Mean whole-Bay zooplankton species, figures of all individuals, under 10 individuals L⁻¹, between April 2012 and April 2014 in Cardiff Bay, and over 100 individuals L⁻¹. Taken between April 2012 and April 2014 in Cardiff Bay.

Mean monthly zooplankton biomass significantly correlated with mean monthly air temperature, mean monthly water temperature, mean monthly lake flushing rate, mean monthly DO, mean monthly nitrite, and mean monthly TDN (Figure 6.4).

Multiple regression was carried out to determine if monthly zooplankton biomass was controlled by air temperature, water temperature or lake flushing rate (Equation A). Mean monthly air temperature influenced 62% of the variation in monthly zooplankton biomass. Water temperature and lake flushing rate were insignificant.

Equation A: $(zooplankton) = 2.4 + 3.8 (air\ temperature); F_{1,9} = 14.38, P < 0.01, r^2 = 0.62$

Regression was carried out to determine if DO was controlled by zooplankton (Equation B).

Equation B: $(dissolved\ oxygen) = 37.4 + 3.7 (zooplankton); F_{1,9} = 14.04, P < 0.01, r^2 = 0.61$

Multiple regression was carried out to determine if the relationship between mean monthly zooplankton and nitrite or TDN was more significant (Equation C). Mean monthly zooplankton explained 68% of the variation in nitrite. TDN was insignificant.

Equation C: $(nitrite) = 5.1 + 4.4 (zooplankton); F_{1,9} = 19.06, P < 0.01, r^2 = 0.68.$

Mean monthly zooplankton biomass weakly correlated with mean monthly River Taff discharge ($r = 0.63, n = 9, P < 0.05$), mean monthly River Ely Discharge ($r = 0.7, n = 9, P < 0.05$), mean monthly combined river discharge ($r = 0.65, n = 9, P < 0.05$), mean monthly ammonium ($r = 0.7, n = 9, P < 0.05$), and mean monthly silica ($r = 0.71, n = 9, P < 0.05$).

There were no significant seasonal correlations between mean whole-Bay zooplankton and environmental variables in Cardiff Bay.

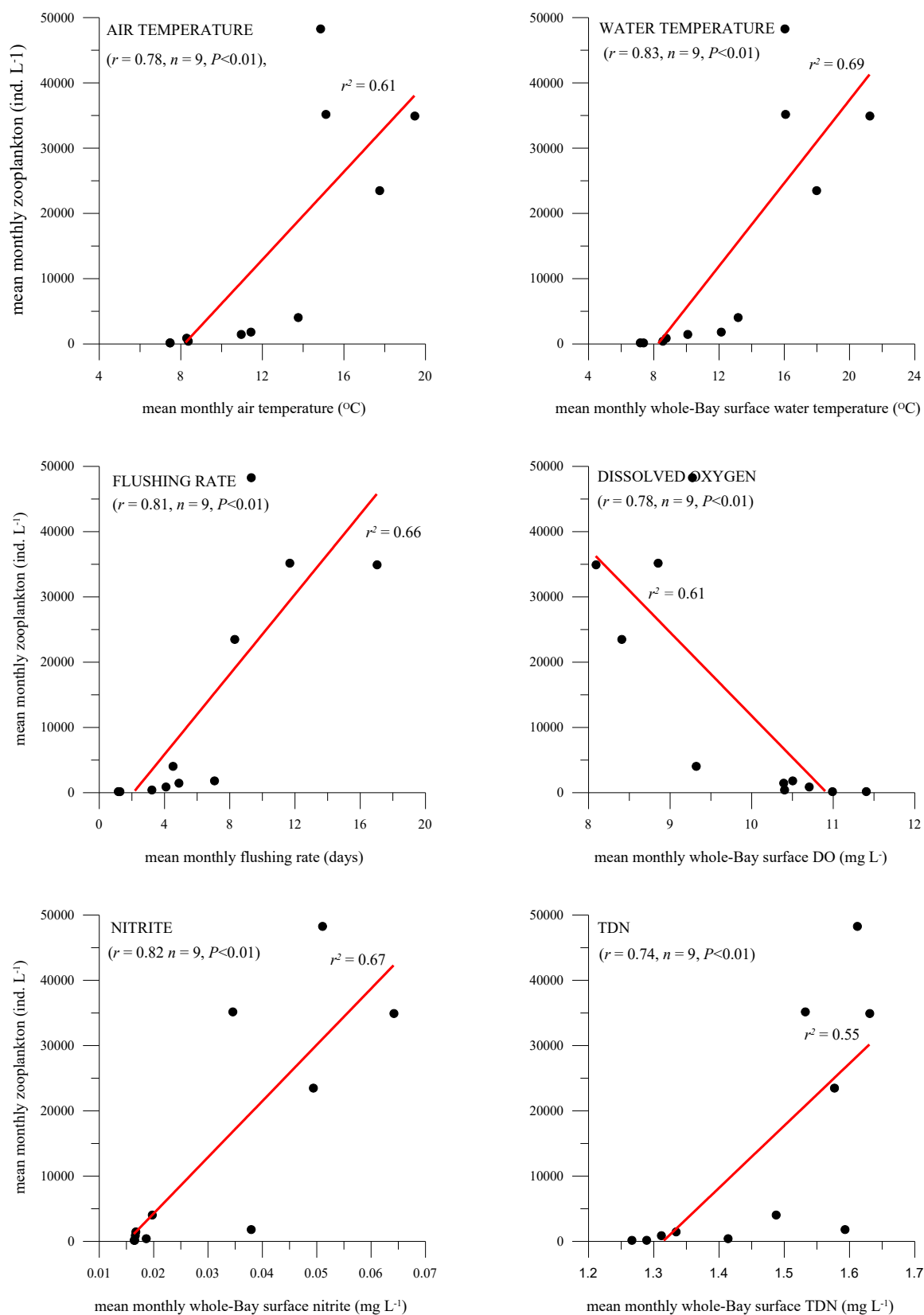


Figure 6.4: Mean monthly individual zooplankton counts and mean monthly air temperature, whole-Bay surface water temperature, lake flushing rate, dissolved oxygen, nitrite and TDN in Cardiff Bay. Data from April 2012 until April 2014.

6.5.2 The Relationship between Zooplankton & Phytoplankton

There was no significant relationship between mean monthly whole-Bay total phytoplankton biomass and mean monthly whole-Bay total zooplankton biomass. Mean monthly zooplankton biomass correlated weakly with mean monthly chlorophytes ($r = 0.67$, $n = 9$, $P < 0.05$). No other phytoplankton groups correlated with zooplankton on a monthly basis.

There was no significant relationship between whole-Bay zooplankton biomass and whole-Bay phytoplankton biomass during any season between April 2012 and April 2014.

6.5.3 The Presence of Zebra Mussels in Cardiff Bay

Between 2008 and 2009, Alix (2010a) found live zebra mussels at 3 sites (sites 8, 10 and 14) out of 20 locations within Cardiff Bay. All 20 of these sites were resurveyed in July 2014 and live zebra mussels were found in 11 out of the 20 sites (Figure 6.5). The sites with the most zebra mussels were site 18 (South of the lake), with 925 mussels counted, site 13 (North-East), with 829 mussels counted, site 15 (East of the lake), with 600 mussels counted, and site 12 (North-East), with 545 mussels counted (Table 6.1, Figure 6.6).

The most abundant age range of zebra mussel found in Cardiff Bay in July 2014 was 1-2 years (measuring 9-16 mm) with 2,192 individuals (62% of those surveyed), (Figure 6.7). The least abundant age of zebra mussels was those over 3 years of age (measuring $25 < \text{mm}$), with (1.7% of those surveyed), (Figure 6.7). In 8 out of the 11 sites where zebra mussels were found, the most abundant size range of zebra mussels was 9 – 16 mm (1 – 2 years). At site 11 (in the marina) and 19 (at the mouth of the River Ely), the most abundant size range was 5 – 8 mm (6 – 12 months in age), and site 15 (East of the lake) had an abundant size range of 17-24 mm (2-3 years in age) (Figure 6.8).

At each site, 81 cm² of substrate was sampled (total of three grabs). The total substrate sampled over the 20 grab sites was 162 m². A total of 3,535 live zebra mussels were found in this area. Cardiff Bay has an area of 2,000,000 m². Therefore, it can be predicted that there are c43,639,575 mussels on the bed of the lake (this excludes all the hard substrate on the walls of

the barrage and structures within the lake). Zebra mussels filter between 1-2 L⁻¹ day⁻¹. It can be predicted that between 43,639,575 and 87,279,150 L⁻¹ of lake water is filtered daily by zebra mussels. The volume of Cardiff Bay is 6,734,000,000 L⁻¹, therefore the volume of Cardiff Bay is filtered every 77 - 154 days by zebra mussels.

The side scan sonar transects within the lake were not able to be used to determine the presence of zebra mussels on the lake substrate. The determination between zebra mussels and rocky substrate on the lake bed was not clear, and the presence of numerous dead zebra mussel shells collected in the grab survey made any determination between live and dead individuals using the sonar results alone impossible.

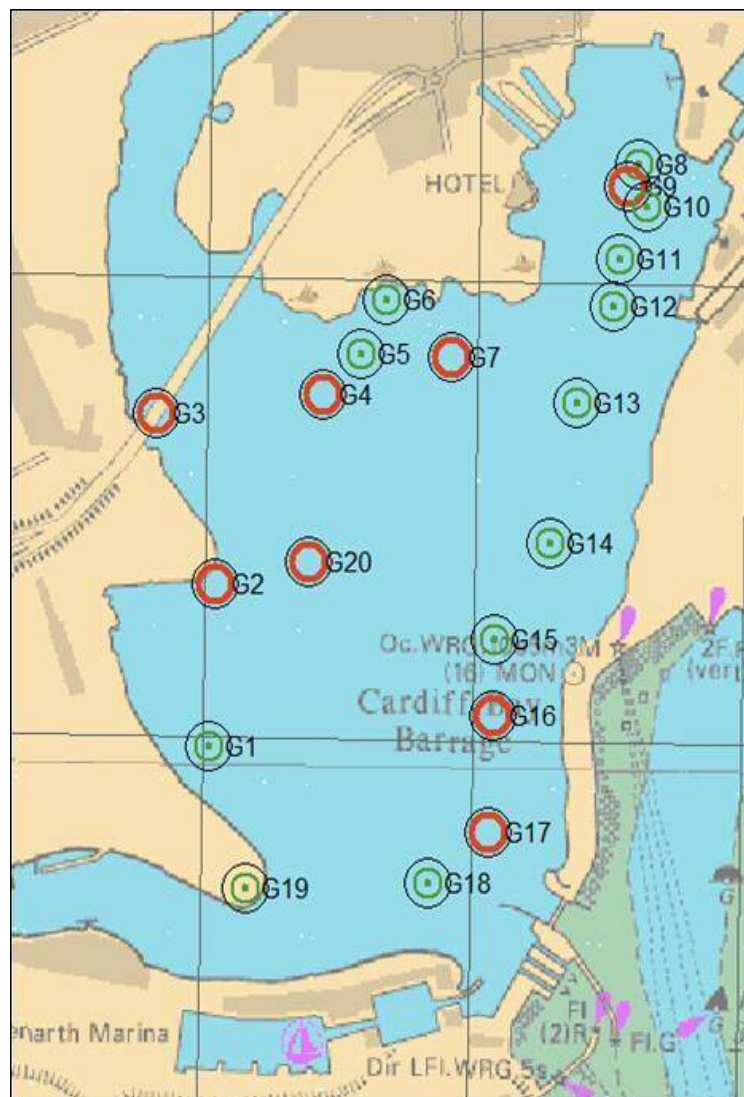


Figure 6.5: Sites within Cardiff Bay where grabs were taken to determine Zebra mussel presence. Green circles: sites where mussels were found, red circles: sites where no mussels were present.

Table 6.1: The number and size of zebra mussels collected in Cardiff Bay during July 2014

Site	Total	<5 mm	5-8 mm	9-16 mm	17-24 mm	>25 mm
5	275	0	16	226	33	0
6	58	0	7	49	2	0
8	8	1	0	3	1	0
10	68	1	3	62	2	0
11	151	18	65	64	4	0
12	545	1	21	441	64	18
13	829	0	13	665	250	6
14	75	0	1	60	14	0
15	600	0	1	106	460	33
18	925	52	227	516	128	2
19	1	0	1	0	0	0

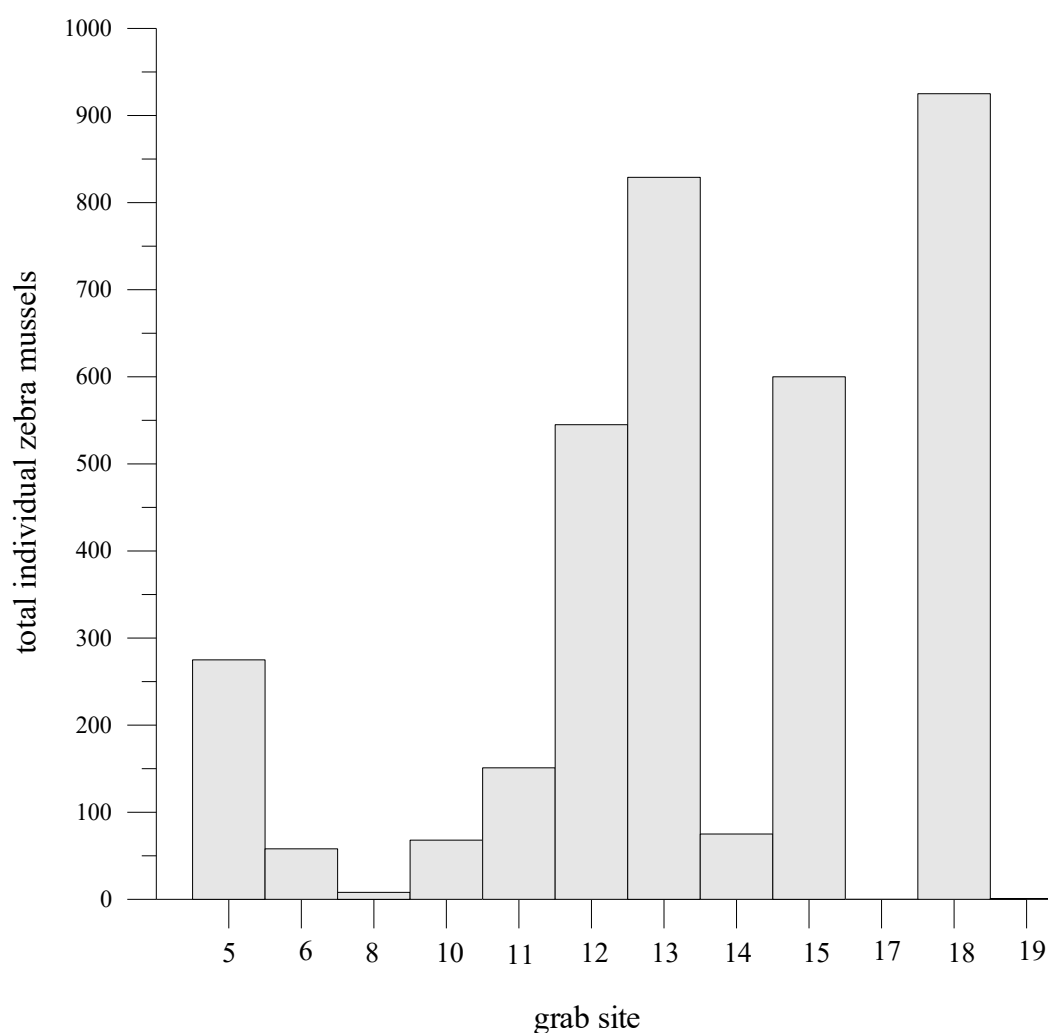


Figure 6.6: Total number of zebra mussels recorded from three grabs at each site in Cardiff Bay in July 2014.

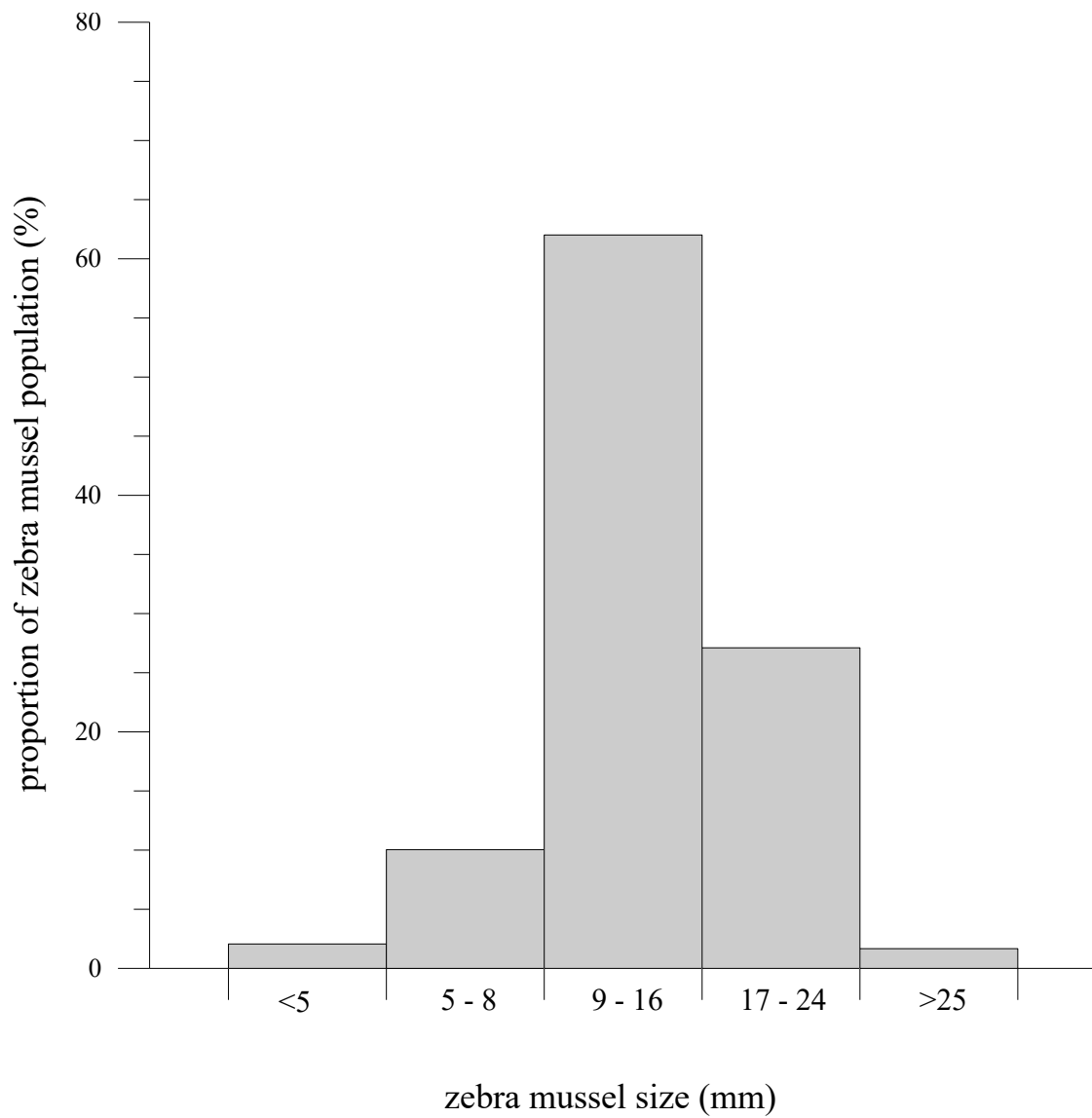


Figure 6.7: The size and total percentage of zebra mussel population recorded from 20 sample sites in Cardiff Bay in July 2014.

