Water Quality and Ecology of Heavily Modified River Irwell: The Lower River Irwell and Upper Manchester Ship Canal

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The Author

The author has a degree in Life sciences and a previous experience in academic educational fields which led to enrolment for the MSc. Degree in Pollution and Environmental Control at AL-Zawia University, Libya. Further experience in academia, industrial pollution and environmental regulation led to an increased interest in integrated aquatic biology, life sciences and environmental science/management resulted in the author registering for a PhD, the result of which is the current thesis.

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Abstract

One of the most prominent topographical features of Greater Manchester is the River Irwell which drains much of the conurbation. Arguably the most important part of its catchment in terms of the local economy and amenity is the lower Irwell which includes the upper reaches of the Manchester Ship Canal (MSC). This lower Irwell is highly modified, being subject to re-engineering to facilitate navigation and flood control. It also drains industrialised and densely populated parts of Greater Manchester and is therefore subject to organic and inorganic pollution from storm water overflows, waste water treatment works and industrial effluent. Due to the re-engineering there is a change in the hydrology from an erosional to a depositional system which acts as a 'sink' for pollutants from upstream, including suspended solids which settle out of the water column. Although a considerable amount of work has been done to improve water quality, most of the River Irwell has not yet met the standards required by the EU Water Framework Directive. The water quality and ecology of the lower Irwell was examined by interrogation of an historical data set from 2001 to 2012 plus a more intensive field and laboratory survey in 2013-14. The lower Irwell is characterised by a flashy hydrograph due to run-off from the highly urbanised catchment. It is also subject to elevated concentrations of nutrients, in particular phosphate, and a variety of contaminants from point and non-point sources that include suspended solids with a significant organic component. Sewage and other organic material increases the BOD and ammonia concentrations. Other contaminants include heavy metals from past and current industrial activity, including a legacy of past contamination of the sediments. Some improvements in water quality have occurred since 2001 although phytoplankton primary productivity in still constrained by the episodically high turbidity despite continued elevated levels of phosphorus. The benthic invertebrate community remains highly degraded, being characterised by pollution- and disturbance-tolerant taxa. It is concluded that the lower Irwell/upper MSC displays all the symptoms of the urban stream syndrome including a flashy hydrograph, elevated concentrations of nutrients and contaminants, altered channel morphology, and reduced biotic richness, with increased dominance of tolerant species. However, rivers can be a catalyst for urban regeneration and improvements in water quality and habitat diversification in the lower Irwell should be pursued as part of the wider strategy of urban renewal.

1 Introduction

1.1 Types and sources of urban fresh water pollution

Pollution is very broad term that has many different meanings; however, it can be considered as anthropogenic concept which results from human activities (Mason 2003). On the other hand, similar effects can result from natural environmental changes, such as biogeochemical cycling of elements or ecosystem processes. There is a broad definition quoted by Holdgate (1979) explaining pollution as “The introduction by man into the environment of substances or energy liable to cause hazards to human health, harm to living resources and ecological systems, damage to structure or amenity, or interference with legitimate uses of the environment". Thus, pollution is defined on the basis on anthropogenic-induced introduction of substances or energy.

The most important pollutant categories in freshwater are acids and alkalis, anions (e.g. sulphide, sulphite, cyanide), detergents, domestic sewage and farm manure, food processing wastes (including processes taking place on the farm), water soluble gases (e.g. chlorine, ammonia), metals (e.g. cadmium, lead, zinc), nutrients (especially phosphates, nitrates), oil and oil dispersants. Other factors include heat, organic toxic wastes (e.g. formaldehyde, phenols), pathogens, pesticides, polychlorinated biphenyls and radionuclides. However, there are two important categories of fresh water pollution which are particularly very important in urban systems. These two kinds of pollution consist of organic and trace metal pollution. In addition, there are many other human-induced changes that also have a significant effect on urban systems such as particular hydrocarbons in, for example, fuel oils. With the definition given above, almost anything produced by humans can be considered as pollutant at some occasions. For example, to the farmer whose land is about to be lost under a new reservoir scheme, pure water is itself a pollutant in almost every sense of the above definition. The substances essential to life such as the trace metals copper and zinc can be highly toxic in large amounts. Currently, there are about 1500 substances have been listed as pollutants of freshwater ecosystems (Mason 2002). The effect of each pollutant on an organism or community depends on the concentration of that compound and the duration pollutant exposure. As a result, a certain pollutant might bring acute or chronic effects on the target organism. Acute effects manifest themselves very quickly within hours or days, and usually results in death. Moreover, ecosystem effects of acute pollution are often irreversible, or recovery is very slow. In contrast, chronic effects develop after long exposure over weeks, months or years to low doses. The negative effects may continue for long periods of time after exposure, but they are not always fatal. A pollutant might function in different biological levels ranging from organism to community level. At organism level, sub-lethal doses of a pollutant might affect the physiological or behavioural properties of the organism. These impacted organisms might and as result exhibit reduced growth, reproduction rate or increased susceptibility to diseases in response to pollutant exposure (Farmer, 1997). At the community or ecosystem level, chronic pollution is unlikely to cause irreversible effects. However, in the case of radioactive pollution, the whole ecosystem might be subject to permanent damage. The general effect of pollution is recorded in the loss of some species, with possible gain in others tolerant of the pollutant. Generally, there is a reduction in diversity but not necessarily in the numbers of individual species that are dominant, plus a change in the balance of such process as predation, competition, and material cycling (Mason, 2002).

The nature of freshwater system, either standing (lentic) or flowing (lotic), will influence the ability of the system to retain or remove the pollutants, and hence their short and long-term impact on the ecosystem. (F. Richard Haure, 2006).

1.1.1 Organic and nutrient pollution

Organic and nutrient pollution is currently the most significant pollution problem in rives and are likely to expand globally due to increasing use of inorganic fertilizers and fossil fuels, the two dominant sources of nutrients (Robertet al., 2002). Organic pollution and nutrients also arise because of the large amounts of organic material discharged to water courses and which is considered being the main source of nutrients for the microbial community in many freshwaters (Mason, 2002). Erosion is a further source of nutrients to rivers, including anthropogenic-induced soil erosion. Deposition of sediment is also an important sources of organic pollution of freshwater courses as a result of erosion taking place close to aquatic systems (Jonsson and Carmant, 1994).

According to Sparks (2012), there are three basic sources of organic and nutrient pollution: sewage effluent, urban runoff and industrial effluent; sewage effluent can be raw or treated. This anthropogenic material commonly enters the water bodies by means of either combined sewer overflows (CSOs) or waste water treatment works (WwTWs). Run-off contributes to organic pollution by means of urban storm drains and agricultural waste from farms and their surrounding area. Industrial effluent has many kinds of origins, for example food processing plants. This type of effluent contributes to organic pollution as the final effluent enters to water courses directly without any treatment in many developing countries, or via combined sewage systems in many developed regions. This affects water quality as many food processing procedures result in very high nutrient effluent consisting of nitrate, phosphate and ammonia. These nutrients are considered to be the main cause of eutrophication.