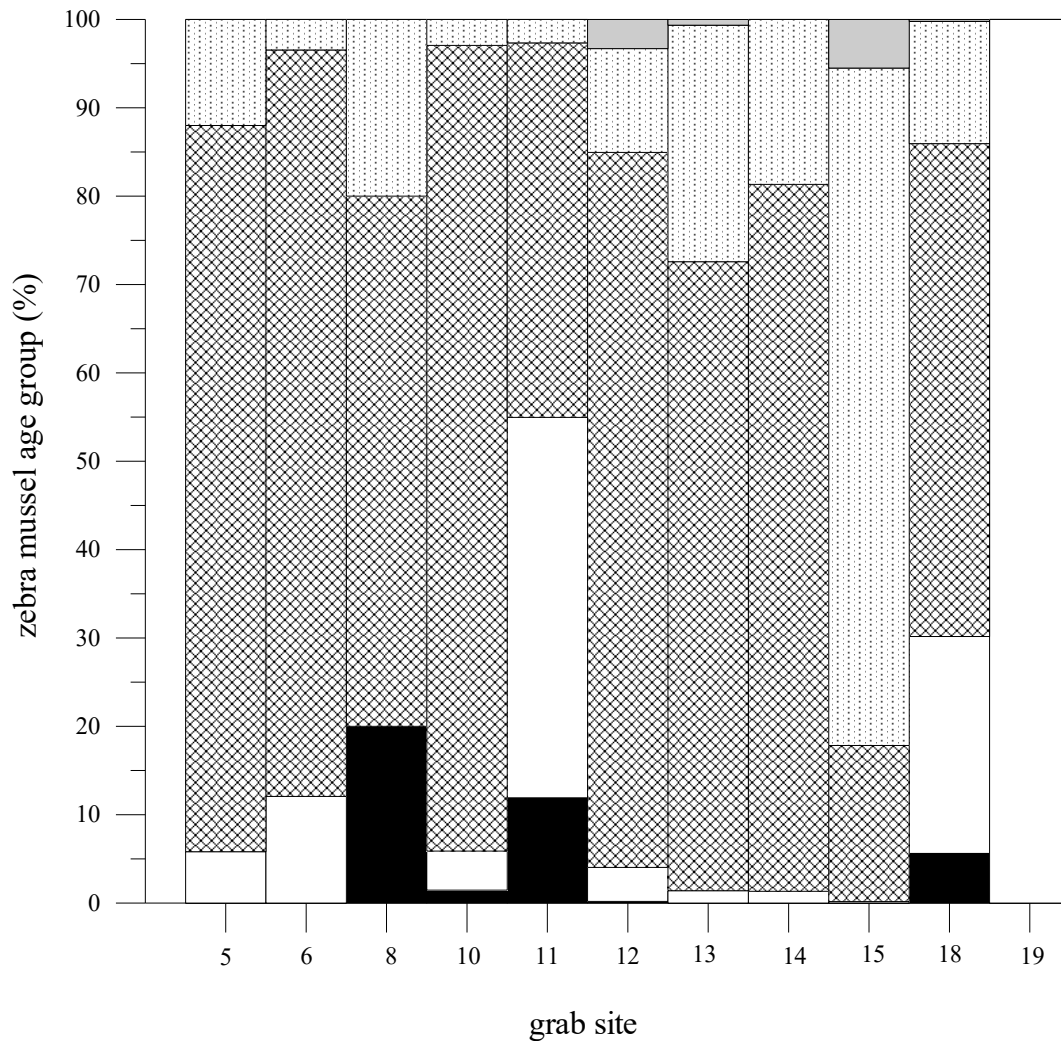


Figure 6.8: Percentage composition of zebra mussels per size at each grab site in Cardiff Bay taken in July 2014. black bars: <5mm, white bars: 5-8mm, hatched bars: 9-16mm, dotted bars: 17-24mm, grey bars: >25mm.

6.6 Discussion

The zooplankton population in Cardiff Bay is significant enough to influence the phytoplankton biomass of Cardiff Bay, however it was not abundant in phytoplankton-grazing species such as *Daphnia*, likely as a result of predation by fish. The zooplankton population experienced seasonal succession and did not vary spatially within the lake. Therefore, the growing population of zebra mussels found in the lake are likely to be filter feeding the

phytoplankton biomass, aiding the hydrological control of the phytoplankton community of the lake.

6.6.1 The Role of Top-Down Processes in Cardiff Bay

The artificial lake of Cardiff Bay is controlled by lake flushing rate and the presence of an aeration system. As a result, the lake does not stratify at any time of year and is polymictic, with no spatial or vertical variation within the main lake body. The lake contains a healthy phytoplankton population which is not nutrient, or light limited. However, the fast flushing rate of the lake restricts the ability of the phytoplankton biomass to increase therefore levels remain at very low throughout the year. The introduction of the zebra mussel (*Dreissena polymorpha*) into the lake system raises the question of whether it competes with the existing zooplankton population (Wu & Culver, 1991) regarding the top-down control through grazing of the phytoplankton population. Determining whether the zebra mussel population is an influential top-down control on the phytoplankton population within Cardiff Bay will enable the relationship between zooplankton and zebra mussels to be explored. The outcome of which will allow management recommendations to maintain water quality to be made accordingly.

6.6.2 Zooplankton Grazing in Cardiff Bay

There was a relatively significant zooplankton population found in Cardiff Bay. As a comparison, Harapan Lake, Malaysia is a small (7,200 m²) artificially formed lake and has an overall average of 78 individuals L⁻¹ (Ismail & Adnan, 2016); Lake Timsah, Egypt (area of 16 km²) has an average count in summer of over 40,000 ind. L⁻¹ (El-Sherbiny et al., 2011). The seasonal variation of zooplankton biomass (represented by zooplankton individuals L⁻¹) is a typical seasonal succession witnessed within temperate lakes, where summer exhibited the highest zooplankton biomass and individuals L⁻¹ are at their lowest during the winter (Mclaren, 1963, Mckinstry & Campbell, 2018). Zooplankton productivity often increases during summer as temperatures rise and food becomes available (Mclaren, 1963).

There was no evidence that zooplankton grazing had an impact on phytoplankton biomass in Cardiff Bay. The dominant species of zooplankton found in Cardiff Bay were not efficient

grazer species of phytoplankton. *Daphina* are often the favoured zooplankton to control phytoplankton as they are effective grazers of cyanobacteria (Ekvall et al., 2014).

No relationship was found between phytoplankton and zooplankton biomass. This may be as a result of the main food source of the dominant species of zooplankton not being the phytoplankton present in the lake, or the low biomass of phytoplankton prevented any relationship being formed.

There was no spatial difference in zooplankton within Cardiff Bay likely as a result of the polymictic nature of the lake. The fast flushing rate and mixing induced by the aeration system keeps the lake well-mixed, preventing spatial variation. However, the absence of a relationship between zooplankton and lake flushing rate, and the size of the zooplankton population in Cardiff Bay indicates that there is another variable controlling the rate of zooplankton growth, other than by lake flushing. Air temperatures were found to contribute to this variation of zooplankton growth. An increase in temperature resulted in an increase in zooplankton. This is likely to be an indirect relationship; an increase of air temperature causes water temperatures to increase and zooplankton productivity rises (Heinle, 1969). The relationship between zooplankton and air temperature was stronger than the relationship with water temperature. This is likely as air temperatures are often higher than water temperatures, causing the correlation to appear more significant.

The zooplankton biomass of Cardiff Bay may also be partially controlled by fish grazing. Zooplankton populations found in lakes with high fish densities are often dominated by small copepods (Hunt & Matveev, 2005), which was the second most abundant zooplankton found within the lake. Cardiff Bay supports a healthy population of fish and it contains a fish-pass as the area is particularly important for the run of migratory fish such as salmon (*Salmo salar*), sea trout (*Salmo trutta*) and sea lamprey (*Petromyzon marinus*), the latter of which has a population larger than any other UK estuary (Cardiff Harbour Authority, unknown). *Daphina* are a preferred food source for many fish species, possibly attributing to the low numbers of this zooplankton species found in the lake. As they are relatively large-bodied, *Daphina* often rely on thermal stratification to avoid predation by residing in low-oxygen areas in the water body where fish do not reside (Klumb et al., 2004). However, the aeration system and the fast flushing rate of the lake prevent any thermal stratification in Cardiff Bay, making any *Daphina* available for predation by fish.

6.6.3 The Role of Zebra Mussels in Cardiff Bay

The zebra mussel population of Cardiff Bay has increased since 2009, with mussels being found at 40% more samples sites in 2014 than in 2009. Mussels were found throughout the lake, with no preferred location. The majority of the population was comprised of 1-2 year old mussels. This age group is the most reproductive (Ram et al., 1996), indicating that the zebra mussel population is likely to still be expanding. Zebra mussels prefer to reside on hard substrate (Ram et al., 1996), these areas can be found on the barrage walls, along the lake walls at the sides of the Bay and the structures within the lake, particularly found in the marina, or the pipes of the aeration system. It was not feasible to sample these sites during this study; therefore, it is likely that the estimation of the size of the zebra mussel population within Cardiff Bay is low. Therefore, the predicted filtration rates are likely an underestimation. Zebra mussels reduce high numbers of phytoplankton by filter feeding (Jantz & Neumann, 1998), therefore their presence, and the predicted population makes the hypothesis that zebra mussel filtration is likely to be contributing to the control of the phytoplankton population within Cardiff Bay a reasonable one. It is also likely that the zebra mussel population is helping to limit the cyanobacteria in Cardiff Bay as it is one of the preferred phytoplankton for them to feed on (Aldridge et al., 2004). Zebra mussels also filter feed on crustacean zooplankton such as *Bosmina* (Aldridge et al., 2004), therefore they have the potential to limit zooplankton biomass too.

6.6.4 The Implications for Future Management of Cardiff Bay

The impact of zebra mussels on freshwater ecosystems have arguably been greater than many anthropogenic impacts (Strayer, 2009). The mussels are able to cope with increasing water temperatures (Griebeler & Seitz, 2007) and are likely to thrive under the threat of an increase in temperature as a result of climate change. The further impact that zebra mussels will have on freshwater ecosystems is unknown. As zebra mussels are an invasive species, the immediate management response is removal. However, this is notoriously difficult, if not impossible, therefore it is worth considering, particularly as Cardiff Bay is an artificially formed lake, whether the presence of the zebra mussel could be advantageous, or even positively aid the management of the phytoplankton population.

6.7 Conclusions

The top-down biological controls on the phytoplankton community of Cardiff Bay were investigated for the period between April 2012 and April 2014. The zooplankton abundance and diversity were determined using individual counts and species identification from samples taken fortnightly in the lake. The potential for zebra mussel filtration of the phytoplankton community in Cardiff Bay was also discussed, and grabs were collected at 20 sites in the Bay during Jul 2014 to determine the extent of the zebra mussel population. To manage the phytoplankton population of Cardiff Bay, the biological controls must first be understood.

The zooplankton counts collected throughout Cardiff Bay determined that the population was significant but did not show any spatial variation. The population was not abundant in phytoplankton grazing species such as *Daphnia*, and there was no significant relationship found between zooplankton and phytoplankton within the lake. Therefore, it is unlikely that zooplankton grazing plays a significant role in phytoplankton biomass control. The diversity of zooplankton within the lake was likely to be controlled by fish predation which removed the larger zooplankton from the water column and was aided by the aeration system making zooplankton accessible at sufficient DO conditions through mixing.

The grabs taken at various locations within the lake indicated that the zebra mussel population of Cardiff Bay was expanding. It was also found to be of reproductive age. Both of these factors indicate that the filtration rate by zebra mussels calculated for the lake was an underestimation, therefore it is sensible to hypothesise that zebra mussels are controlling the phytoplankton population of Cardiff Bay.

The grazing pressure from zooplankton, but more significantly from zebra mussels within Cardiff Bay are contributing to the limitation of phytoplankton biomass and therefore cyanobacteria within the lake. Therefore, it can be determined that the phytoplankton biomass of Cardiff Bay is being controlled primarily by hydrological factors (lake flushing rate) and in part by biological factors (grazing).

GENERAL DISCUSSION

The study site of Cardiff Bay, South Wales was presented in this thesis as an artificial lake which has been managed since its formation in 2001. Despite this ongoing management, the state of the phytoplankton population within the lake in relation to water quality has not been analysed in depth since its formation, creating the overarching aim of this study; to use phytoplankton as the focal point for the management of Cardiff Bay, by determining factors which control them.

It has been increasingly recognised that to maintain the water quality of aquatic ecosystems worldwide, management is essential, particularly due to the increasing threat of a changing climate. The introductory chapter of this thesis describes the environmental variables that influence lake systems and their potential to change in the future. It highlights the important role that phytoplankton play within a lake food-web and how monitoring their community structure can provide information about a lake system which is pivotal to the success of lake management. The common factors that control a lake phytoplankton population were introduced and discussed, with reference to current methods of lake management and water quality action plans.

This study has demonstrated that Cardiff Bay is a highly managed artificial freshwater system that is unlike other freshwater lakes and reservoirs in the UK, or elsewhere. It has been determined that the typical processes of ‘bottom-up’ nutrient limitation in conjunction with trophic cascade, or ‘top-down’ control (Scheffer et al., 2001) are of far less importance than the hydrological control of lake flushing rate. Having said this, although nutrient control of phytoplankton biomass does not currently appear to be important, biological control through filter feeding by zebra mussels, and potentially zooplankton grazing, may contribute alongside the dominating hydrological control of lake flushing rate. Once the phytoplankton population of Cardiff Bay has been outlined, this discussion will draw together the evidence for these statements regarding; Hydrological control (Section 7.2), Nutrient control (Section 7.3), and Biological control (Section 7.4) of the phytoplankton biomass in Cardiff Bay.

Cardiff Bay was constructed primarily for recreational use (Cowell & Thomas, 2002) therefore it can be assumed that the main aim of managing the lake is to maintain a clear water system through the prevention of harmful algal blooms (HABs), rather than to attain a balanced ecological system. This could be achieved through the maintenance of a macrophyte/microalgal balance, something that is often the goal of water managers (Srivastava et al., 2008) but is beyond the scope of this study. One of the critical management implementations within Cardiff Bay is the presence of the aeration system, used to ensure that DO does not drop below 5 mg L⁻¹ to allow vital fish stocks to survive; and an indication that the ecological system within the lake may not be at the forefront of management goals, but it is still being considered.

The low phytoplankton biomass found in Cardiff Bay and the likely abundance of zebra mussels prevents the lake from being identified as a water body of importance with regard to ecological status under the Water Framework Directive classification (the small sectioned area of reedbed within the Cardiff Bay Wetland Reserve may be an exception) (Priestley, 2015). With a focus on maintaining a clear water system in Cardiff Bay by ensuring that phytoplankton biomass levels are low, and that HABs do not form, it is paramount that the management of the lake considers changes in climate that may take place in the future, and the consequences of them. Therefore, the sections at the end of this discussion will focus on; Climate Change Risk (Section 7.5), and Management Recommendations (Section 7.7).

7.1 The Phytoplankton Population in Cardiff Bay

The main lake body of Cardiff Bay supported a healthy and productive phytoplankton population, although phytoplankton biomass was low (Chapter 4, Section 4.5.1). The phytoplankton community was dominated by diatoms throughout the study and displayed seasonal succession; where biomass was at its highest in spring and lowest during winter (Chapter 4, Section 4.5.2). The remainder of the phytoplankton community was mostly comprised of chlorophytes and cryptophytes, the former of which also displayed seasonal succession. There was an insignificant population of cyanobacteria, as levels were so low it did not present a threat to the lake ecosystem under the current lake conditions (Chapter 4, Section 4.7).

Photic depth within the lake did not have any relationship with phytoplankton, therefore there was no light limitation in the Bay (Chapter 4, Section 4.5.4). Photic depth did not vary seasonally and the only difference in photic depth within the main lake body was experienced next to the marina at site C, where macrophyte presence was likely to have cleared the water column (Chapter 3, Section 3.6.8).

The hypotheses from this study regarding phytoplankton, and their outcomes are outlined below:

- There will be seasonal variation in phytoplankton biomass, productivity and assemblage within the lake.

Phytoplankton biomass in the lake varied seasonally, with the highest biomass concentrations being observed during spring. However, there was no variation in productivity or assemblage in the lake. The phytoplankton community was healthy and dominated by diatoms throughout every season (Chapter 4, Section 4.6.2).

7.2 Hydrological Controls in Cardiff Bay

Cardiff Bay is a polymictic lake; the results presented in Chapter 3, Section 3.6.6 show that the lake did not stratify at any location during any time of the year and there was no evidence of a thermocline being present. As discussed in Chapter 1, this is relatively unusual for a temperature lake system and was caused by a combination of the fast lake flushing rate and the aeration system constantly mixing the water column. This constant mixing not only prevented any variation with depth of water temperature, DO or phytoplankton biomass, there was no horizontal spatial variation in water temperature, DO, phytoplankton biomass or photic depth within the main lake body.

The lake was subject to seasonal temperature cycles typical of a temperate lake, where water temperatures peaked during summer and temperatures were at their lowest in winter (Chapter 3, Section 3.5.2). There was also a seasonal change in precipitation, with autumn and winter levels being higher than those experienced in spring and summer (Chapter 3, Section 3.5.3).

7.2.1 The Influence of the Aeration System in Cardiff Bay

The DO concentration of Cardiff Bay showed no relationship with temperature or phytoplankton biomass (Chapter 3, Section 3.6.7). It is likely that the phytoplankton biomass within the lake was too low to impact DO concentrations, and the use of the lake aeration system may have overshadowed any natural variation in DO caused by phytoplankton. The aeration system was used to ensure that lake DO concentrations were maintained at, or above, the legally binding 5 mg L^{-1} , and as a result caused the frequent supersaturation of DO in the lake (Chapter 3, Section 3.6.7). However, the process of aeration not only ensures that the oxygenated surface waters are transported to the bottom of the lake to increase DO concentrations, it mixes the water column. This process prevented any vertical stratification of DO in Cardiff Bay, as well as the formation of a thermocline, a prominent feature of many lakes.

Having said this, the aeration system was not solely responsible for the regulation of lake DO concentrations. DO in Cardiff Bay did vary seasonally in response to temperature changes, suggesting that the use of the aeration system did not eradicate all the natural seasonal changes of the lake (Chapter 3, Section 3.6.7).