Decomposition of organic and nutrient pollutants is a fundamental process in the cycling of essential elements through the ecosystem. When the decomposition process is slow as a result of harsh climate, natural or anthropogenic acidity this can lead to limited supply of essential nutrients, the presence of recalcitrant organic matter in soil and to the immobilization of essential nutrients. It is also recognized that high concentrations of heavy metals can have deleterious effects on organic matter decomposition and soil biological processes. In addition, oxygen is consumed in the biological degradation of organic and nutrient pollutants within aquatic ecosystems. This change in levels of dissolved oxygen can be used as an indicator for measuring water quality (Mason, 2002).

1.1.2 Trace Metal Pollution

In contrast to organic and nutrient pollution, trace metals are considered as a group of chemical elements that are present in the ecosystem in very small amounts. However, due to their involvement in many fundamental processes, they play an important role in maintaining the health of biological organisms. Trace metals include chromium, cadmium, lead copper, cobalt, selenium, and zinc. Some trace elements are essential to life and are therefore called micronutrients - these include copper and zinc. Non-essential trace elements include aluminium, arsenic, cadmium, lead, mercury and nickel. It has been shown that excessive exposure to specific non-essential trace metals is linked to some severe human disorders such as immunological, kidney, and neurological disorders (Mason, 2002; Luoma and Rainbow, 2008).

Erosion is mainly responsible for the natural contribution to the trace element content of local and regional ecosystems, including of course rivers. The dispersion of trace elements in different environmental regions is basically governed by weathering and mass transport. The extent and manner to which chemical elements of the underlying rocks are dispersed in their surroundings depends first of all on the chemical composition of the parent material, thus many trace metal pollutant concentrations vary greatly from one river catchment to another. Secondly, the pH strongly influences the absorption behaviour and retention or release of specific elements, including trace metals (Pfeifer et al., 2000). Despite the fact that solid sediments are not toxic to the indigenous biota especially at a depth of more than 5cm (Machesky et al, 2004), sediment redox and aeration are within the most important parameters that contributes to the level of trace metals (and nutrients) in the freshwater environment (Miao, DeLaune, and Jugsujinda, 2006). In general, the level of trace metals mainly is linked to point and diffuses sources in most lowland freshwater environments (Rothwell et al., 2010).

Trace metals, as a chemical component of the lithosphere, are produced from a wide range of natural and anthropogenic sources in addition to weathering. In fluvial environments, metal pollution can result both from direct atmospheric deposition, and through the discharge of agricultural, municipal, residential and industrial waste production, plus mining (Demirak *et al*, 2006).

1.2 River regulation and modification

1.2.1 Introduction

Throughout human civilization, rivers are one of the most significant natural resources that have been subject to engineering and other forms of modification. Nowadays, almost a half of freshwater fluxes are controlled by dams and impoundments (Roland et al., 2000). As early civilizations were closely associated with rivers, river engineering is one of the oldest branches of engineering. It has also been defined as the process by which a society reaches an advanced stage of social development. As a result, river regulation and flow management require collaborative efforts to control and regulate the flow of the river to maximise water resources for drinking and irrigation. Throughout history, people have constantly intervened in the natural course and behaviour of rivers to manage water resources for protection against flooding, recreation and transportation. From Roman times, rivers have additionally been used as a source of hydropower. From the late of 20th century, river engineering has raised environmental concerns broader than the immediate human benefit, and some river engineering projects have been concerned exclusively with the restoration or protection of the environment (Surian and Rinaldi, 2003)

Hydro-modification is a specific term that includes the systematic response to alterations to a river's structure. The U.S. Environmental Protection Agency (EPA) has defined hydro-modification as the "alteration of the hydrologic characteristics of coastal and non-coastal waters, which in turn could cause degradation of water resources''. All of these modifications can have many unintended consequences. Around the world, there has therefore been an increasing effort to preserve and conserve rivers and other fresh water resources to additionally maintain their social, biological and ecological functions (Acreman and Dunbar, 2004).1.2.2 Legislative framework for the regulation of river hydrology and water quality.

The most important regulation that currently impacts upon river modification in the European Union, including the UK is the Water Framework Directive (WFD). This is one of the most important rules of environmental legislation created in Europe which aims to improve water resources and protect their ecosystems at the catchment level. In addition, it is also points to reduce anthropogenic impacts on freshwater and to assist in reducing floods and conserving water to prevent droughts (Borja, 2005). There is also an increasing need to control and managing the shared water or an international water such as the Nile and Danube as transboundary rivers are a characteristic of many countries (Abu-zeid, 2008).

In accordance to the requirements of the WFD, the Scottish Parliament has approved the Controlled Activities Regulations (CAR) in June 2005. CAR is considered a risk based regulatory system that controls many activities that can affect the water environment. It is also a risk-based regulatory approach that relies on different levels of authorisation to apply according to the risk to the water environment. This enables the Scottish Environment Protection Agency (SEPA) to concentrate its regulatory efforts where the risk to the water environment is greatest, without imposing heavy regulatory burdens on activities with low environmental risk (Hanley, Wright, and Alvarez-Farizo, 2006)

1.2.3 The Water Framework Directive

The water framework directive (WFD; 2000/60/EC) is European Union legislation which was established to protect freshwaters, estuarine and coastal ecosystems. Although it is appreciated that the decision in June 2016 to withdraw from the EU will have implications for water management, including compliance with the WFD, EU standards will still apply until withdrawal, and possibly thereafter. The WFD has the specific aims of prohibiting further impairment of water quality and ecology, and promoting the preservation and maintenance of renewable sources. Reusing water is therefore highly recommended. This framework also aims to protect aquatic resources, alleviate droughts and water overflow patterns via minimising water pollution. This can be done by educational facilities to ensure the better understanding of the importance of water sources and how to contribute to protect them from the negative impact of urbanisation. Generally, the most important aim is to ensure that all waters achieve "good" status by the end of 2015. There is a requirement on the part of member states to examine the Ecological status (EcoQ) of water sources. The basic principles that contribute to EcoQ status are based on biological, physiochemical and hydro-morphological characteristics (Angel, 2005). The range of water bodies includes surface and ground inland water courses, transitional and coastal waters. The most crucial approaches are controlling water pollution, protection available sources and provide renewable resources. In addition, an economic analysis of water demand is required to achieve a complete analysing of aquatic systems, particular pollutant elements and substance must be considered principally which possibly able to have a negative impact on biological diversity, with respect to any other correlated factors such as sediments that gives a clear concept of anthropogenic on coastal and estuaries regions (A. Borja et al, 2004). Regarding to the WFD, there are five ecological classes of water bodies, High, Good, Moderate, Poor and Bad. As it has been mentioned each member states in one way or another should attain the minimal status of (Good) for water courses by the end of 2021. The most important classification reference is biological condition and measuring the degree of deviation of water body on specific biological structure and taxonomic of fish, invertebrates and macrophytes (Acreman and Ferguson, 2009).

Regulatory impact assessment (RIA) forms part of the WFD and relates to the impact of river flow modification of water quality and hydrology. This assessment will be influenced by economic consequences of projected changes and should consider safety (i.e. flood risk), scientific rationale, benefits and business issues. In addition, some river is divided to specific sections depending on flow characteristics and discharge rate. Specific categories have been added under the WFD to reflect river re-engineering; these reaches as designated as Heavily-modified Rivers because of modifications such as removal of bends. Artificial waters such as canals and reservoirs are also separately designated under the WFD (Holland, 2002). Thus sustainability of freshwater depends widely on water flows which are influenced by physical and geomorphological patterns of the catchment area. Heavy modification of water course flows thus increases the impact of changes in water level and hence the possibility of flooding, plus negative effects on the structure of the aquatic ecosystem (Gilvear et al., 2002).

1.2.4 Canalization

Waterways are one of the most important cornerstones of human civilization from the very early ages of human history. The main uses are still transportation, agriculture and domestic. More recently safety has been a consideration in river management because of flooding affects (Nienhuis, 2008). Canalization is one of many fingerprints of human impact on the global environment and is designed to facilitate the above uses. The very simple definition of canalization is the modification of waterways to follow a specified and precise routes; this is very crucial in terms of providing transportation facilities for industrial and economic growth, including creating a social network for remote and riverine communities (Rim-rukeh, 2014) .

Canalization leads to improved inland waterways which were crucial during the industrial revolution to facilitate transport of raw materials and products. Canalization affected to only the economy, but also social life due to enhanced communication of people and ideas. More recently such water bodies are recognised as contributing to a sustainable environment; they also increases the benefits of tourism and provide a beautiful background to both historical and modern cities [http://www.ronaldwaterman.com/page3/page3.html]

On the other hand, canalisation of rivers due to urbanization and to facilitate economic growth can lead to many negative impacts, including water quality and abstraction (Krielb, 2008). There are many factors that can cause deterioration in freshwater quality and canalisation is considered a major cause of water quality deterioration. It affects both surface parameters and patterns of ground water movement such as water velocity, erosion, sediment deposition and nutrient supply; it also increases flooding risk. Although canalization might in some cases however reduce flooding due to altering waterway course direction, it can affect water clarity because of increased turbidity by disrupting both erosion and sediment deposition. This also leads to reducing dissolved oxygen due to resuspension organic debris during rainfall events which by then will affect the biological fauna within the river environment (Rim-rukeh, 2014). There is often a slow degradation in river's health due to canalisation to control runoff, plus dam construction and installation of weirs. Therefore, river hydrology and morphology are increasingly considered among the most important indices of river's health (Peng, 2011)

Canalization can also influence fish populations. Freshwater fish are situated on the top food chain and are an important food resource in many parts of the world. Due to canalisation and construction of dams and locks, fish populations can be severely damaged (Bobori, Economidis, & Maurakis, 2001). However, there are some studies suggest that canalized waterways can provide refugee points that offer a protective environment for some species within a fresh water ecosystem, including fish fry that are particularly vulnerable to predation. Although canalised rivers occur all over the world and extending rapidly through the developed and now the developing world, they are still relatively rare artificial ecosystems. Some researchers consider these ecosystems very important for fish dynamics, distribution and conservation as many heavily modified rivers behave more like natural lakes (Arlinghaus, 2002).

Canalisation often facilitates recreational activities such as boating. However, there are some research pointing the downsides of water based recreation, both with respect to water and shore based activities. It was mentioned that boating, swimming and sewage residues are responsible for chemical pollution. This can affect plant and animal populations, but plants are more threatened than animals. Therefore, more research is still required to provide a better understanding and providing knowledge to control negative aspects of recreational activities to water courses (Scoroe, 1980).

1.2.5 Dams and Impoundments

The main purpose of dams was originally for irrigation, flood control and provision of drinkable water. By the first millennium BC, water was also impounded so that its subsequent controlled release could provide a source of energy through the use of waterwheels, and much later by the use of hydroelectric generators. Other purposes include the maintenance of an adequate river flow through the year for navigation, and the provision of facilities for recreation. Most modem reservoirs are multi-purpose. They impounded water during periods of higher flows that may be released gradually during periods of lower flows. In addition, impoundments provide a new body of standing water. This, for instance, can be used for fishing, boating and waste-heat dissipation from thermoelectric generating plants. Early dams were constructed using widely different approaches, for example blocking a stream with earth, and such dams are still constructed. In its simplest form, an earth-filled dam is a pile of compacted earth extending across a stream with a fairly gentle slope both upstream and downstream. Similar to earthed-fill dams are rock-fill constructions composed of quarried rock, boulders or gravel with a layer of impervious material on the upstream face. A later development in dam construction was the invention of the masonry dam which probably was first constructed in Spain. The earliest masonry dams were of the gravity type and are held in place by their own weight pressing against the foundations, and usually having a long sloping downstream toe to prevent overtopping. Man is not the only dam-building animal, the dams of beavers, although smaller than many of those made by man, can cause spectacular changes in certain areas. Streams may also become dammed temporarily or permanently by various natural or human-induced accidents such as clogging with masses of floating vegetation, including trees, obstruction by landslides or lava flows. Lake Tana in the Ethiopian Highlands, the source of the Blue Nile, is an important example of a lake formed by the damming of a stream by the flow of lava (Baxter, 1977).

There are typically two kinds of constructed dams according to the height; small dams that less than fifteen feet height, and large dams that are more than thirty feet height.

There are over 3,000 dams in Pennsylvania, and over 100,000 throughout the US. Most of these dams are small and the majority of these small dams in the US and often elsewhere are in poor condition. An increasing number of small dams have been removed in several states of the US during the last two decades to minimize safety due to failure. Although safety and liability concerns are usually the primary factor influencing decisions to remove dams, some government agencies and environmental organizations have also suggested dam removal as a method of restoring fish passage and improving the health of stream and river ecosystems. This idea is derived in part from the extensive literature documenting various effects of large dams, along with the concept that dam removal might reverse several natural and ecological consequences mentioned above (Nilsson, Reidy, Dynesius, and Revenga, 2005).