7.2.2 The Role of the River Taff and River Ely

The River Taff and River Ely experienced variations in environmental factors that the main lake body did not. Both rivers experienced DO concentrations that were different to the main body of the lake; supporting the theory that the aeration system does have an impact in the lake itself.

The photic depth in the River Taff was lower than the sites within the lake, as a result of a shallower water column and therefore a more turbid environment (Chapter 3, Section 3.6.8). The River Taff and River Ely had a lower phytoplankton biomass than the main lake body. The River Taff experienced higher nutrient concentrations than the sites surveyed in the main body of the lake (Chapter 5, Section 5.5.1).

Increases in precipitation caused River Taff and River Ely discharge rates to increase, therefore rates were at their highest during autumn and winter (Chapter 3, Section 3.5.4). Precipitation and the discharge of the River Taff and the River Ely played an important role in the seasonal variation within the main lake body, as both strongly influenced the flushing rate of the lake (Chapter 3, Section 3.6.5), causing the lake to be flushed more frequently in winter and less often in summer (Chapter 3, Section 3.5.5).

The river input from both the River Taff and River Ely is significant. Not only does the river discharge rate influence the flushing rate of the lake (Chapter 4, Section 3.5.5), indirectly the rivers also control the external input of nutrients into the lake, through the control of flushing rate (Chapter 5, Section 5.5.3.1).

7.2.3 The Significance of Lake Flushing Rate

The flushing rate of Cardiff Bay is relatively quick, making it the dominant natural hydrological variable that controlled processes within the lake (Chapter 3, Section 3.6.6). The turnover of the water in the lake is high, and due to the influence of the aeration system, constant mixing of the water column takes place. Cardiff Bay can be considered as being more like a slow-flowing river system rather than a typical lake as it has low phytoplankton biomass, and the absence of a thermocline, but most importantly a high flushing rate which reduces phytoplankton exposure times to enhanced nutrients which contributes to the prevention of nutrient limitation (Newman et al., 2006, Rueda et al., 2006).

The flushing rate of Cardiff Bay influenced phytoplankton abundance; the highest numbers of phytoplankton were found during periods of slower flushing rate (spring and summer), where water retention was at its greatest. During periods of slower flushing the phytoplankton productivity outweighs the loss by outflow, allowing the biomass to increase in the lake. Cryptomonads were influenced by flushing rate during the spring, summer and autumn. Chlorophytes and cyanobacteria were influenced by autumn flushing rates. These relationships during autumn are due to the faster lake flushing rates causing phytoplankton biomass to be lost quicker than it can be replaced, combined with reduced productivity during this season.

7.2.4 Hydrological Hypotheses

The hydrological hypotheses and their outcomes from this study are outlined below:

- Seasonal air temperature and precipitation changes will induce seasonal lake water temperature change.

There was a seasonal variation in air temperature, which resulted in the seasonal variation in lake water temperatures of Cardiff Bay. Although a seasonal change in precipitation was observed, this did not influence the water temperature of the Bay (Chapter 3, Section 3.5.3).

- The photic depth of the lake will fluctuate seasonally due to changes in water column stability. Spatial variation in photic depth will occur due to differences in sampling site characteristics (e.g. susceptibility to mixing, sediment re-suspension).

There was no seasonal or spatial variation in photic depth within Cardiff Bay as a result of the fast lake flushing rate and aeration system causing a polymictic system and preventing any spatial differences from occurring in the main body of the lake. However, the River Taff experienced a lower photic depth than the other sites studied due to the shallow nature of the water column and the impact of river discharge on sediment resuspension and turbidity (Chapter 3, Section 3.6.8).

- Lake dissolved oxygen concentrations will vary spatially and temporally due to the influence of water temperature and thermal stratification.

Dissolved oxygen concentrations did not vary within the main body of the lake as a result of the polymictic nature of the system. However, DO concentrations in the River Taff and River Ely varied from those in the main body of the lake as a result of being subject to different characteristics. Thermal stratification did not occur in the lake due to mixing of the water column caused by the aeration system and the fast lake flushing rate, however DO concentrations altered seasonally as a result of water temperature changes, and likely the influence of the aeration system (Chapter 3, Section 3.6.7).

- High precipitation rates will cause an increase in river discharge and therefore cause lake flushing rates to quicken. Temporal changes in flushing rate will occur as a result of seasonal changes in precipitation.

Precipitation influenced River Taff and Ely discharge rates, an increase in these hydrological factors caused the flushing rate of Cardiff Bay to quicken. Seasonal variation occurred, causing summer to be characterised by lower precipitation and river discharge rates, resulting in a slower lake flushing rate (Chapter 3, Section 3.6.5).

- The phytoplankton biomass, productivity and assemblage within the lake does not vary spatially or with depth within the Bay.

The result of a fast flushing rate and the aeration system mixing the water column of Cardiff Bay prevented the biomass, productivity and assemblage of phytoplankton within the lake from varying spatially or with depth (Chapter 4, Section 4.6.2).

- Phytoplankton biomass is controlled in part by flushing rate, such that increases in flushing rate reduces biomass due to out-flushing, and low flushing rate reduces biomass out-flushing.

The phytoplankton biomass of Cardiff Bay was significantly controlled by lake flushing rate. A slower flushing rate during summer slowed the out-flushing of biomass from the lake, allowing concentrations to increase (Chapter 4, Section 4.6.5).

7.3 Nutrient Controls in Cardiff Bay

Lake phytoplankton populations are often controlled by ‘bottom-up’ processes, specifically nutrient availability (Du et al., 2015). Phosphorus is the most common limiting nutrient in a lake as it is required for growth by most phytoplankton. However, some phytoplankton, such as diatoms, primarily require silica, this can cause relative limitation; where nutrient concentrations are adequately available, but one nutrient is favoured over the others, In the case of diatoms the favoured nutrient is silica. This is the case in Cardiff Bay as a result of diatoms being the most abundant phytoplankton throughout the timescale of the study.

The polymictic characteristics of Cardiff Bay caused concentrations of phosphorus, nitrogen and silica to be uniform throughout the water column (Chapter 5, Section 5.5.1). Spatially within the lake, nutrient concentrations were not significantly different, except for at two sites. In the River Ely, the concentrations of nitrate and total dissolved Nitrogen (TDN) were higher than the sites studied in the lake (Chapter 5, Section 5.5.1). The discharge rates in this area may have reduced the opportunity for denitrification to occur as a result of the reduced impact between sediment and the water column (Reisinger et al., 2016). The site in the River Ely was also the deepest site studied and it has been documented that N removal can decrease with depth (Reisinger et al., 2016). The area adjacent to the wetlands experienced low TDN concentrations in comparison to the rest of the lake (Chapter 5, Section 5.5.1). One explanation of this is the close proximity of the site to the wetland area, allowing the impact of nitrogen uptake by wetland plants to reduce nitrogen in the surrounding waters (Saunders & Kalff, 2001).

7.3.1 Nutrient Limitation in Cardiff Bay

There was no evidence of the phytoplankton population in Cardiff Bay being limited by nutrients. The Redfield Ratio of 16:1:1 for TDN:TP:Si was used to differentiate between the limiting nutrients. The TDN:TP ratio was lower than 16:1 during the whole study (with the exception of 2 months), indicating that the lake was not limited by TP (Chapter 5, Section 5.5.3). The ratio of TP:Si was below 1:1 for the majority of the study, however ratios between 2012 and 2014 indicated that silica was limiting relative to TP in the lake. TDN was found to be limiting relative to silica throughout the study as TDN:Si was only above 1:1 for 2 out of the 136 months, but the fortnightly study between June 2012 and April 2014 indicated that silica was limiting relative to TDN (Chapter 5, Section 5.5.8). Ultimately however, the phytoplankton population were not seen to be limited by TDN, TP or Si. Only 57% of diatom biomass changes were influenced by silica in spring and total phytoplankton biomass was only influenced by nutrient concentrations during autumn (Chapter 5, Section 5.5.10.1). Phytoplankton biomass did not increase within the nutrient addition mesocosm bags (Chapter 5, Section 5.5.11.1), when TP and TDN concentrations were increased. There was also no increase in phytoplankton biomass in the lake when TP was reduced from inputs on the River Taff and Ely (Chapter 5, Section 5.5.11.1). Both processes indicate that the phytoplankton

population is not controlled by nutrient concentrations. It was determined that the flushing rate of the lake had a stronger influence on phytoplankton biomass in Cardiff Bay than nutrient availability.

7.3.2 The Importance of Internal and External Nutrient Loading in Cardiff Bay

The external loading of nutrients in Cardiff Bay by precipitation and river discharge is more significant than internal loading from lake sediments. In addition to this, the impact of lake flushing rate determines the length of time nutrients are available for utilisation. The absence of P-limitation within Cardiff Bay discards the concern for phosphorus release from the lake sediment, despite the significant pools of Fe/Al-P and labile-P found within them (Chapter 5, Section 5.6.5). There was no conclusive evidence of vertical changes in distribution of phosphorus within the sediment, suggesting that there is little upward migration of dissolved P from deeper sediment layers. This indicates that only low-level internal loading of P from the Fe-bound P fraction into the water column may be taking place. However, this loading is not currently at levels high enough to negatively impact the ecosystem.

7.3.3 Bottom-Up Control Hypotheses

The bottom-up hypotheses and their outcomes from this study are outlined below:

- The concentration of phosphorus, nitrogen and silica within the lake do not vary spatially or with depth.

The concentration of phosphorus, nitrogen and silica did not vary spatially in the main body of the lake or with depth, but nutrient levels in the River Ely were higher than those in the lake. Nutrient concentrations did not vary with depth (Chapter 5, Section 5.5.1).

- There is a seasonal variation in phosphorus, nitrogen and silica concentrations within the lake.

TP and silica demonstrated seasonal variation. TP concentrations peaked in summer and were at their lowest during winter (Chapter 5, Section 5.5.2.1). In contrast, silica concentration peaked in winter and were at their lowest during summer (Chapter 5, Section 5.5.4). There was no seasonal variation observed in SRP or nitrogen concentrations.

- Compared to nitrogen and silica, phosphorus is the most limiting nutrient in Cardiff Bay, causing phosphorus concentrations to determine phytoplankton biomass.

The phytoplankton biomass within Cardiff Bay was not limited by phosphorus, nitrogen or silica availability. Therefore, phosphorus concentrations did not determine the phytoplankton biomass within the lake.

- Phosphorus exchange between the lake-bed and the water column is relatively low due to the water column being well-mixed.

It is possible that low-level phosphorus exchange between the lake-bed and the water column is taking place within Cardiff Bay. However, the biological utilisation of phosphorus in the lake is being restricted by the fast flushing rate and mixed water column, minimising the diffusional release of P from the sediment (Chapter 5, Section 5.6.4).

7.4 Biological Control

The phytoplankton within a lake can be controlled by grazing; top-down control by organisms higher up the trophic level. In this study the top-down control of phytoplankton in Cardiff Bay was undertaken through the study of zooplankton and zebra mussel abundance.

7.4.1 Zooplankton in Cardiff Bay

The zooplankton biomass of Cardiff Bay appeared to be significant enough to impact the phytoplankton population through grazing. However, the dominant zooplankton were Copepods which are not efficient grazer species and there were few Daphnids and Bosmina present (Chapter 6, Section 6.5.1). Therefore, zooplankton grazing of the phytoplankton

population was likely to be small but not insignificant in summer, when the zooplankton population increased.

Zooplankton biomass was able to increase to significant levels in the lake, suggesting that lake flushing rate was not the only control on the lake system. This indicates that the phytoplankton biomass within the lake must be exposed to some top-down grazer control, as well as hydrological control through flushing.

There was no spatial difference in zooplankton biomass within the lake as a result of the polymictic characteristics of the system (Chapter 6, Section 6.5.1), and no relationship between zooplankton biomass and lake environmental factors was found. The low phytoplankton biomass prevented any significant relationship from being formed with zooplankton biomass either.

7.4.2 The Zebra Mussel Population of Cardiff Bay

The zebra mussel population within Cardiff Bay increased over time. Young individuals of reproductive age were the most abundant age group, indicating that the population is healthy, and numbers are likely to increase in the future (Chapter 6, Section 6.5.3). The presence of zebra mussels in the lake is important as their filtration rate is fast enough to negatively influence the phytoplankton biomass, as well as the zooplankton biomass within the lake. Therefore, in addition to the hydrological control of phytoplankton biomass, top-down control through zebra mussel grazing is also occurring.

7.4.3 Biological Control Hypotheses

The bottom-up hypotheses and their outcomes from this study are outlined below:

- The zooplankton of Cardiff Bay varied seasonally but did not alter spatially due to the lake being well-mixed.

The zooplankton biomass of Cardiff Bay varied seasonally, experiencing a peak in biomass during summer. There was no spatial difference in zooplankton as a result of the fast lake flushing rate and aeration system mixing the water column (Chapter 6, Section 6.6.2).

- Zooplankton control the phytoplankton biomass within the lake through grazing.

The zooplankton biomass was not abundant in efficient grazers, but the significant biomass levels would have resulted in zooplankton grazing contributing to the control of phytoplankton biomass in the lake (Chapter 6, Section 6.6.2).

- The zebra mussel population of Cardiff Bay has the potential to contribute to top-down control through phytoplankton filtration.

The growing zebra mussel population of Cardiff Bay is likely to be influencing the phytoplankton biomass in the lake, and the indication that the zebra mussel population is growing makes it likely that in the future that they could significantly contribute to the top-down control of phytoplankton in the lake (Chapter 6, Section 6.6.3).

7.5 Summary of Cardiff Bay

The main lake body of Cardiff Bay supported a healthy and productive phytoplankton population, despite biomass being low. Phytoplankton biomass was primarily controlled by a combination the fast lake flushing rate; where phytoplankton removal by flushing exceeded phytoplankton productivity. Top-down control; primarily through zebra mussel filtration and accompanied by zooplankton grazing, also contributed to the control of the population. The phytoplankton population was not found to be nutrient or light limited and did not contribute to the DO concentration of the lake.

The polymictic nature of the lake was caused by the combination of a fast lake flushing rate and water column mixing by the aeration system. This prevented any spatial or vertical differences from being observed within the main body of the lake and highlighted how the conditions within the River Taff and Ely were different to those experienced within the lake. The characteristics of the lake did not favour cyanobacteria, and the population was dominated by diatoms which thrive in a mixed water column.

Despite the combination of a fast flushing rate and the use of an aeration system, Cardiff Bay was still subject to natural seasonal variations. Warmer temperatures and lower precipitation rates experienced through spring and summer caused a slowing of the lake flushing rate, an increase in phytoplankton and zooplankton biomass, lower DO and a variation in TP and silica concentrations. These relationships can be seen in Figure 7.1.

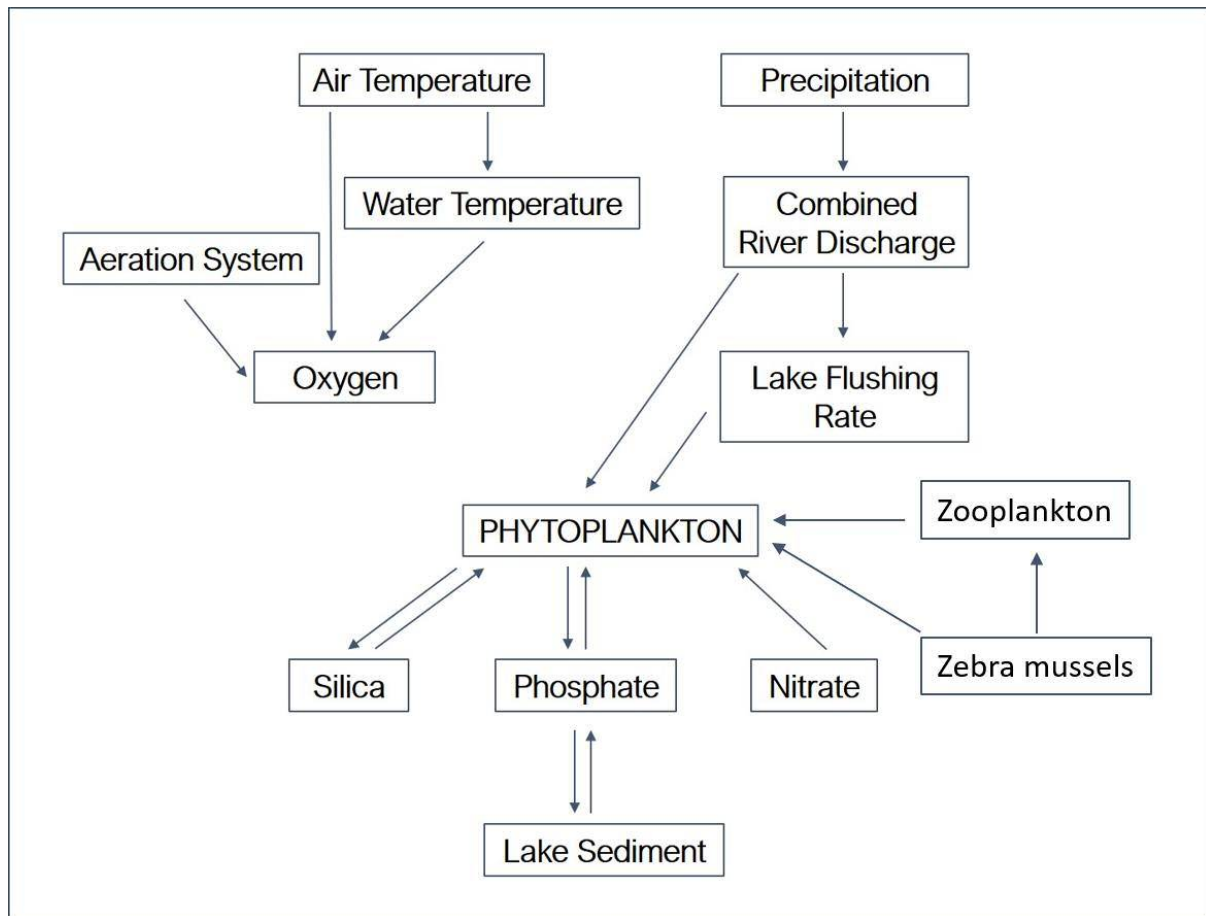


Figure 7.1: A schematic of the processes that take place in Cardiff Bay and the relationships between them

Combining the results from this study, the trophic state of Cardiff Bay can be determined as oligotrophic according to Carlson's Trophic State Index (TSI). This index uses chlorophyll *a*, Secchi depth and TP measurements to calculate a TSI. The seasonal TSI for Cardiff Bay was calculated for between Summer 2012 and Spring 2014 (Table 7.1), and the lake was classified as oligotrophic during every season. An oligotrophic lake can be defined as a clear water lake,

which has the potential to become anoxic during the summer (Devi Prasad & Siddaraju, 2012). Oligotrophic lakes are often unproductive, which Cardiff Bay arguably is, however, characteristically an oligotrophic lake has a low supply of nutrients, which is not the case for Cardiff Bay. This indicates the limitations of using an index to calculate the trophic state of a lake, as well as highlighting the unusual nature of Cardiff Bay; low phytoplankton biomass is commonly caused by nutrient control, rather than being controlled by hydrological factors.