River systems can be affected by the impact of large dams, in particular which change key parameters including the flow regime and channel shape, the degree of sediment transport, water temperature and chemistry. As a result, marked changes have been recorded in populations of algae, benthic macroinvertebrates, riparian vegetation, and resident and migratory fish due to the construction of dams. The nature and magnitude of these effects are likely to depend, however, on dam size and other stream and catchment characteristics. Therefore, it is unclear whether the existing information on large dam effects is applicable to smaller dams or not. A better understanding of the effects of dams — particularly across a range of dam sizes — is needed to inform management decisions and maximize the effectiveness of river restoration projects (Roland Jansson et al., 2000).

Dams are very crucial to control flooding and to provide water reservoirs and there have been many directives and other regulations issued to manage them and reduce their negative aspects. In April 2010, for instance, the Flood and Water Management Act became law. This act applies to England and Wales and aims to create a simpler and more effective means of managing the risk of flooding as well as coastal erosion. The Act also aims to help improve the sustainability of UK water resources and protect against potential droughts. Flooding aspects, also, have been considered by the Act, and includes provisions that are related to flood control and management of the ground water risk. In addition, an article elucidates the usage of the EU directive to control the risk of flooding within the UK (McDonald, Bosshard, and Brewer, 2009).

1.3 Human impacts on water quality

Rivers are a key component of fresh water within the natural ecosystem. This component of the hydrosphere constitutes a smallest component of freshwater within the natural ecosystem but plays a very important role in the structure and function of the natural ecosystem. Flowing water is one of the key components of water (hydrological) cycle, as well providing a considerable amount of drinking water, transport, movement of nutrients according and waste removal – the latter contributing to biological productivity within the aquatic ecosystems (Allan and Flecker, 2014)

1.3.1 Organic pollution

Organic pollution is now increasing as a result of population growth with nearly two-thirds of the population living near or close to rivers. Moreover, the urban population exceeded the rural for the first time in 2007 and may exceed 60% by 2015. Such urbanisation occurs mainly in metropolitan cities which are mostly located close to rivers. Urbanisation contributes to the organic pollution issues by means of industrial activities, domestic disposal, run-off and agricultural processes along the sides of water flow or close from catchment area (Mason, 2002). Malmqvist and Rundle (2002) found that organic pollution in rivers increases the competition among biological structure of the river which in turn affects the biological productivity of polluted fresh water ecosystems. Many developed countries have a tendency to provide high quality services, including treatment of organic pollution, to maintain high welfare standards. By providing such facilities, they have also introduced new kinds of pollutants to the global ecosystem. (Pfister, Koehler, and Hellweg, 2009)

Organic pollution can affect different levels of the ecosystem. At the organism level, they are released as excessive nutrients that boost metabolism and growth of many aerobic microorganisms. As a result, there is an increasing demand for oxygen and light supply for these species to quickly expand underwater. In a long-term scale, the pollutants can dramatically alter these balances, thus heavily manipulating freshwater ecosystems. Therefore, biochemical oxygen demand (BOD) and light penetration are used as key water quality indicators. There are three major categories of organic pollution of fresh water. These include untreated or treated domestic sewage, urban run-off, industrial discharges (including food processing wastes) and disposal of organic material from farms, in particular silage and effluent from factory farming. Among these factors, human sewage and animal wastes contribute to the majority of pathogenic microorganisms to freshwaters. These pathogenic microorganisms are categorised into viruses, protozoan, helminths and bacteria. Land run-off also contains pathogenic microorganisms. The pathogenic ability of microorganism refers to specific conditions which cause harm to human health, including direct contact with pathogenic microorganism, presence of pathways to enter the human body and the pathogenic ability and nature of the microorganism (Laws, 2000). Although there are a wide variety of microorganisms associated with sewage that reach rivers, the most common microorganism in terms of biomass belong to either bacteria or fungi (Mason, 2002; Sigee, 2004).

One of the most important problems can be result from organic pollution is eutrophication. This may be either natural or due to human impact as a result of urbanization and population growth. Eutrophication is mainly defined as an over loaded of water courses by specific inorganic nutrients that are very essential for plant growth. The basic inorganic nutrients are nitrogen and phosphorous which results in an excessive primary production in aquatic ecosystems (Mason, 2002; Sigee, 2004). On the other hand, there are of dangerous compounds called 'persistent organic pollutants (POPs)'. POPs are one of the most important concerns of freshwater biologists because they are very toxic with low tendency for biodegradation. In addition, POPs are also transported easily among food chains and accumulate among different species within most or all biological tissues and, lastly, affects human health and the environment (Macgregor, Oliver, Duguid, & Ridgway, 2010).

1.3.2 The impact of trace metal pollutants

Trace metal pollutants are found in specific ecosystems in very small (commonly part per billion) quantities and some of them are very rare. However, they are often toxic to the biota even at such low concentrations. Their undesirable consequences often result from the fact that they are commonly accumulated at high concentration within the physiological structures of organisms in natural ecosystems (Markert, Friese, Markert, & Kayser, 2000). Freshwater pollution by trace metals has however only been studied properly in the last few decades. This is in part due to improvements in advance technology that makes it possible to provide more accurate data on environmental concentrations

Trace metal pollutants also contribute to the pH instability of freshwater through the release of pyrite minerals, particularly during mining, which are oxidised generating sulphuric acid; pH plays the main role in physiological regulation and ion exchange in freshwater fishes. fish and other biota, including macroinvertebrates (Law et al., 1992). Furthermore, the contribution of combined effect of both pH graduation and redox potential reflect in an increase and a decrease of specific metals especially within the sediment composition of freshwater environments as well as the increases of mobilisation and uptake of trace metals (Machesky et al, 2004).

There are many environmental problems related to trace metal pollution. Although gaseous trace metals are contributing to climate change and ozone layer destruction, most of gaseous metals returns back to aquatic systems via precipitation and then enter organisms due to bioaccumulation in which by turn negatively effects aquatic biodiversity (Domy C. Adriano, 2001).

Aquatic invertebrates are used in bio-monitoring programs as they accumulate trace metals in their tissues in many cases in approximate proportion to that in the environment, whether they are essential (e.g. copper, zinc) and hence are used in an organism's metabolism or are non-essential (e.g. cadmium, mercury) (Rainbow, 2002). Biological indicators provide a cost-effective and valuable method to determine both biological bioavailability and also indicates biological variations in the water (Phillips, 1977).

One of the most negative aspects of urbanization is the contribution of metal contamination that supplements that found in the natural environment. Mercury for instance has risen significantly in the environment in many urban areas due to release from industrial processes. It is then accumulated within organism’s tissues due to the accumulation (and sometime biomagnification) along the food chain and can in particular contaminate in individuals that rely on local fish consumption (e.g. Byrne, 1992). It has been shown that metal pollutants contribute to public health and ecological concerns. In the US, for example, two percent of women have an excessive Mg level in their blood, this contribute with neurological disorders of new-born babies as they were in childbearing age (Ue, Riscoll, Arby, and Ontesdeoca, 2014).

1.3.3 River regulation and modification

There are a number of reasons for water regulation to protect human life, facilitate commerce and improve agriculture. In addition, it may consider protecting of the natural environment as well. However, many of them negatively affect biodiversity and water quality due to excessive use of natural water resources, reservoirs to facilitate navigation, sluice barriers and river modification for recreation and flood control. Each of this increase, as a result of the expansion of urban areas, may increase discharges of toxins and nutrients. All of these changes have dramatically increased from the middle of the nineteenth century in Europe and North America and have markedly increased during the last few decades throughout the world, in particular in Asia (Lewin, 2013).

It was pointed out by Boon (1988) that there have been few studies on the effect of river modification on invertebrates in Great Britain although Brookes et al (1982) noted that nearly 35,500 km of river are canalised, and the impact of canalisation and river modification should be taken into consideration in terms of the environmental impact studies. More recent studies of the impact of impoundment of invertebrate biodiversity found I that there was a fall in invertebrate density and diversity following construction of dams. Also, there was a marked change in community structure downstream due to the resulting change in water flow, nutrients, and temperature (Horsák et al, 2008). Changes in flow regime, especially at the phytolittorial zone, led to a decrease in organic sediments and bottom fauna due to the change in the hydrodynamic system which means that sediment accumulation could provide better substratum for the biota. This heterogeneity needs to be more studied for better understanding of the adverse impact of river modification on invertebrate and fish biodiversity (Poznańska et al, 2009;) Drinkwater and Frank, 1994). Moreover, there are an obvious impact of river regulation and impoundment on thermal behaviour of the water, including changes in the degree of water column stratification. This can adversely affect the productivity and life cycle of some freshwater communities, including invertebrates, fish and plants. Water velocity in some rivers may decrease sufficiently to facilitate the development of a planktonic community. Temperature changes can be a result of “downstream flow modification, releasing stored water and so effecting stratification processes within the reservoir’’ (Webb and Walling, 1993).

Regulation and modification in slow-flowing rivers results in considerable fall of phytoplankton abundance, e.g. in the San Joaquin River in the USA from 1980 onwards, which impacts through the food chain, including with respect to large species such piscivorous fish; on the other hand some studies suggested there are a decrease of general biodiversity and biomass due to the tidal and discharge events, water diversion and dam construction which affects point and non-point nutrient sources (Jassby, 2005).

The major effect on fresh water and estuary ecosystems are due to construction of dams. Dams affect the migration of fish for spawning such as salmonids. It also contributes to reduction in the productivity of biological communities as a result of nutrient decrease carried by run-off. Moreover, water pollution may occur due to a reduction in the degree of dilution (Jager, Chandler, Lepla, and Winkle, 2008).

1.3.4 Effects of urban water pollution on human health and the economy

Just under half of global population are living in countries that encounter a shortage of freshwater resources, the majority in developing countries (Laws, 2000). Urbanisation has additionally led to water pollution due to release from industry and domestic use of water plus farming and other agricultural processes upstream. A considerable number of the human population therefore lacks access to both drinkable water and proper sanitation. These deficiencies are mainly responsible for many polluted water-linked diseases that are account for 3.2 million deaths, which is nearly 6% of the total world-wide (Sperling, Augusto, and Chernicharo, 2005). Providing advanced water management technology is one of the most important issues in developing countries as many water resources are negatively affected and need to be treated properly. Therefore, there is some concern regarding water tariffs as regulators of water use and effluent treatment as well. However, an excessive use of water makes it more difficult to manage water resources because of increasing use and demand (Ramesh and Elango, 2006).

Water pollution is considered an important cause of climate change due to the contribution of specific gases that originate from polluted aquatic systems. However, the most obvious impacts on humans related to water pollution are cholera, diarrhoea typhoid, and malaria. In addition, there is a wide range of health risks related to climate change due to urbanization and fresh water resources as stated earlier (Haines, Kovats, Campbell-Lendrum, and Corvalan, 2006) Thus, there is a growing concern to control freshwater ecosystems to maintain specific quality standards at different kinds of aquatic ecosystems (Janusz Niemczynowicz, 1993).

1.3.5 Effects of urbanization on plankton

Phytoplankton is responsible for nearly a half of productivity on the Earth due to the ability of the photosynthesis system to work under changeable conditions and wide range of environmental stress. During photosynthesis processes, plankton convert both photic energy and chemical nutrients to organic molecules for use in metabolism and growth (Hugh L. MacIntyre, 2000). Phytoplankton and zooplankton are the primary component of food webs in many aquatic ecosystems, in particular those of standing (lentic) and slow-flowing lotic systems. The plankton are of course responsive to different environmental and therefore can be used as biological indicators to assess water quality regulation (Telesh, 2004).

An ecological study of Australian rivers by Lloyd et al., (2004) suggested that many ecological changes were connected to dams and other re-engineering projects in many Australian rivers since. Changes include decrease of rainfall level, increase in evaporation, due to reduced flow and extremely changeable stream flow. It has also been found that water regulation may result in presence of undesirable plankton communities such as an increase in toxic cyanobacteria – e.g. in the Barwon-Darling river in Australia. Also, genetic changes due to the historical effects of flow and the opportunity for invasion and competition between different species have also been observed as a result of change in the flow regime.

1.4 Water pollution, river regulation and modification of the River Mersey Catchment

1.4.1 Background

The River Irwell forms part of the River Mersey catchment. The catchment area of the River Mersey is nearly 5,000km^2 including the sub catchments of tributary rivers. The River Mersey is one of the biggest rivers in the United Kingdom. It has four main estuarine components: the upper estuary which is 17 km. The Inner estuary, it is large section around 20 km length. The Narrows channel is more than 30 m in depth. The outer estuary is linked to Liverpool Bay via an intertidal bank. There is an expansion of sediments in the estuary as the water enters the sea interferes with the tidal movements and which results in the fresh water taking nearly a month to reach the Irish Sea (Fox et al, 2001). The Irwell Catchment encompasses an area of nearly 800 km^2 and is highly urbanised, including the towns and cities of Rochdale, Oldham, Bolton Salford and Manchester (Figure 1). There are however agricultural areas to the north and east of the catchment; largely pasture for sheep and cattle, including upland moorland. The main rivers are in addition to the Irwell, the Roch, Croal, Medlock and Irk, all discharging, via the Irwell, into the Manchester Ship Canal. The rivers and tributaries have a combined length of nearly 400km. The River Irwell is about 63 km in length, and its source is the Irwell springs on Deerplay Moor' north of Bacup. It then passes through Rawtenstall, Ramsbottom, Bury and Radcliffe before turning south into the cities of Salford and Manchester to join the Manchester Ship Canal. The River Irwell receives significant flows of the Rivers Roch, Irk, Medlock and Croal (Irwell Rivers Trust, 2012). The industrial revolution resulted in severe pollution of the rivers of the Mersey catchment by the mid nineteenth century and the River Irwell is heavily urbanised and receives more than two third of its volume via surface run-off and via combined sewer overflows and storm drains (Sparks Jakie, 2012). The long-term daily flow between 1976 and 2012 was 16.6 m^3/s and low flows of around 4.8m^3/s at Adelphi weir have been recorded. The annual average rainfall in the catchment is 1253mm.