Table 7.1: The seasonal Trophic State Index for Cardiff Bay

Season	TSI
Summer 2012	40.15
Autumn 2012	39.49
Winter 2013	39.07
Spring 2013	39.98
Summer 2013	40.11
Autumn 2013	39.60
Winter 2014	39.40
Spring 2014	39.96

7.6 Climate Change Risk

The importance of lake flushing rate for the hydrological control of phytoplankton biomass was a major finding of this study. Driven by the discharge rates of the River Taff and River Ely, the flushing rate is also linked to precipitation rates within the rivers' catchments. Whilst

it would not be acceptable to utilise the data from this study to attempt accurate modelling of climate change impacts on Cardiff Bay, it is important to highlight how the controls on the phytoplankton population may change in order to identify climate change risk potential for the management of the lake.

The risk of climate change in Cardiff Bay is approached from the perspective of the IPCC predictions, where temperatures in summer are elevated and summer precipitation reduces. The annual average air temperature in Wales has been predicted to rise by 1.3 °C by the 2020's, 2.0°C by the 2040s and 3.3°C by the 2080s under the medium emissions scenario (Welsh Assembly Government, 2009b). As a result of the relationship that air temperature has with the water temperature of Cardiff Bay (Chapter 3, Section 3.5.2), it is extremely likely that lake water temperatures will rise in the future. An increase in lake water temperature will reduce DO concentrations, firstly as a result of a reduction in oxygen solubility, and secondly as increasing temperatures result in the higher metabolism and growth of organisms, which utilise DO. In addition, warmer temperatures have the potential to impact the biological controls in the Bay. It can be hypothesised that water temperatures will increase the metabolism and growth rate of the zebra mussel stock, inducing a greater filtration pressure and enhancing biological control. However conversely, the zooplankton community structure can become unstable under elevated temperatures (Moore, 1996), therefore it could also be hypothesised that elevated summer temperatures might reduce grazing pressure on the phytoplankton in the lake.

Average summer precipitation has been projected to decrease by approximately 30% by 2050 in South Wales, and average winter precipitation has been projected to increase (Welsh Assembly Government, 2009b). This change in precipitation in the catchment of Cardiff Bay has the potential to impact river discharge rates, ultimately decreasing lake flushing rate in summer. This will not only reduce the dilution of TP and SRP concentrations in the lake therefore increasing nutrient availability, but more significantly a reduction in lake flushing rate will occur. These changes in lake characteristics provide the potential for a substantial increase in phytoplankton biomass as phytoplankton productivity is given the opportunity to outweigh the outflow of phytoplankton during the summer.

The impact of these changes in climate could be hypothesised to be especially significant during summer. The combination of an increase in water temperature and a decrease in

precipitation could cause the lake to stratify. A reduction in lake flushing rate has the potential to reduce water column mixing, and a rise in temperature, particularly within the epilimnion of the lake, could result in a difference in temperature between the surface and bottom waters of the lake, thus causing the lake to stratify and a thermocline to form. External nutrient loading levels are likely to be maintained, as smaller levels of river discharge will provide a more concentrated nutrient supply to the lake. This, coupled with a decrease in outflow of nutrients from the lake as flushing rates slow, could facilitate the accumulation of a greater phytoplankton biomass within the Bay. A shift in the phytoplankton community structure could take place, where diatoms are outcompeted by cyanobacteria, making the system vulnerable to the formation of HABs; increasing the importance of biological top-down control of the phytoplankton population. It would be overly speculative to predict the magnitude of the impact of temperature and precipitation on the system; however, it is suggested below (Section 7.9) that this is an essential area of further study in Cardiff Bay.

The low phytoplankton biomass of the lake prevented strong relationships from being formed with the environmental variables within the Bay (Chapter 4, Section 4.5). However, several weak relationships were discovered. With an increase in phytoplankton biomass or a shift in species dominance as a response to a warming climate, these relationships are likely to strengthen and become significant and have the potential to heavily impact the current ecological system of the lake.

7.7 Study Limitations

The importance of flushing rate was highlighted throughout this study, however due to the absence of available discharge output data, it had to be assumed that the output equalled the input due to the lake volume being kept at a constant. This limited the method used to calculate flushing rate and the role that evaporation and transpiration might play within the lake system was not able to be determined.

The identification of phytoplankton and zooplankton species was not as comprehensive as intended at the onset of this study. This was largely due to the large amount of debris and damaged organisms present in the collected samples, both of which hindered species identification and were likely caused by high flushing rates and water column turnover.

The effect of the aeration system on the mixing of the water column was not analysed as the use of the system was not documented. Understanding the natural mixing processes compared with the artificial mixing process within the lake would have been advantageous when analysing the effects of mixing on the phytoplankton community.

7.8 Management Recommendations

This study has highlighted important implications for the management of Cardiff Bay to maintain the clear water state of the lake in the future, under the stressors of changes in climate. The following recommendations are suggested in order to advance understanding of the controls on water quality and in particular the phytoplankton biomass of Cardiff Bay.

The aeration system is an essential component within the lake, not primarily to maintain the DO concentrations of the lake, as the water column is often supersaturated, but to ensure that the water column is continuously mixed. This is especially significant if a reduction in energy from river discharge occurs, slowing lake flushing rates. Under the predicted conditions the aeration system may still be required for oxygenation purposes to maintain DO above 5 mg L⁻¹, as an increase in lake water temperature will cause DO concentrations to become vulnerable to change. However, the potential decrease in water column mixing as a result of slowed lake flushing rates may be more significant and require an increased use of the aeration system to maintain a well-mixed system. The aeration system should be accurately monitored and its use dependant on the level of vertical and horizontal mixing of the Bay water.

The flushing rate of Cardiff Bay has been found to be the main driver of phytoplankton biomass control. There is the potential in the future for river discharge to decrease low enough to cause an increase in phytoplankton biomass, particularly during summer where temperatures are predicted to increase. Therefore, careful monitoring of river inflow volumes and lake flushing rate is needed to determine if the flushing rate of the lake needs to be artificially maintained to prevent the formation of HABs. Careful study of biomass as a function of accurate inflow and outflow rates is required.

Although nutrient limitation was not found to be a controlling factor of phytoplankton biomass within the lake, it is still important to determine the risk of internal P loading. If external P

loading is reduced as a management option to mitigate reduced lake flushing rate, there is the potential for an increase in P release from lake sediment to occur. An in-depth study of sediment P dynamics and the potential for sediment Fe-P release should be undertaken.

It is difficult to justify an investigation into the zooplankton grazing within the lake based on the limited data presented in this study. However, the findings strongly implicate zebra mussel filtration as a control of phytoplankton biomass due to the large volume filtered by the growing population. Further investigation is needed to quantify this accurately, and importantly should determine how changes in mussel metabolism may result through elevated summer temperatures. Although they are an invasive species, they are renowned for being extremely difficult to eradicate from a lake once established, and although a radical and controversial suggestion, it would be beneficial to consider the presence of zebra mussels within Cardiff Bay as being advantageous for the lake as a natural way to control the phytoplankton population under the changing climate conditions predicted for the future.

Overall, modelling of the phytoplankton community of Cardiff Bay with regard to changes in flushing rate and elevated summer temperature is highly recommended. This can be done using a process-based computer model such as PROTECH (Phytoplankton Responses To Environmental Change) such as the study undertaken by (Elliott & Defew, 2012) which analysed the changes in retention time and temperature upon the phytoplankton in Loch Leven. This model would take the basic form:

$$\begin{aligned} \text{Phytoplankton Biomass} = & -f_1 (\text{river inflow volume}) -f_2 (\text{zebra mussel filtration}) \\ & -f_3 (\text{zooplankton grazing rate}) +f_4 (\text{nutrient availability}) \end{aligned}$$

Where f_1 to f_4 are functional relationships determined by the study areas suggested above, taking in to account climate change predictions for summer rainfall and temperature.

This study provides important findings which could contribute to the formation of reports for Cardiff Harbour Authority concerning the impact of lake flushing rate on Cardiff Bay, and resource management of the lake.

7.9 Future Research

To be able to analyse the lake system without the presence of an aeration system would be beneficial for determining the natural mixing of the lake and the consequences for management; it would aid the decision as to whether the aeration system is required and how the system would function without it and the consequences of it. This would likely need to be carried out ex-situ as the result of turning off the aeration system within the lake may have undesirable consequences which could lead to Cardiff Harbour Authority not meeting their water quality requirements.

It was evident that the mixing of the water column within this study was a vital characteristic defining the processes within the lake. The analysis of lake turbidity would complement this data and the addition of suspended sediment would strengthen the argument that the water column mixing is an important feature. The effect of wind-mixing on the system and its significance in comparison with discharge inputs would be advantageous.

Groundwater flow can be determined using radon-222 analysis (Kluge et al., 2007), where the sensitive natural tracer, radionuclide ^{222}Rn can be used to detect and determine groundwater inflow into lakes. In addition to the rate of inflow it can also provide information on the spatial distribution of groundwater inflow into a lake. This could enhance the understanding of inputs into the lake.

The effect of zebra mussels on the phytoplankton and zooplankton populations, and nutrient availability in Cardiff Bay could be investigated using extensive mesocosms simulating environments with and without zebra mussels present. This would be beneficial to the understanding of the impact these invasive species have on the ecosystem. The effect of the high flushing rate on the zebra mussels could also be analysed as an understudied research avenue which could analyse a zebra mussels' response to a new location and environmental conditions. Longer-term mesocosms to analyse the combination of nutrient enrichment, including the addition of silica addition and also flushing rates on the phytoplankton community would also expand these findings.

It was suggested that macrophyte growth within the lake may be influencing photic depth and sediment resuspension. To analyse the distribution, growth and effect of macrophytes within this system would be advantageous.

The 13 years of constant data collection in Cardiff Bay has the potential to provide a valuable long-term comprehensive data set to monitor changes in climate and the effects they have on an artificial ecosystem such as this, which would also be extremely useful when forming future management initiatives. This could be expanded to include an increased number of water column temperature readings to analyse water column stability on a smaller scale. This longer-term timescale would allow the assessment of natural variability and its consequences on such a unique system.

In summary, at present the phytoplankton biomass of Cardiff Bay is currently being controlled by a combination of natural (flushing rate) and artificial (aeration) methods. However, with changes in climate predicted for the future, it is highly likely that management methods will be required to adapt accordingly. This may require a shift towards more artificial (increased aeration and additional nutrient reduction) and alternative natural (zebra mussel filtration) management methods.

7.10 Final Conclusions

This study has demonstrated that Cardiff Bay is a highly artificial freshwater system; with the presence of the aeration system, controlled outflow through the barrage, and previous phosphorus reduction. Despite the phytoplankton population of the lake not being nutrient limited, the clear water system is maintained through the combination of a fast lake flushing rate, an aeration system mixing the water column, and filtration by zebra mussels.

The clear water system is at risk from changes in climate, particularly during the summer months when temperatures are predicted to increase, and precipitation is predicted to decrease. The avoidance of HABs is reliant on the flushing rate of the lake, a hydrological control that may be significantly reduced as a result of a decrease in precipitation.

Cardiff Bay is a unique site that acts more like a river system than a lake. Its' polymictic nature prevents the lake from stratifying, and its low phytoplankton biomass is controlled by fast flushing rates and top-down control of grazing by the invasive zebra mussels. These lake characteristics are not found elsewhere in the UK, making the lakes' response to changes in climate and how management is able to deal with these changes an important case study to observe and continue to learn from in the future.

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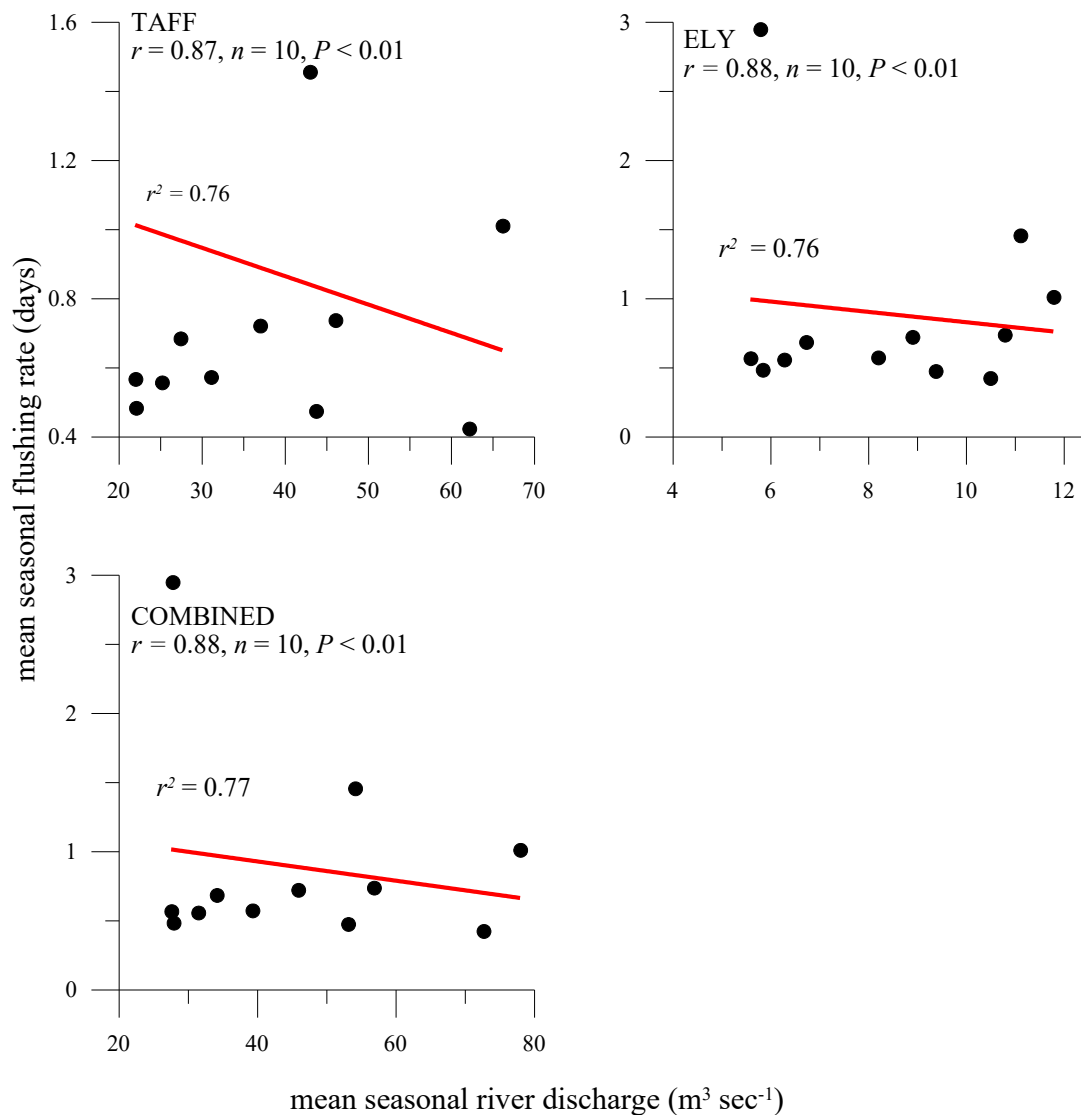
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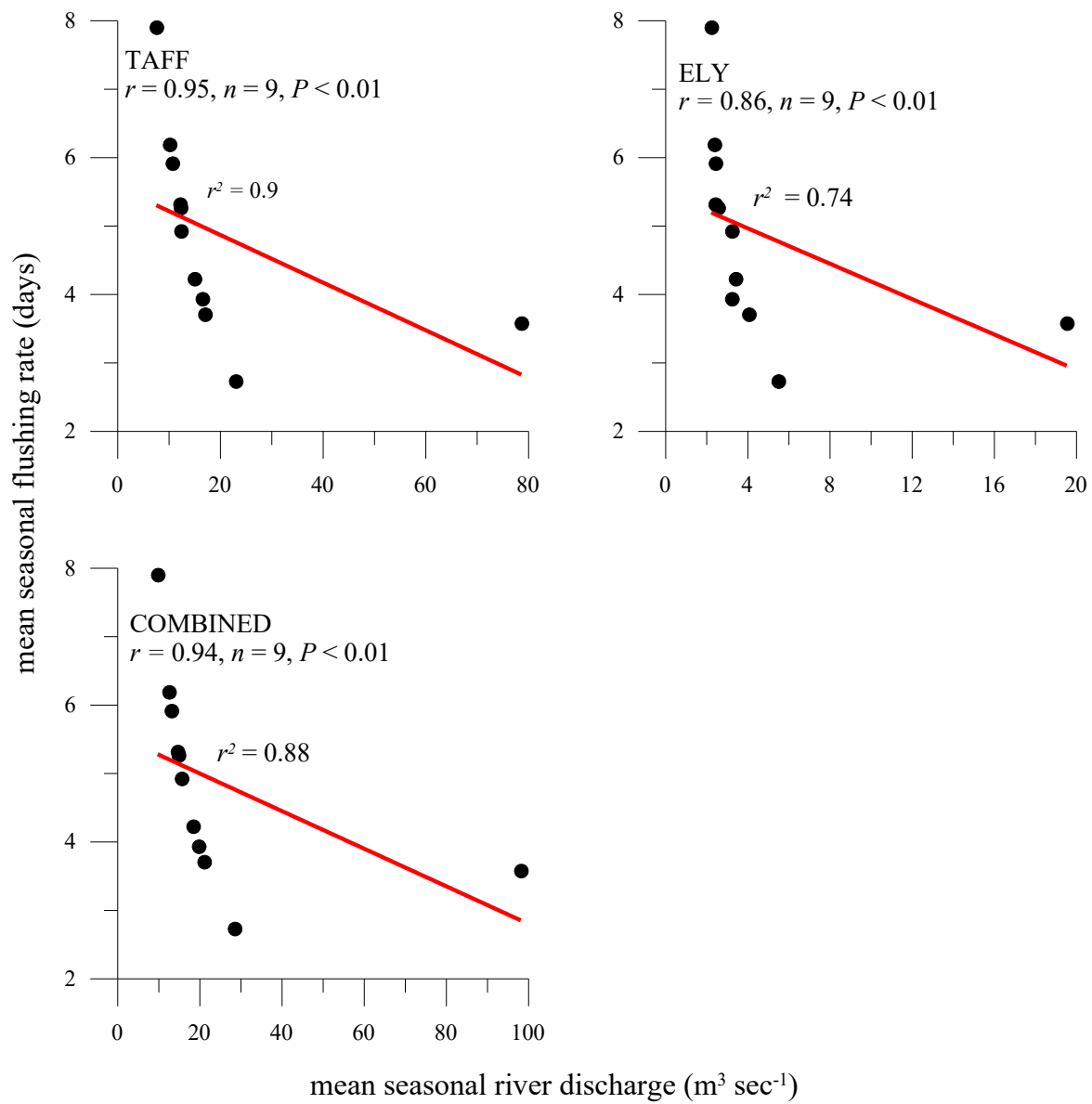
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APPENDIX

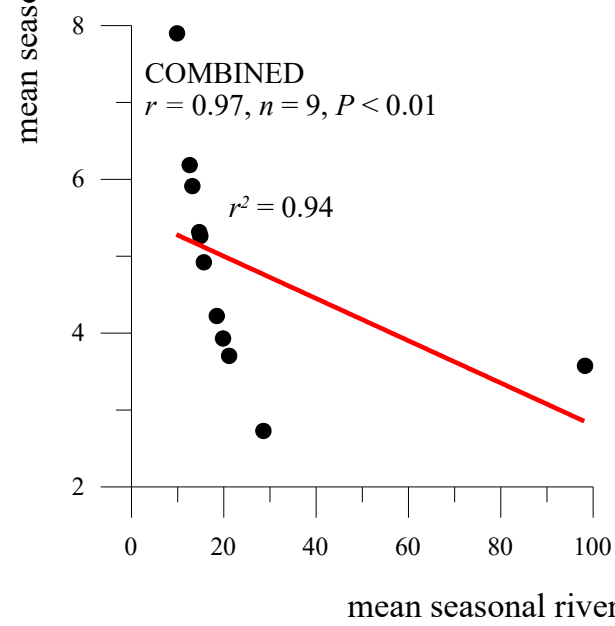
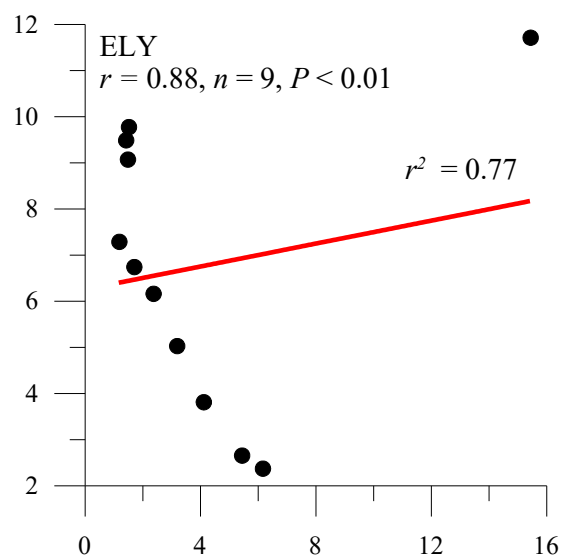
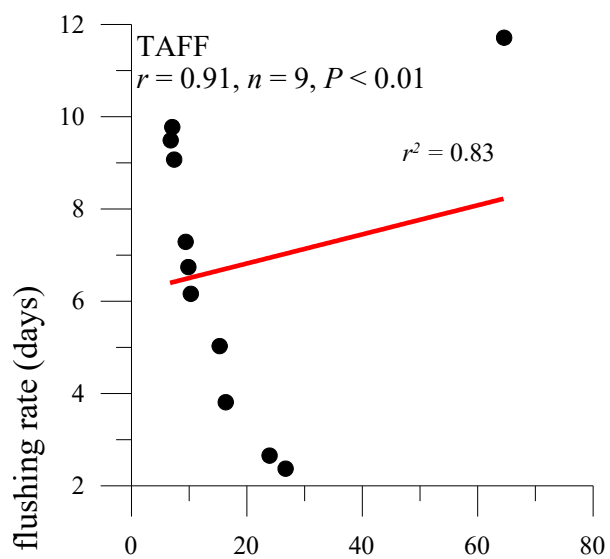
This appendix contains examples of weak relationships discovered in Cardiff Bay between 2003 and 2014. Due to the large number of weak relationships found, only a sample, and not the full number of figures are contained within this appendix.



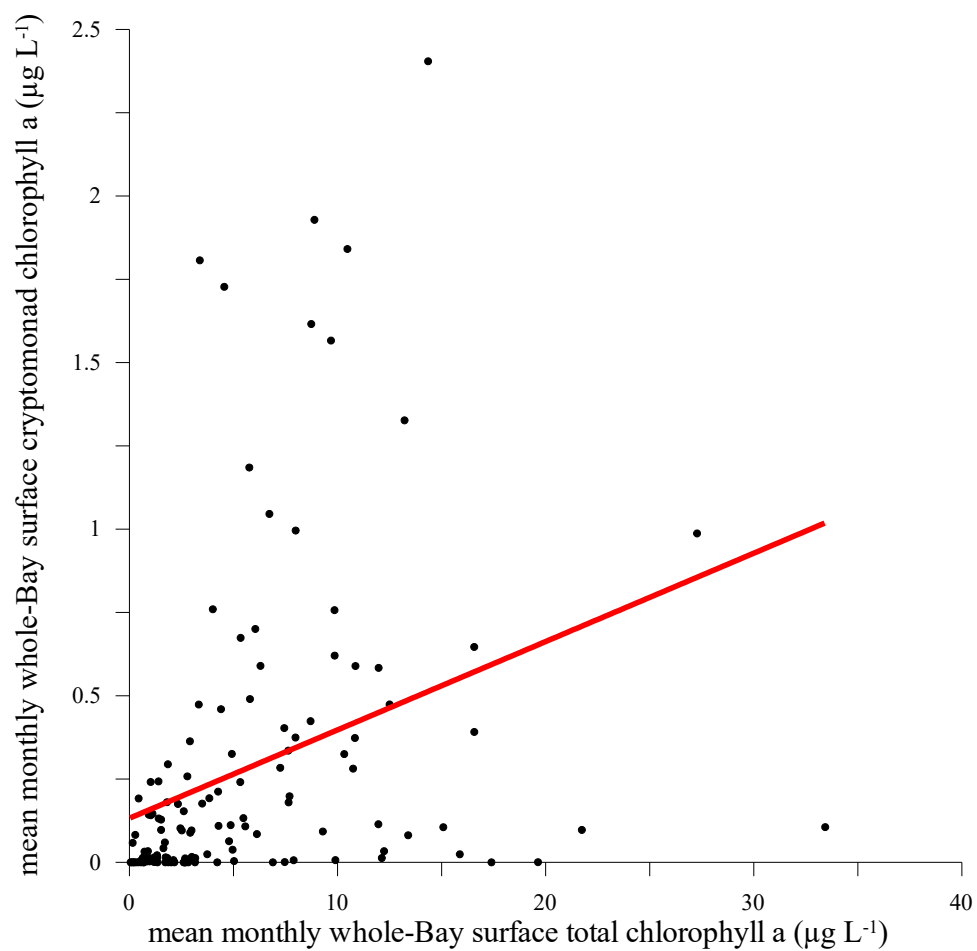
Appendix Figure 3.1: Mean winter River Taff, Ely and combined discharge and mean winter lake flushing rate. Data from 2003 until 2014.



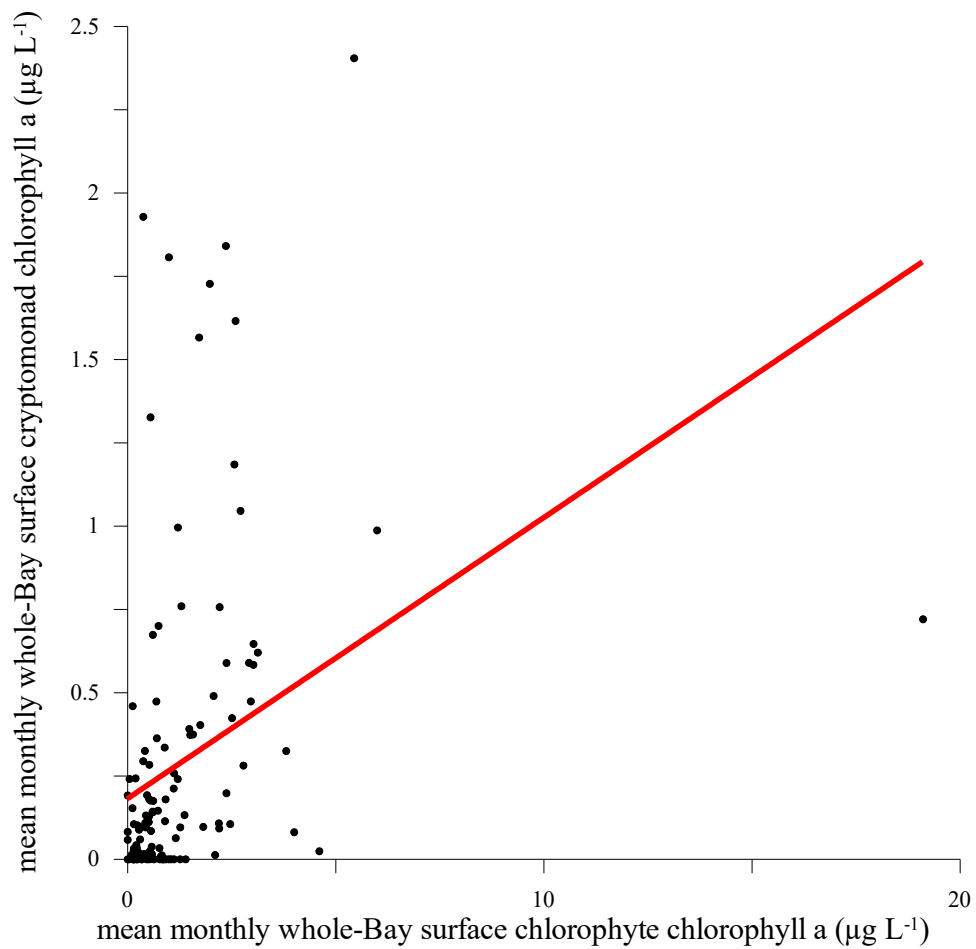
Appendix Figure 3.2: Mean spring River Taff, Ely and combined discharge and mean spring lake flushing rate. Data from 2003 until 2013.



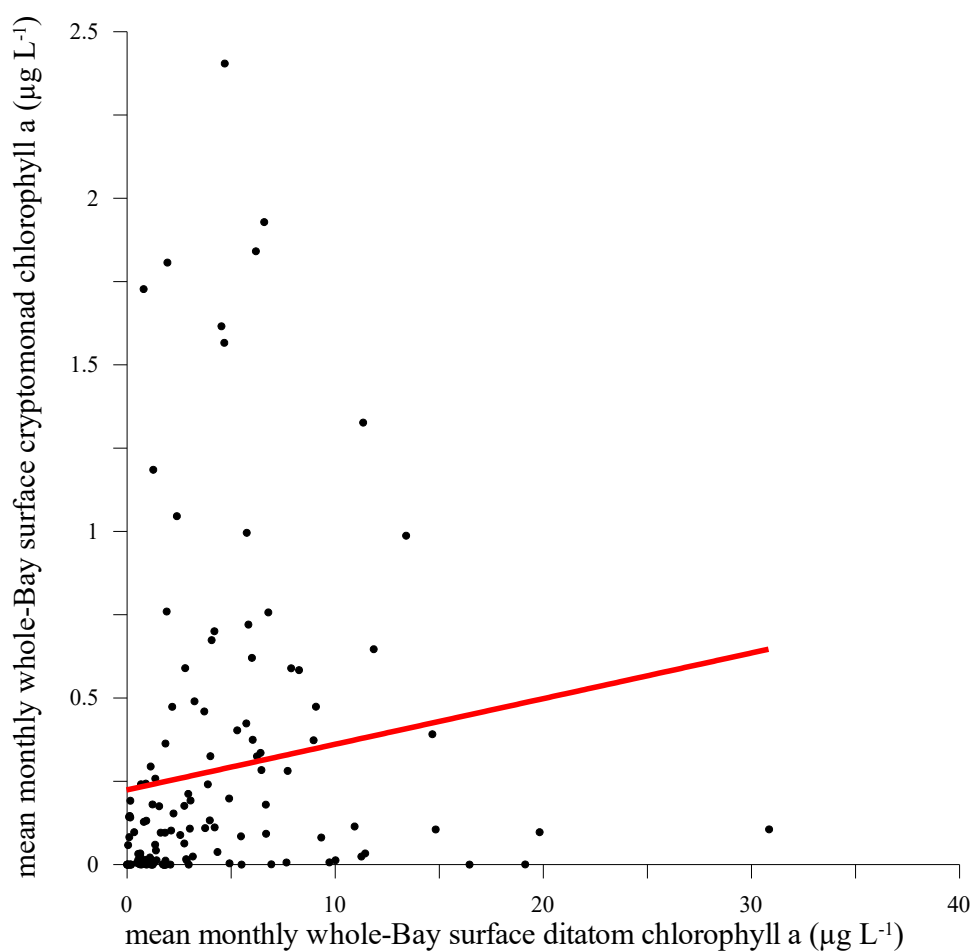
Appendix Figure 3.2: Mean summer River Taff, Ely and combined discharge and mean summer lake flushing rate. Data from 2003 until 2013.



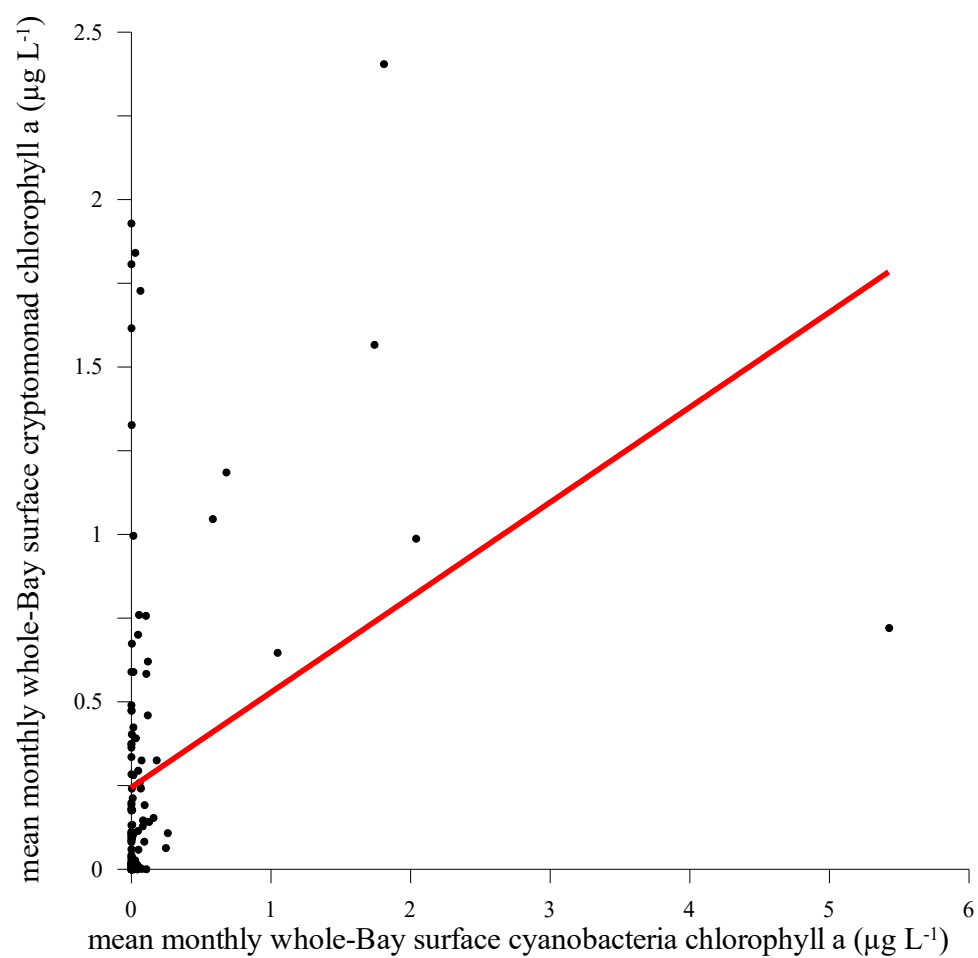
Appendix Figure 4.1: Mean monthly whole-Bay surface total chlorophyll *a* and mean monthly whole-Bay surface cryptomonad chlorophyll *a* in Cardiff Bay. Data from January 2003 – April 2014.



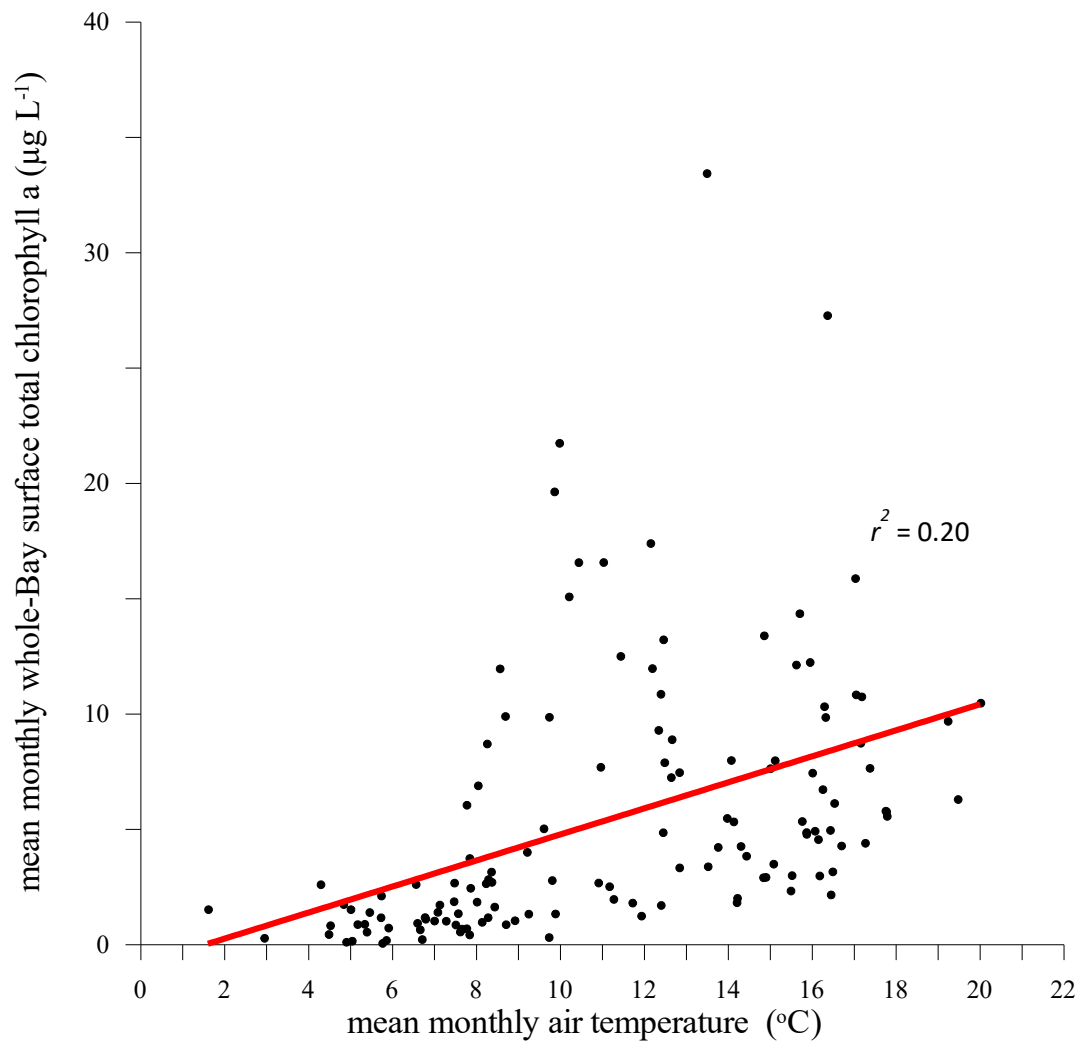
Appendix Figure 4.2: Mean monthly whole-Bay surface diatom chlorophyll *a* and mean monthly whole-Bay surface cryptomonad chlorophyll *a* in Cardiff Bay. Data from January 2003 – April 2014.