1.4.2 The Lower River Irwell/ Upper MSC.

The River Irwell has played a key role in Greater Manchester's transportation network from the beginning of the industrial revolution. It connects North West with the sea via the Mersey Estuary, Figure 1. The Manchester Ship Canal (MSC), opened in 1894, allowed ocean-going ships access to Manchester and Salford. The importance of transport is illustrated by the fact that before this the Duke of Bridgewater's canal was opened in 1761 and the first steam railway connected Manchester and Liverpool was constructed in1829 (Rees & White, 1993). The lower Irwell forms the boundary between Salford and Manchester. There were many changes in the nature of the river to suit navigation as a key trade route in the eighteen and nineteenth centuries. The Manchester Ship Canal is considered to be one of the most innovative industrial achievements of the Victorian era because of its design and scale – and that it drains about 3000 Km^2. Although there were some important issues were missed regarding flood management, the MSC made Salford Docks one of the most vibrant and busiest ports in Europe. Today the lower Irwell and MSC still play a key role in the control of flooding within the Greater Manchester (Heatlie, Drake, and Debski, 2007).

Storm water overflows during rainfall events are considered to be one of the most important factors which have a deleterious impact on River Irwell water quality. This is very clear because of increasing suspended solids which correlate with the concept of river's tributaries. Moreover, combined sewers contribute with the rise in suspended sediments during storms that flush urban pollutants and runoff through the river (Rees and White, 1993). According Laws (2000), the River Irwell is considered to be affected with non-point source pollution which mainly caused by urban runoff, and consists a wide variety of contaminants that are responsible for decreasing water quality such as oxygen consuming pollutants, a high suspended sediment load, pathogens and toxic substances such as a trace metals.

Another key characteristic of the lower Irwell, including the upper part of the MSC that is a continuation of the Irwell is the increasing depth, eventually reaching around 7m (Williams et al., 2010). Although much of the river is embanked, the Irwell at Adelphi Weir 5km from the MSC is still relatively shallow at around a metre or less. Below Adelphi Weir the river was deepened and it width increased in the nineteenth century order to accommodate larger vessels travelling to Manchester City Centre. This has resulted in a change from a largely erosional lotic system at and above Adelphi Weir to a depositional lentic system. The system can thus be likened to a ‘linear lake’ (White pers. com.) characterised by features typical of deep (>5) m temperate standing waters such as thermal stratification, a significant plankton community and a homogenous fine-grained sediment (Williams et al., 2010)

There are many approaches that have been used to improve water quality in the River Irwell. For example, significant remedial efforts have taken place in Salford Quays (as Salford Docks were renamed in 1989), previously known as the most polluted region since the last century. A combination of aeration processes by Helixor mixers, removing nutrients and organic matter, visual improvement via removal of debris, algal control and decreasing of bacterial contamination has resulted in marked improvements in water quality (Radway *et al*, 1988). It has been found that the water quality has improved dramatically as the quays have been separated from the MSC by impermeable bunds. The quays have since become suitable for enhancing biodiversity through habitat diversification plus the development of a recreational fishery and other recreation contact sport facilities such as boating and competitive swimming (Williams et al., 2010). However, other parts of the River Irwell including the MSC still highly polluted and of course cannot be isolated from polluting inputs. Plans are place to improve water quality such an extending further water column mixing and improvements in waste water treatment (Struthers, 1984; APEM Ltd pers. com.).

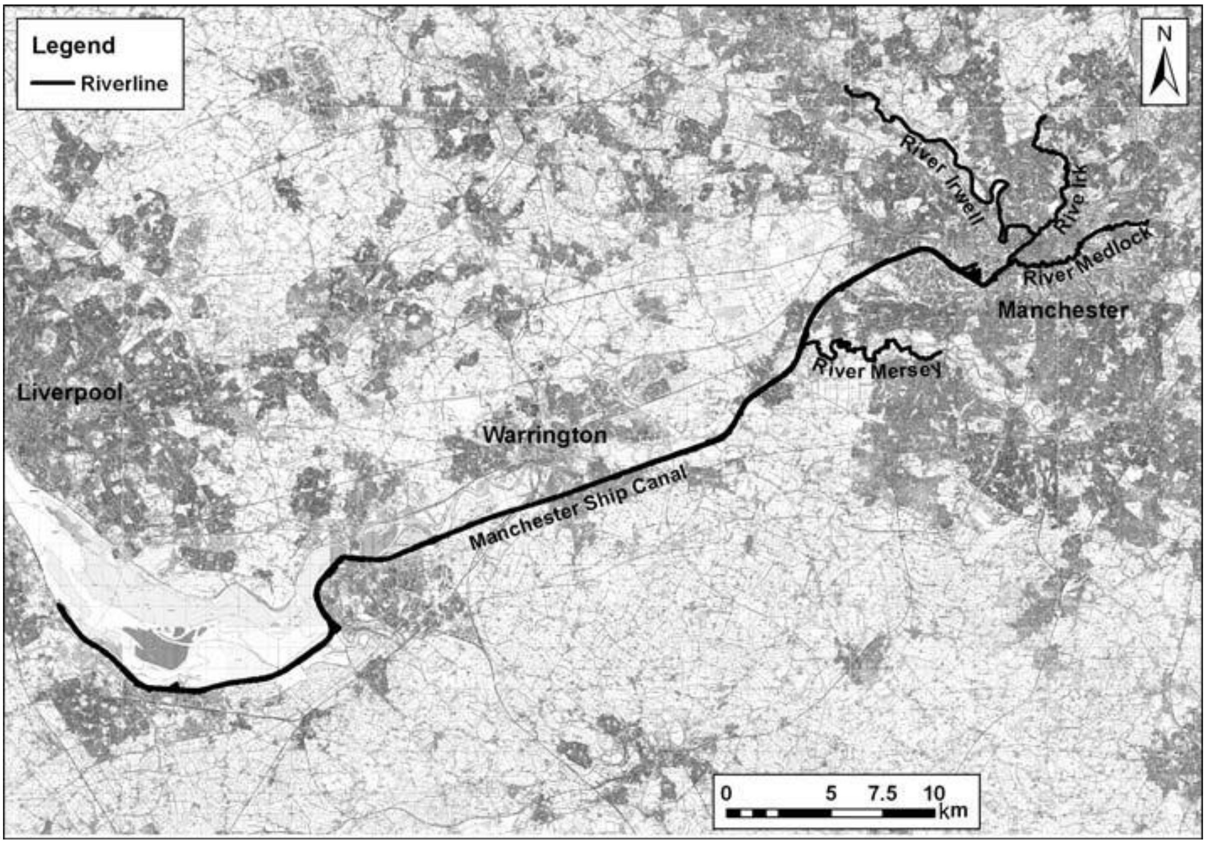

Figure 1- Manchester Ship Canal in contest with the Mersey catchment (Williams et al., 2010)

During the industrial revolution the biota was affected enormously because of freshwater pollution, including by trace metals from the cotton and other industries (Burton, 2003). Although metal pollution has more recently been recorded in the river's water at relatively low concentrations, studies suggest that metal pollution reached very high levels within some benthic macroinvertebrates and sediments in the River Irwell in the past. In addition, there were some differences in accumulation rate between species of invertebrate present in the river because of alkaline pH of the water and physiological ability of different invertebrate species to accumulate metals (Witcomb, 1983). Furthermore, the industrial legacy and contamination from shipping resulted in an increased level of sedimentation rich in trace metals, including the highly poisonous heavy metal chromium. A study carried out at Manchester Ship Canal by Byrne et al, (2016) found that the level of chromium was between 3300 and 1333,400 µg/Kg of sediment. Although chromium is very harmful to the biota, the study pointed to the lack of bioavailability and toxicity because of the low level of sediment oxygen and neutral pH at this specific site. In addition, the chromium source is mainly industrial glass particulates that are considered to be more resistance to mobility processes than other sources of trace metals.

1.4.3 Pollution Abatement schemes in the lower Irwell/MSC

Although, historically, the key aim was to minimize and prevent the negative results of flooding (Heatlie et al., 2007). Pollution control is now an additional priority as result of the requirements of the WFD (Rees & White, 1993). Water quality standards within the Irwell River and the MSC have been considered by the riparian local authorities, water companies and the regulatory bodies because of the economically advantageous impact of pollution abatement schemes and environmental control projects. This river now provides a wide variety of leisure facilities, including recreational fishing (APEM Ltd., 2007). Therefore, significant projects have recently been applied to control contamination sources such as sewage discharges and combined sewer overflows. The most significant approach has been made to improve water quality in the MSC by an oxygen injection scheme to minimize the impact of organic pollution and avoid the effect of the slow water flow by increasing dissolved oxygen levels (Williams, Waterfall, White, and Hendry, 2010). This system has been recently replaced by mechanical mixing apparatus because it is economically cheaper for maintaining and enhancing the oxygen level. Moreover, these improvements have significantly enhanced biodiversity, including fish (Sparks, 2012). Implementation of Water Framework Directive (WFD) and correlation with Strategic Environment Assessment Directive (SEA) will raise the issue of the improving the catchment environment that contributes directly to the poor water quality via run-off and drainage (Carter & Howe, 2006). During the last two decades, there has been a significant positive change of water quality due to the requirement to implement both the Water Framework and Freshwater Fish Directives. Although most of the Greater Manchester rivers, including the upper River Irwell are considered being acceptable according to the EU directives, the MSC is still only slowly recovering due to the historical pollution and canalisation resulted in a degraded r biological structure (Park, 2007).

There are many businesses and other organisations that contribute to maintain and enhance water quality in the River Irwell. United Utilities, for instance is one of the most important authorities responsible for preventing further deterioration and to reverse the decline by investing in water treatment and urban wastewater treatment facilities. They established the Water and Wastewater Service Quality Enhancements Program, and The Strategic Direction Statement (SDS). All of these initiatives relate to the UK current requirements to maintain better quality for water facilities especially in the North West. United Utilities priorities are to minimise the negative impacts of sewage and ensure a better management of water resources and the waste water networks, as well as keeping carbon emissions to a low level (Utilites, 2015).

1.5 Aims and Objectives

1.5.1 Aims

The overall aim of the study is to understand how catchment urbanisation and channel modification affects the hydrology, water quality and ecosystem processes of temperate rivers. This gives rise to the following specific aims.

- Examine the effects of urbanisation on river hydrology and chemistry over a range of timescales to reflect changes in water quality management (years) seasonality (months) and lag time (precipitation versus runoff; hours)

- Examine the impact of channel modification on river hydrology, water quality and ecology

These aims are addressed through an examination of the lower River Irwell, a heavily urbanised and re-engineered water body that forms part of the River Mersey catchment in the North West of England.

1.5.2 Objectives

1. To examine long-term changes in water quality and plankton ecology through an interrogation of archive data from the year 2000 to the present

2. To examine seasonal (monthly) changes in water quality, plankton and benthic invertebrate ecology at a higher spatial resolution and over an eighteenth-month period from March 2013 to August 2014

3. To examine lag time precipitation impacts on hydrology, water quality and plankton ecology over the short term (July September/December 2014)

Aim 1 will be addressed through meeting objectives 1-3 and aim 2 through meeting objectives 2 and 3.

A subsidiary objective is to make recommendations for further management strategies for the River Irwell to mitigate the impact of chemical pollution and canalisation on the water quality and ecology of the River Irwell and other temperate urbanised rivers.

2 Materials and Methods

2.1 Stations and field work

2.1.1 Site description

The reach of the River Irwell covered by this study for the reasons outlined in the introduction is the heavily urbanised and reengineered area immediately above Manchester city centre and the upper reaches of the MSC, Figure 2. This stretch of the River Irwell also encompasses the changes in water hydrology that are likely to occur due to a transition from an erosional to a depositional system. Therefore, the main criteria for choosing the sites were, in addition to accessibility and safety was approximate equidistance between sites and that they reflect the change in the system from erosional upstream as the depth is mostly below 1 m to depositional where the depth was around 5 m. The depth of the final site at the Turning Basin increased to around 8m.

2.1.2 Sampling sites

Within the above reach, four sites were chosen between Adelphi Weir (site 1) and Turning Basin (site 4) to address objectives 2 and 3. These sites are shown in Figure 2. Adelphi Weir was also sampled by the Environment Agency (Adelphi Weir) while APEMLtd sampled a site adjacent to my Turning Basin site (MSC APEM, Figure 4). These two sites were used to examine historical changes in water quality (Objective 1), Figure 2.

Figure 2- Site locations with respect to site 1 Adelphi Weir [A.W]. It is also the site of Environment Agency. Site 2 Mark Addy [M.A]. Site 3 Regent Road Bridge [R.R.B]. Site 4 Pomona Docks [PDs]. Site 5 Old Trafford Road Bridge [O.T.R.B]. Site 6 Turning Basin [T.B] and it is Site 10 of group sites monitored by APEM Ltd shown in see section 3.4 below (source: Google Earth).