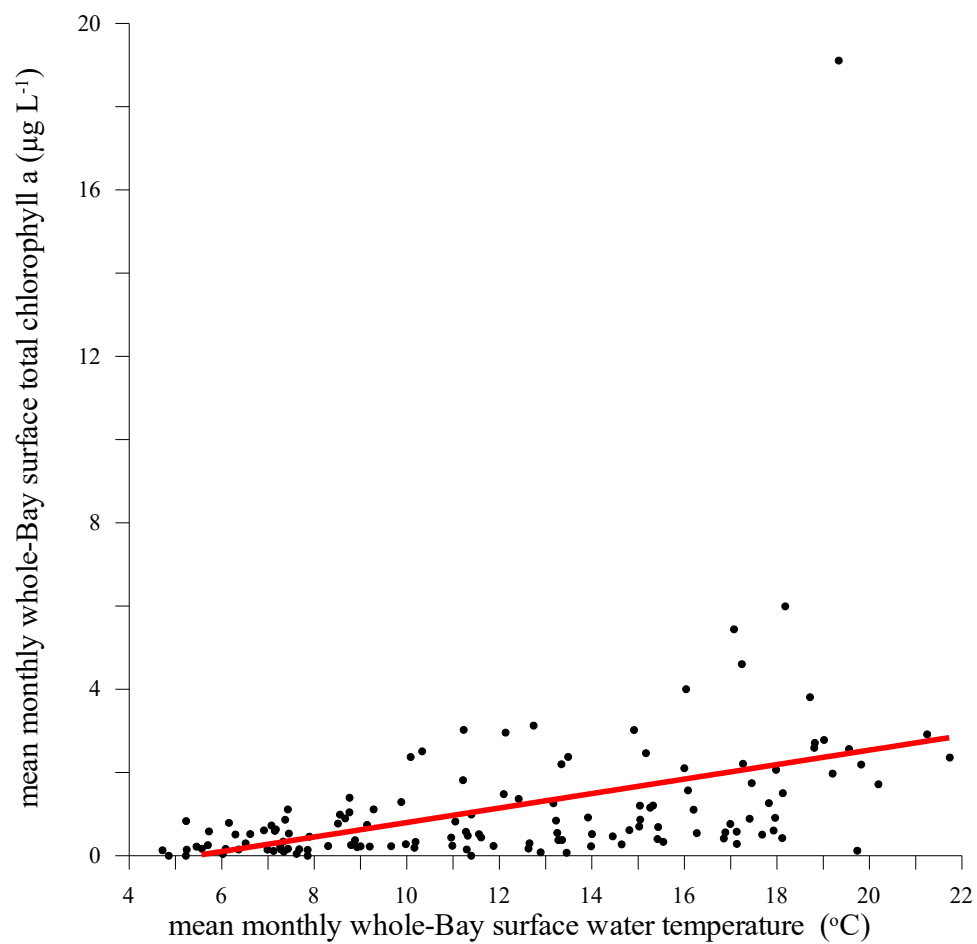
Appendix Figure 4.3: Mean monthly whole-Bay surface chlorophyte chlorophyll *a* and mean monthly whole-Bay surface cryptomonad chlorophyll *a* in Cardiff Bay. Data from January 2003 – April 2014.



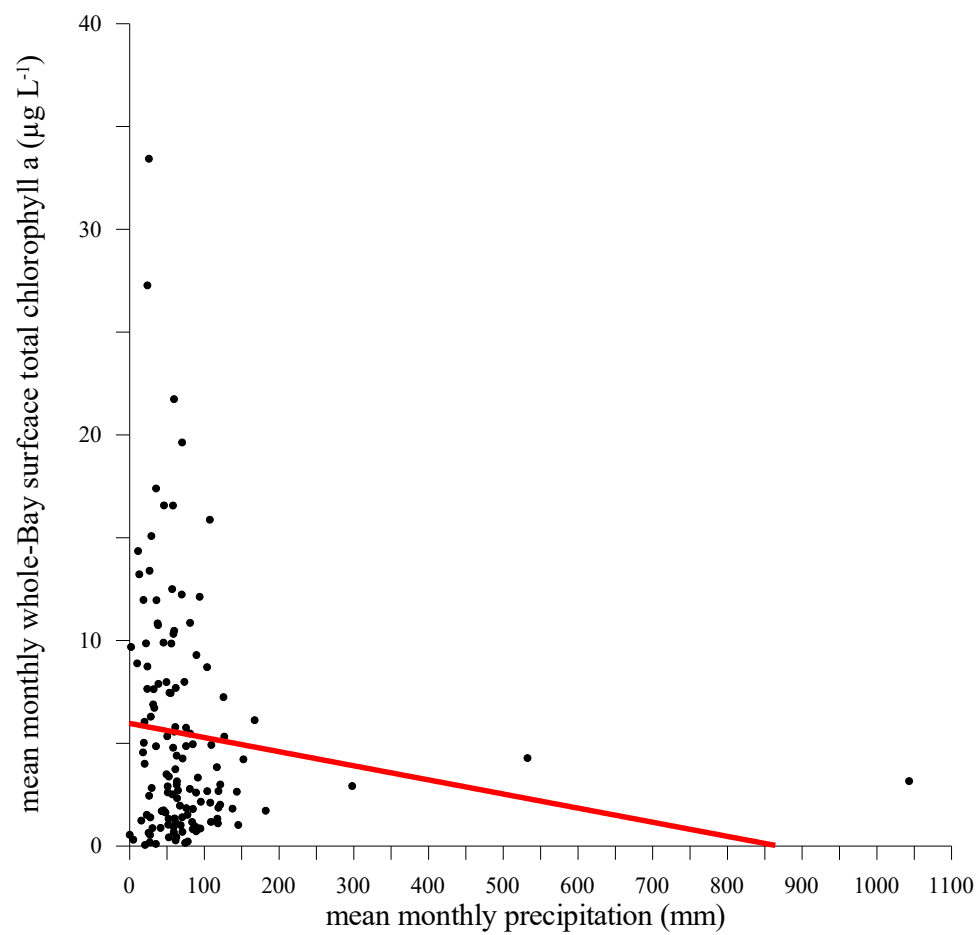
Appendix Figure 4.4: Mean monthly whole-Bay surface cyanobacteria chlorophyll *a* and mean monthly whole-Bay surface cryptomonad chlorophyll *a* in Cardiff Bay. Data from January 2003 – April 2014.



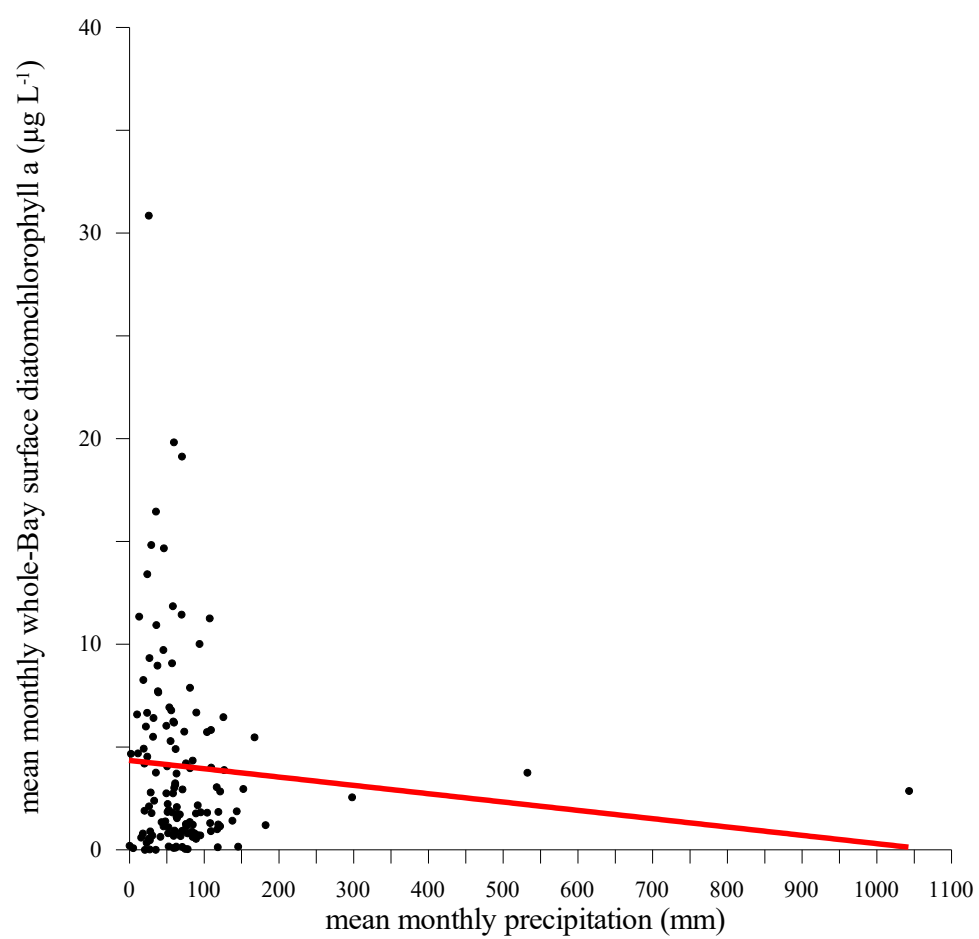
Appendix Figure 4.5: Mean monthly air temperature and mean monthly whole-Bay surface total chlorophyll *a* in Cardiff Bay. Data from January 2004 until April 2014.



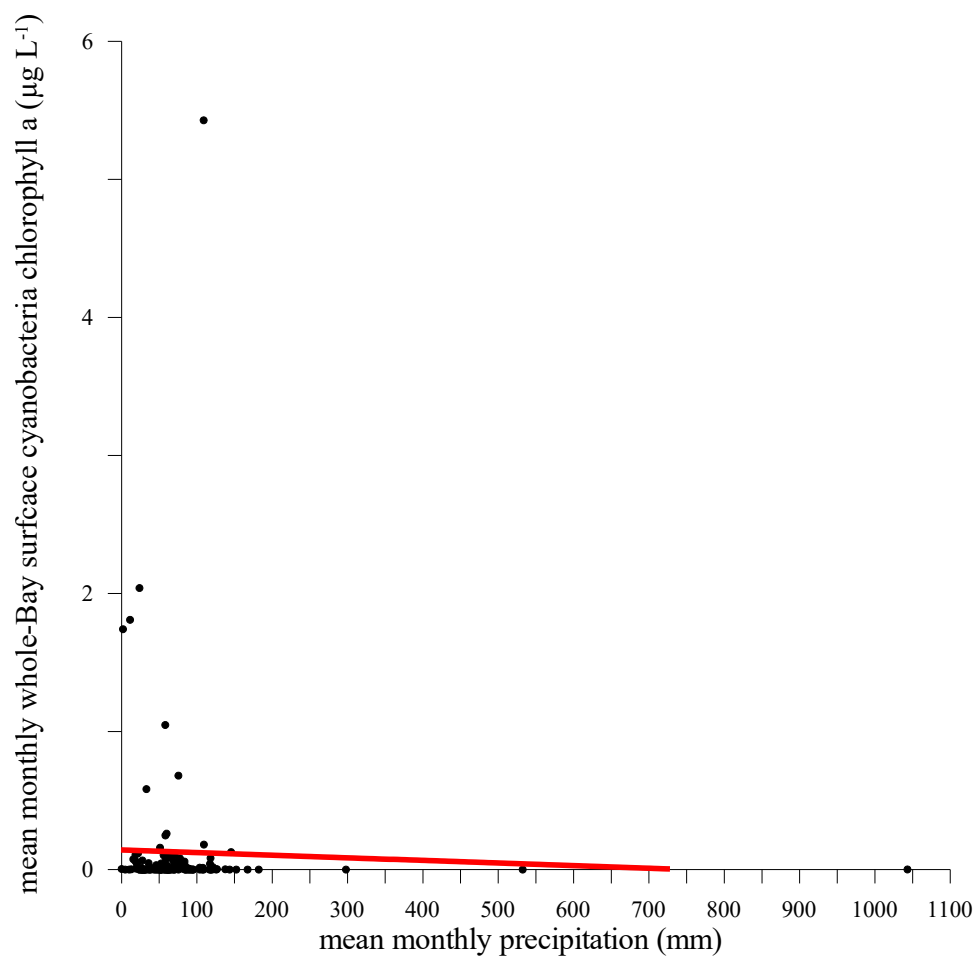
Appendix Figure 4.6: Mean monthly whole-Bay surface water temperature and mean monthly whole-Bay surface total chlorophyll *a* in Cardiff Bay. Data from January 2004 until April 2014.



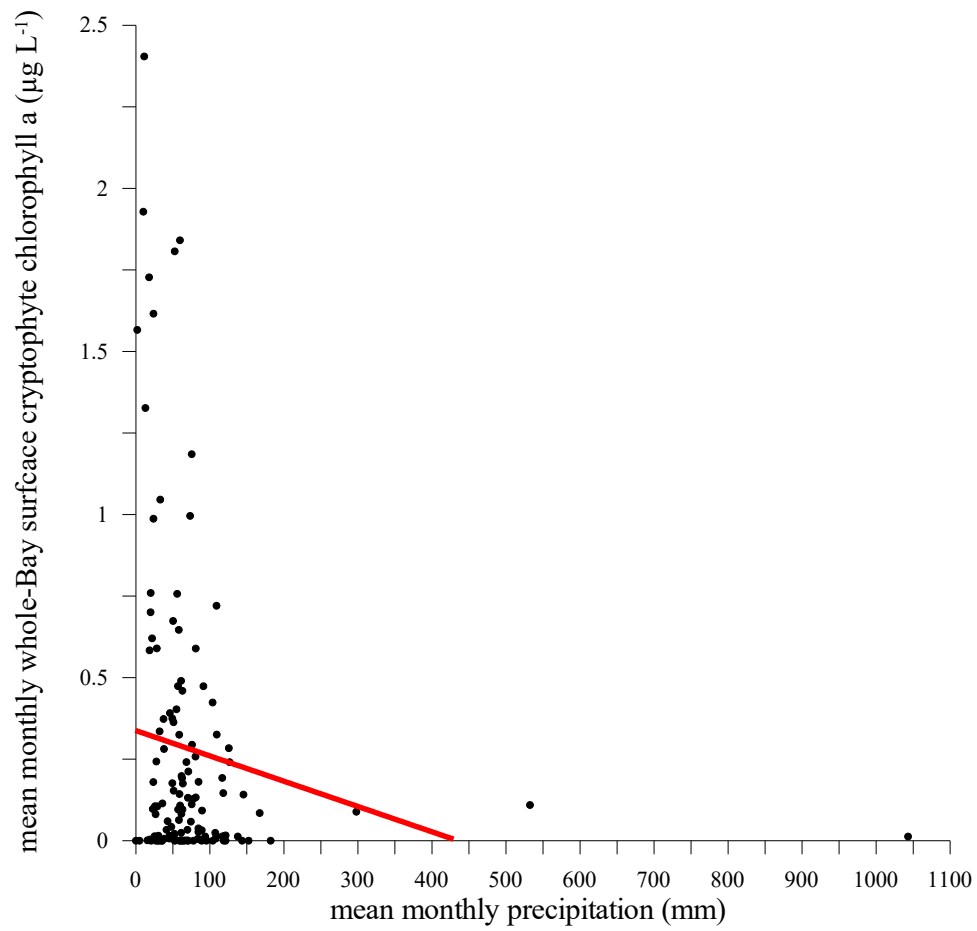
Appendix Figure 4.7: Mean monthly precipitation and mean monthly whole-Bay surface total chlorophyll *a* in Cardiff Bay. Data from January 2003 until April 2014.



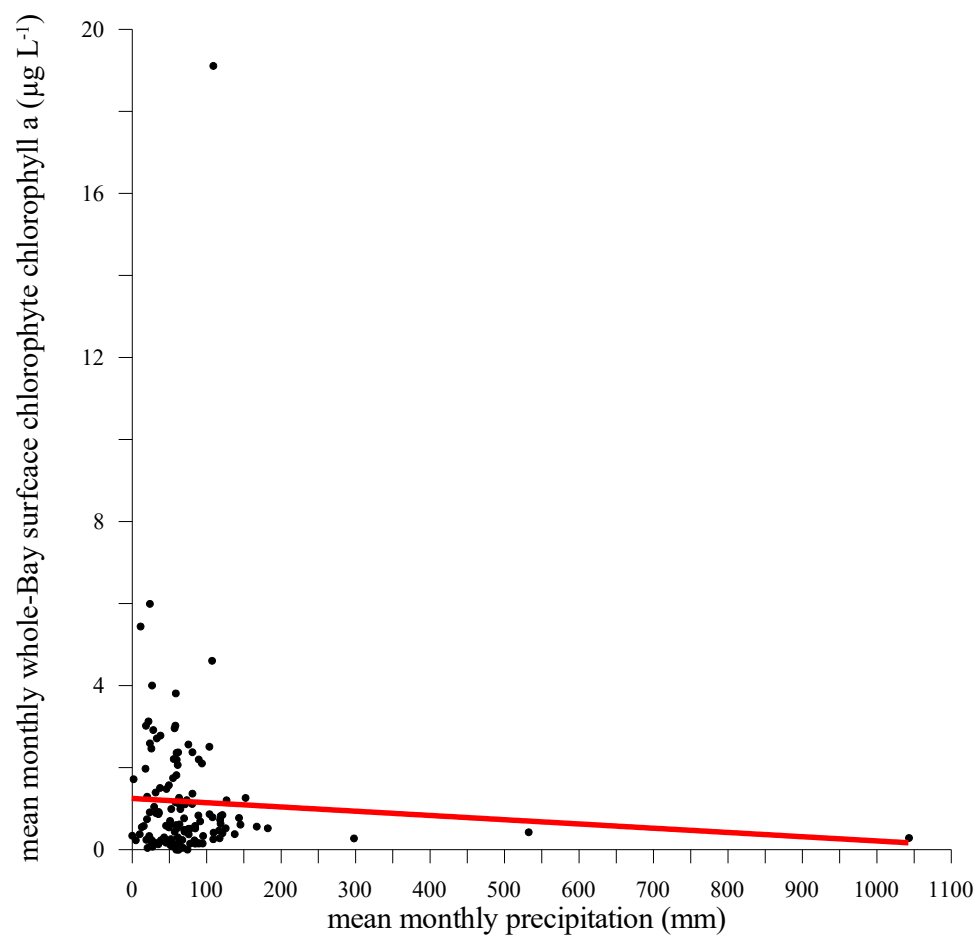
Appendix Figure 4.8: Mean monthly precipitation and mean monthly whole-Bay surface diatom chlorophyll *a* in Cardiff Bay. Data from January 2003 until April 2014.



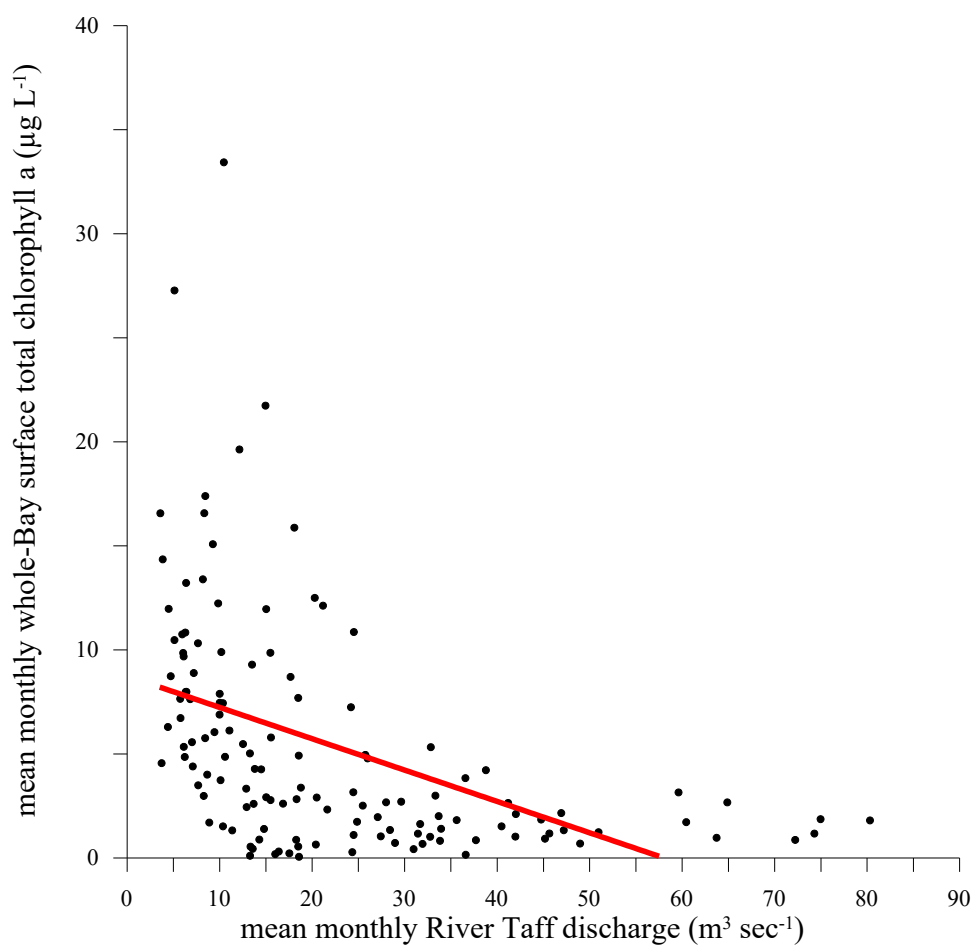
Appendix Figure 4.9: Mean monthly precipitation and mean monthly whole-Bay surface cyanobacteria chlorophyll *a* in Cardiff Bay. Data from January 2003 until April 2014.



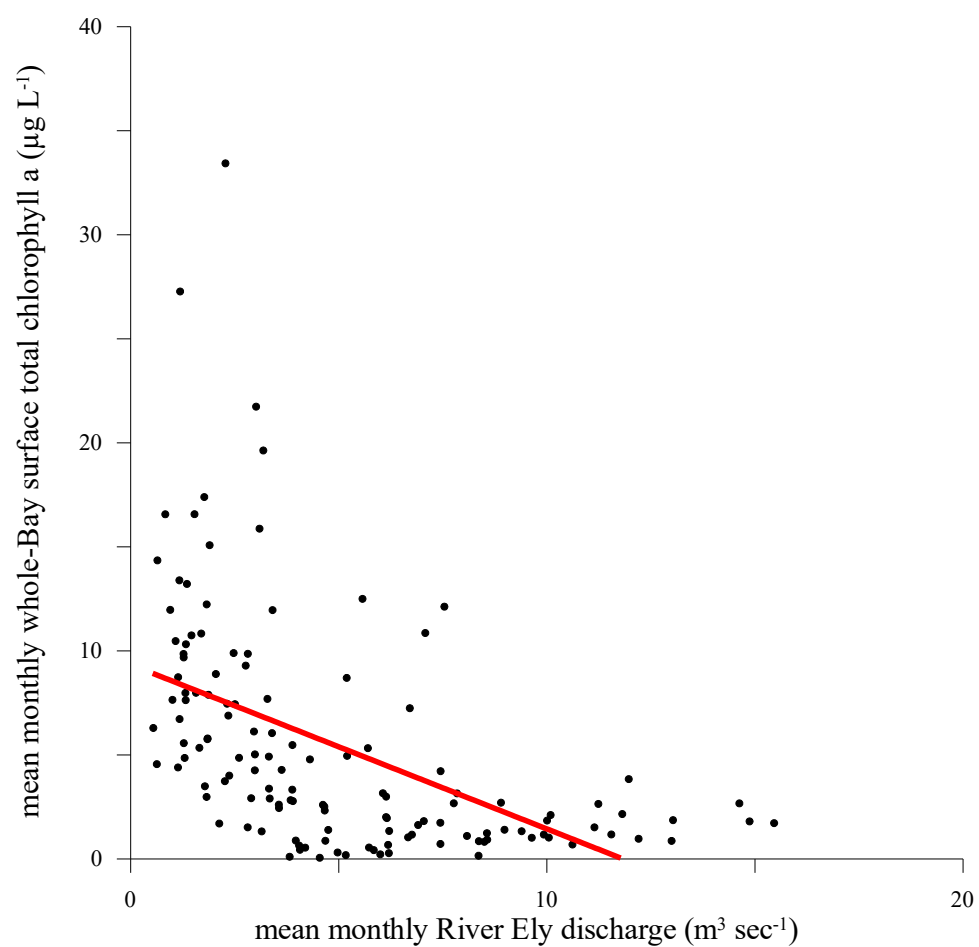
Appendix Figure 4.10: Mean monthly precipitation and mean monthly whole-Bay surface cryptophyte chlorophyll *a* in Cardiff Bay. Data from January 2003 until April 2014.



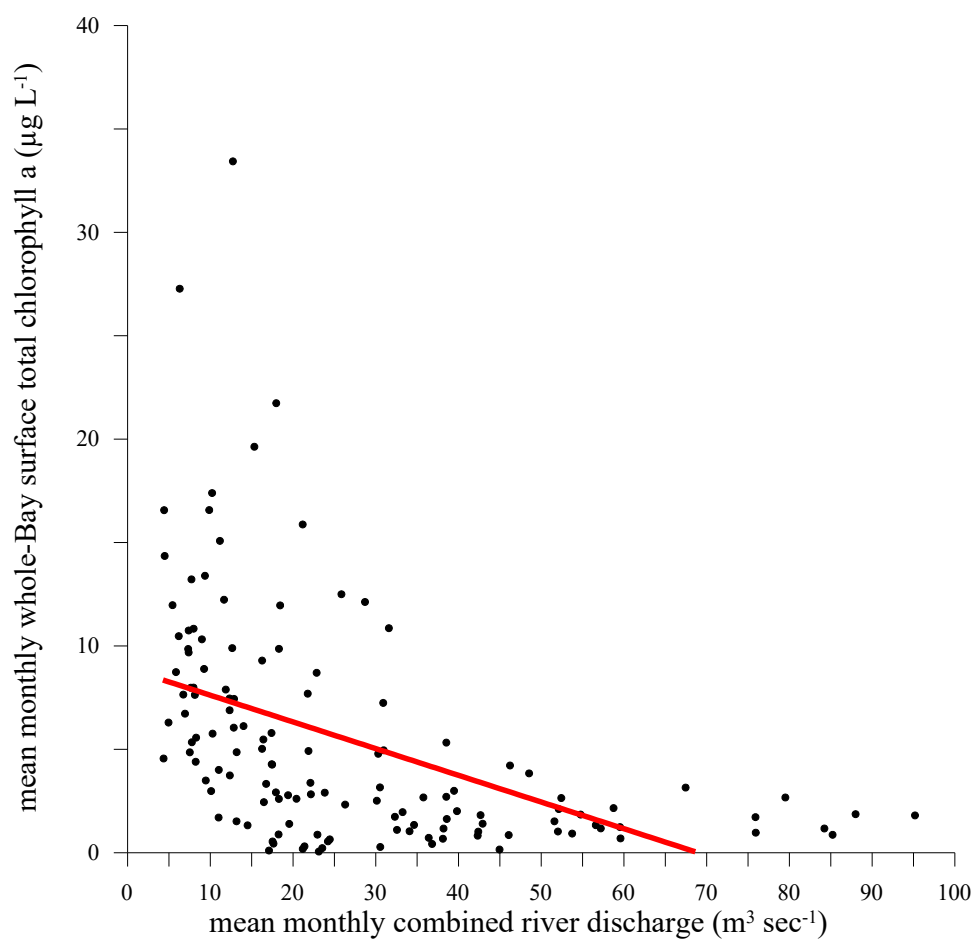
Appendix Figure 4.11: Mean monthly precipitation and mean monthly whole-Bay surface chlorophyte chlorophyll *a* in Cardiff Bay. Data from January 2003 until April 2014.



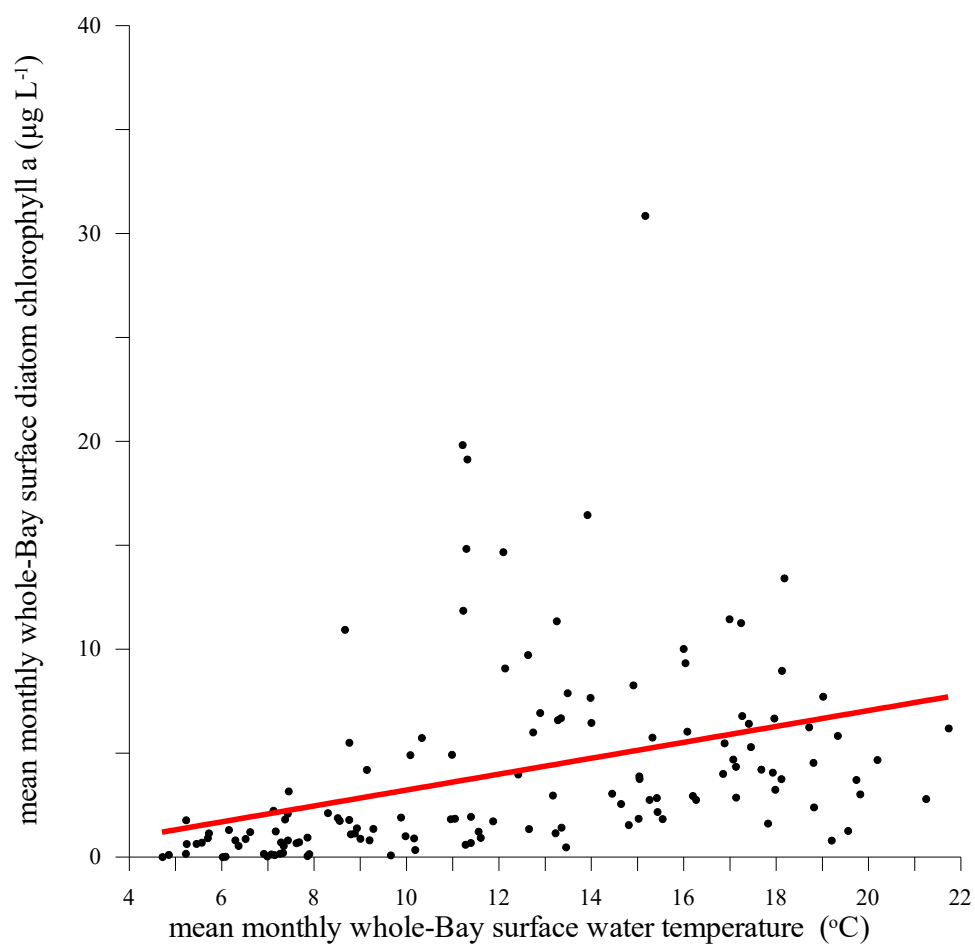
Appendix Figure 4.12: Mean monthly River Taff discharge and mean monthly whole-Bay surface total chlorophyll *a* in Cardiff Bay. Data from January 2003 until April 2014.



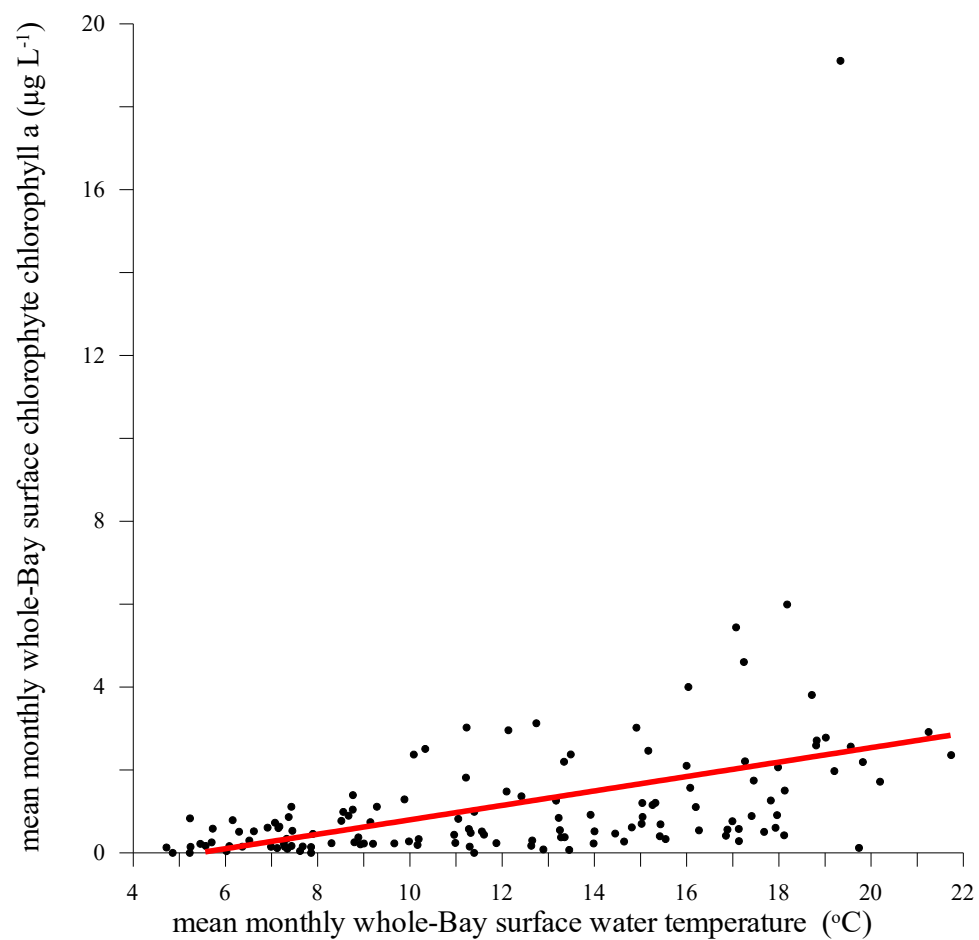
Appendix Figure 4.13: Mean monthly River Ely discharge and mean monthly whole-Bay surface total chlorophyll *a* in Cardiff Bay. Data from January 2003 until April 2014.



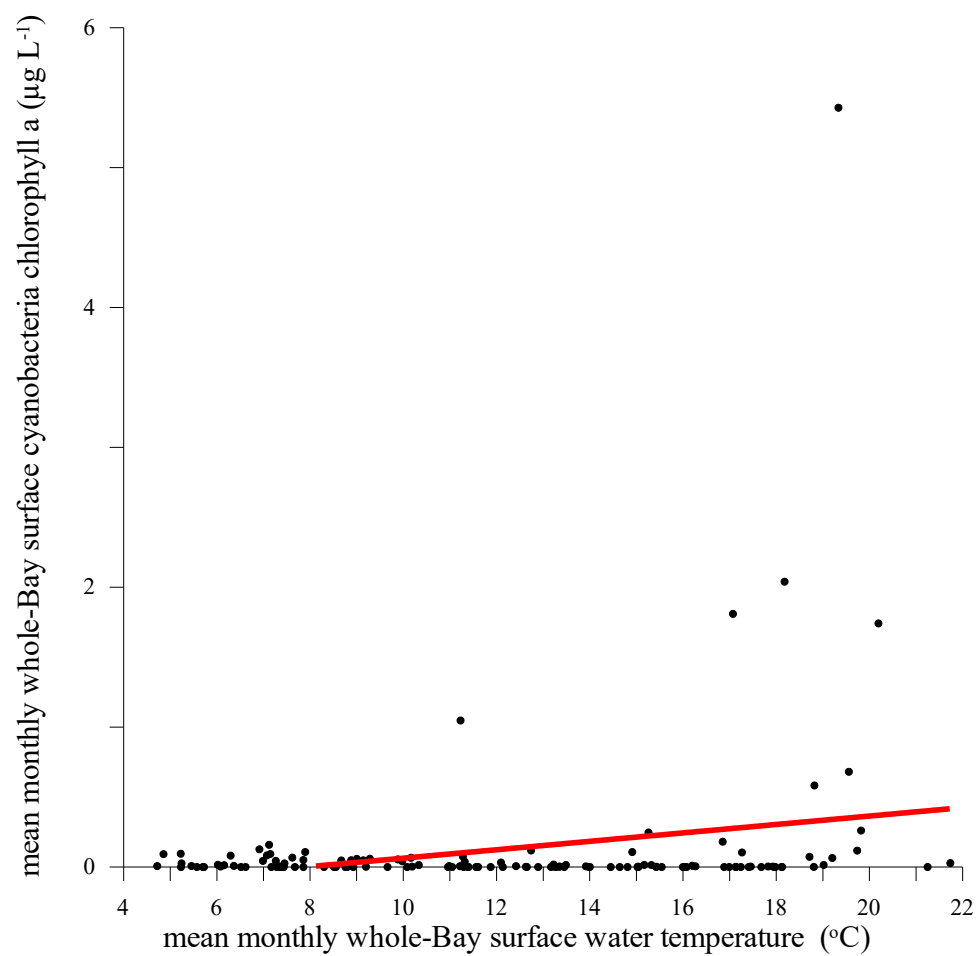
Appendix Figure 4.14: Mean monthly combined river discharge and mean monthly whole-Bay surface total chlorophyll *a* in Cardiff Bay. Data from January 2003 until April 2014.



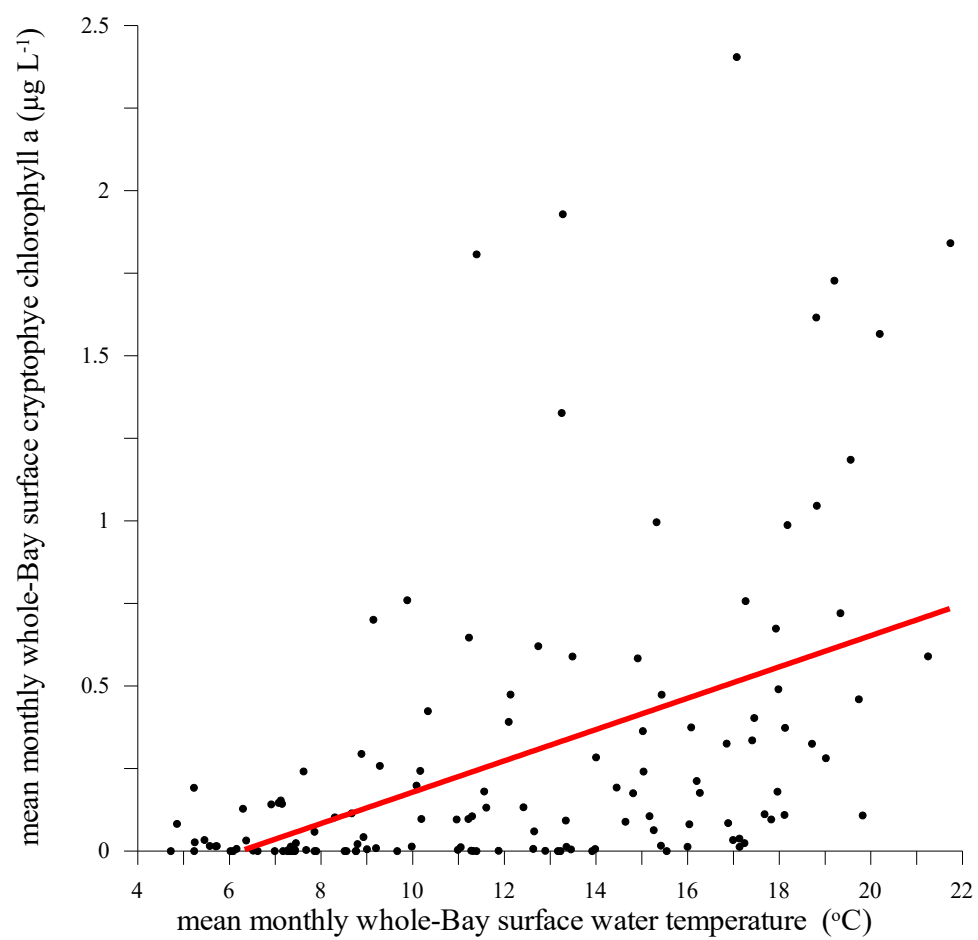
Appendix Figure 4.15: Mean monthly whole-Bay surface water temperature and mean monthly whole-Bay surface diatom chlorophyll *a* in Cardiff Bay. Data from January 2003 and April 2014.



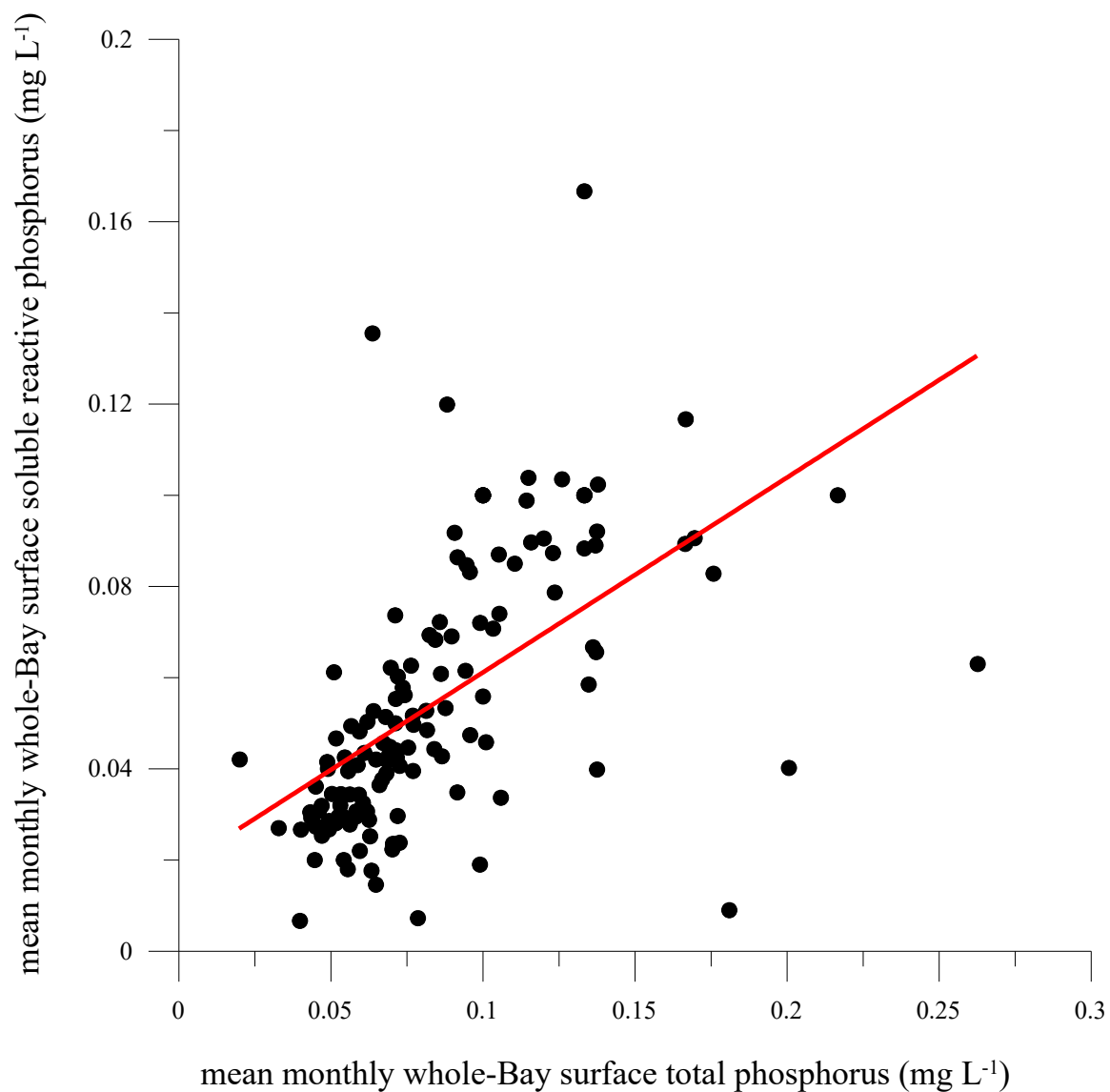
Appendix Figure 4.16: Mean monthly whole-Bay surface water temperature and mean monthly whole-Bay surface chlorophyte chlorophyll *a* in Cardiff Bay. Data from January 2003 and April 2014.



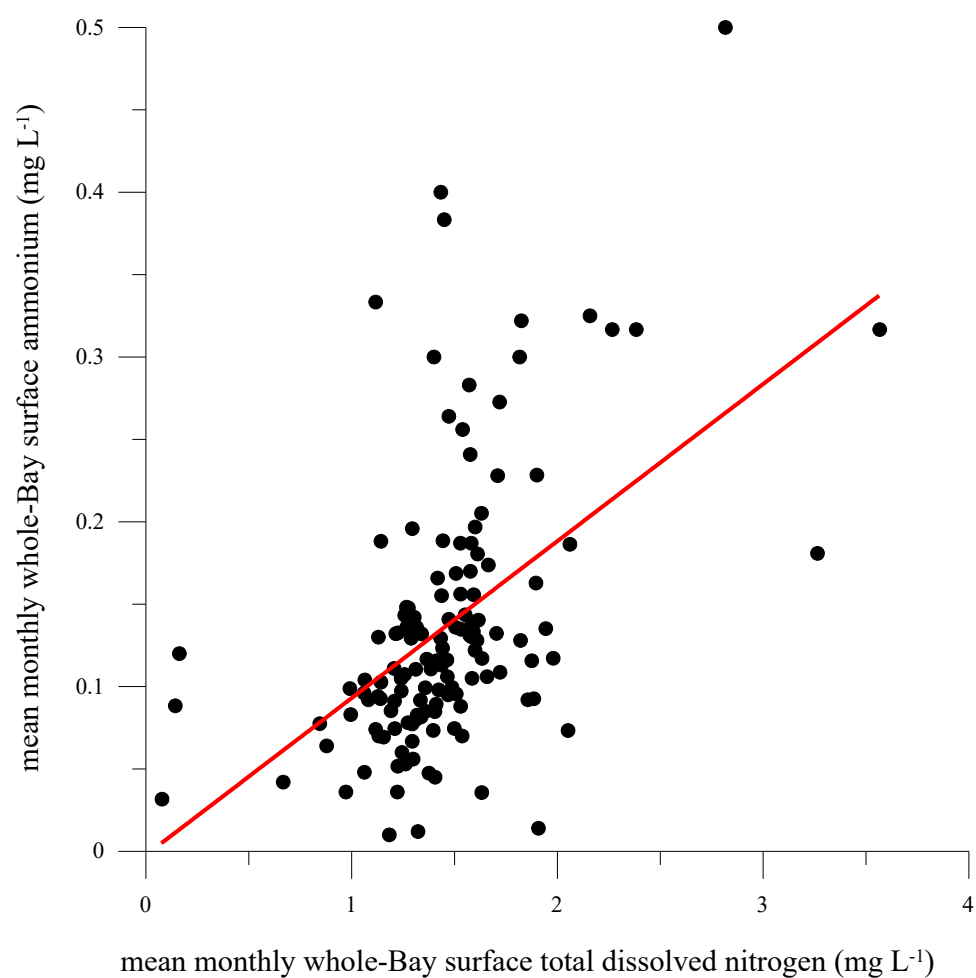
Appendix Figure 4.17: Mean monthly whole-Bay surface water temperature and mean monthly whole-Bay surface cyanobacteria chlorophyll *a* in Cardiff Bay. Data from January 2003 and April 2014.



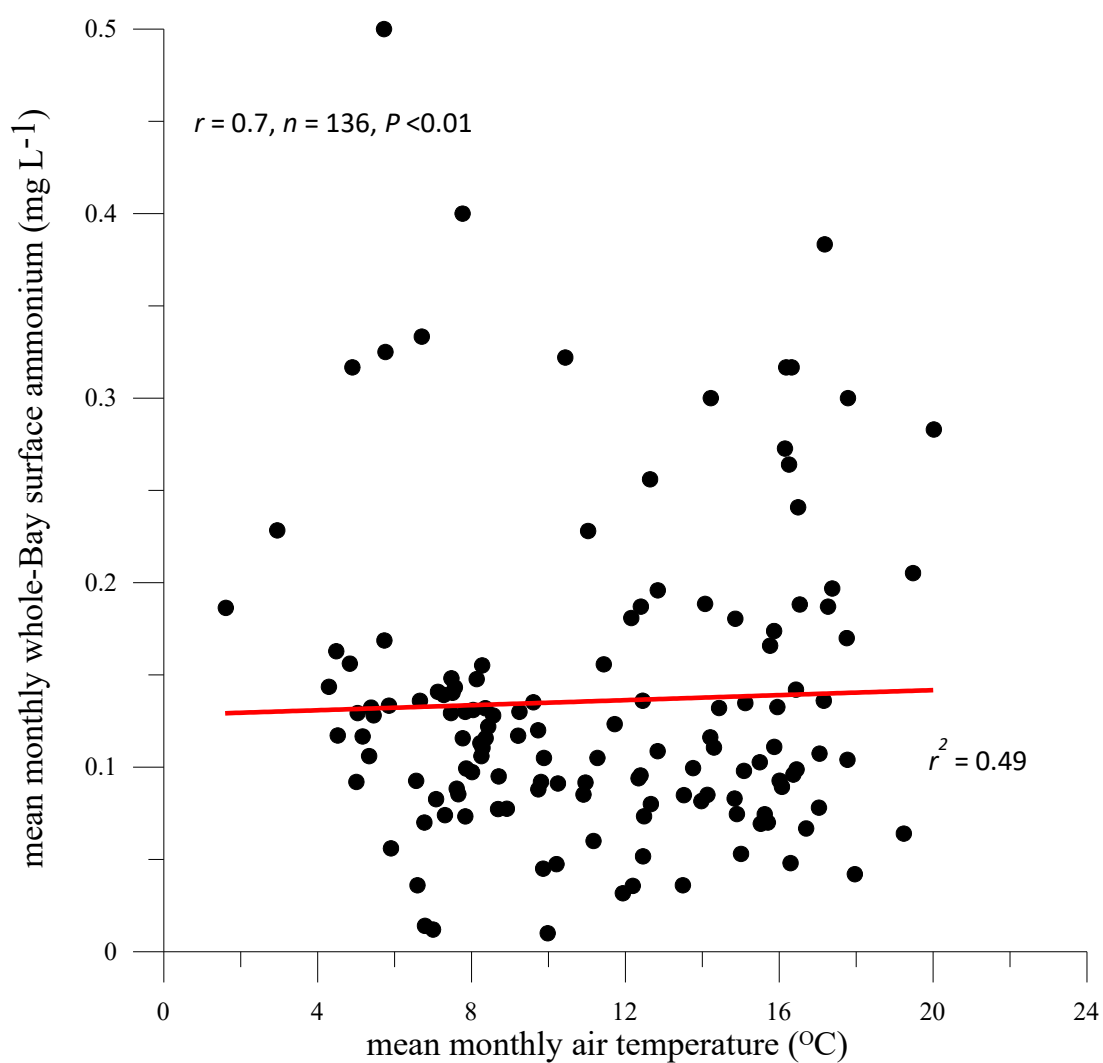
Appendix Figure 4.18: Mean monthly whole-Bay surface water temperature and mean monthly whole-Bay surface cryptophyte chlorophyll *a* in Cardiff Bay. Data from January 2003 and April 2014.



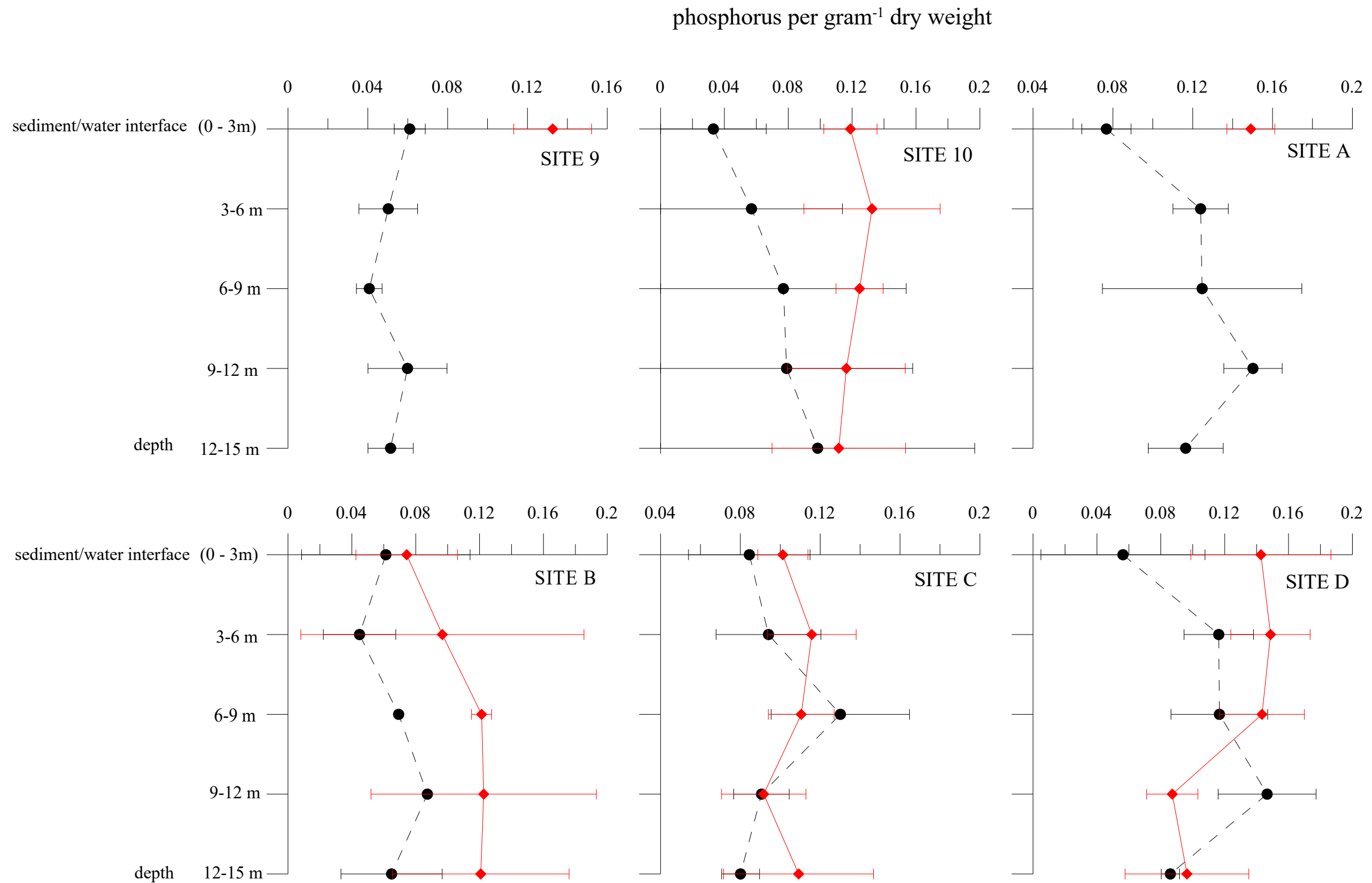
Appendix Figure 5.1: Mean monthly whole-Bay surface total phosphorus concentrations versus mean monthly whole-Bay surface soluble reactive phosphorus concentrations in Cardiff Bay, taken from January 2003 until April 2014.



Appendix Figure 5.2: Mean monthly whole-Bay surface total dissolved nitrogen concentrations versus mean monthly whole-Bay surface ammonium concentrations in Cardiff Bay, taken from January 2003 until April 2014.



Appendix Figure 5.3: Mean monthly air temperature vs mean monthly whole-Bay surface ammonium concentrations. Taken in Cardiff Bay from January 2003 until April 2014.



Appendix Figure 5.4: Ca bound phosphorus per gram⁻¹ dry weight (Pgdw) in the sediment of Cardiff Bay during summer (red bars & diamonds) and winter (black bars & circles) 2014.