Table 1: sampling site locations along the lower Irwell/upper MSC.

Sites	Name	Grid reference	Width(m)	Distance from site
1	Adelphi Weir	SJ 83336 99297	35	0
2	Mark Addy	SJ 82986 99363	37	1000m
3	Regent Road Bridge	SJ 82578 97677	35	2500m
4	Pomona Old Dock	SJ 81989 97128	50	3500m
5	Old Trafford Rd Bridge	SJ 81172 96586	50	4500m
6	Turning Basin	SJ 80849 97053	212	5000m

Site 1 at Adelphi Weir has a depth of <1 m and is reflective of water quality from the upper and less urbanised reaches of the River Irwell. This site is one of the main sites of historical and long-term changes that were monitored by Environment Agency. The river is embanked but not canalised. Site 2 is canalised and located adjacent to the Mark Addy public house. It has a depth of 1to 1.5 m. Site 3 is where the river's depth increases, and it is between 2.5-3m at both near and far sides, whereas the middle could have an average between 3 and 5m as it was used for Ocean going ships. Site 4 and 5 are within the Turning Basin where depth is varied between 5 to 8m and the width is also increases from over 50m to nearly 250m. Table 1 outlines the specific distance between each site which is mostly around 1000m except the distance between Adelphi Weir, Mar4k Addy and Regent Road Bridge where the distance is 1000m between each of site and Old Trafford Road Bridge and the Turning Basin were the distance is 500m, the short distance between O.T.R.B and T.B (500m) was the most significant reason to reduce this site from seasonal changes survey as most of phsio-chemical parameters were identical. Water samples were taken every two weeks and electronic measurement of water profile for temperature and dissolved oxygen, plus pH and conductivity was taken every week to address objectives 2 and 3.

2.1.3 Sampling regime

In order to provide more accurate and reliable data, I conducted a two-year survey on a fortnight basis, each of which lasted for at least a week to keep track with weather change and changes in the river's hydrology over two full seasons (objective 2). In addition, a short-term intensive survey was performed to monitor any significant change in water quality because of episodic events (objective 3). These episodic events may take place on daily basis or as soon as the change occurred following rainfall. Sampling was first conducted in March 2013 for a year. Then I decided to add another site further upstream at Adelphi Weir to address the hypothesis requirements of comparing with a more natural erosional reach of the River Irwell. This started in September 2013 and monthly survey continued until August 2014. Sporadic sampling considered the third quarter of 2014 as a summer or warm season, and the last three months of the year is as a cold season.

2.2 Physical-chemical sampling and Laboratory analytical procedures.

All sites established for this study were used to collect water samples twice a month to obtain sufficient data which was then used to analyse water parameters in terms of effects of seasonality. There are three important generic parameters that affect water quality and hence are used in this study. These are physical monitoring, water chemistry (including trace metal analysis) and biological monitoring. This includes water profiles which were monitored every week to assess the effect of seasonal-induced climate change and discharge on water quality, specifically the potential for water stratification.

2.2.1 Physical monitoring

Most of physical parameters were measured on-site. As the depth varies between three and one meter and a half and the accessible sites (depth in mid channel was greater but this of course could not be examined), three measurements are taken to find out if stratification of the water column occurred. An electronic field meter (YSi 556 Multi probe system YSI, Yellow Springs, Ohio, USA) was used to measure temperature, dissolved oxygen (DO), conductivity and pH. The YSI meter was subject to continuous calibration to ensure accurate measurements. The calibration was taken place for conductivity and pH every month or after four sampling events by pre-prepared solutions, whereas DO was calibrated before each sampling by distilled water to avoid any unreasonable readings during Sampling.

Transparency is an important indicator of particulate pollution of the water profile which affects light penetration suspended solids. This measurement addressed as Secchi depth in which a Secchi disk is used as an indicator of water transparency by lowering through the water column and measure the depth at which the disc disappears due to the absence of light at a specific depth.

Flow rate is one of the most important parameters that play key role in determining if a river reach is erosional or depositional. In addition, water flow is very important as a mechanical influence on the distribution of living organisms in fresh water environments. A traditional approach is used to measure flow rate by using orange or a similar coloured object to measure rate of movement at the water surface at specific distance in specific time. This can be used, in combination with determining the cross-sectional area (width from a map, plus measured depth), to measure the discharge in m^3/sec. By the second year, after March 2014, an electronic flow meter was introduced to record flow rate in m/sec. The collected data afterward is likely to be more accurate and reliable because of the mechanical effect of westerly winds which makes following a floating object problematical. Depth profiles across the river could not be measured and therefore discharge data was obtained from Environmental Agency for Adelphi Weir and two tributaries, the rivers Irk and Medlock. Discharge at the sites downstream of Adelphi Weir was estimated by using the sum of discharges from the three rivers to discharge level each site on the Irwell from the Mark Addy to the Turning Basin. The Irk discharges between Adelphi Weir and the Mark Addy sites and the Medlock between the Reagent Road Bridge and Pomona Dock sites. This prediction is considered reliable as the Irwell did not include sewer overflows or other point sources that could significantly contribute to the discharge (APEM Ltd, pers. com.) and could bias discharge estimation rate.

2.2.2 Water chemistry

Six major chemical parameters were measured as an indicator of water quality. These parameters are ammonia, nutrients nitrogen and phosphorus, dissolved oxygen (DO), five-day biochemical oxygen demand (BOD), total suspended solids, and organic matter. Samples were collected in one litre polypropylene bottles which were already washed in an acid bath of hydrochloric acid 10% to remove all residues from the previous sampling, and then the bottles were washed in distilled water.

Ammonia is analysed by using spectrophotometer at a 685-nm wavelength. Firstly, a sample is filtered through a 0.45 µm filter paper on return to the laboratory. The sample is then analysed immediately to avoid oxidation of the ammonia to nitrate to taking place in the samples. Nessler reagent is added before taking the measurement. Ammonia concentration is determined by using a standard ammonia curve using a range of pre-prepared standards of known concentration. In addition, Hana low range reagent kit was introduced in 2014 for better measurement and accurate reading (HI-93700-01; Hanna Instrument Ltd, Leighton Buzzard, UK). The wave length was 380 nm.

The main categories of nutrients are nitrate and phosphate. The samples are also filtered through a 0.45 µm filter paper. Also, unfiltered samples are taken from each sample to measure total nitrogen and phosphorus. The samples taken in 2013 were analysed for nitrogen using ion chromatography in the School of Geography and phosphorus by Inductively Coupled Plasma- Optical Emission Spectrometry (ICP-OES) in the School of Earth, Atmospheric and Environmental Science. Samples from 2014 were analysed by using a SEAL Auto Analyser 3 High resolution instrument (Seal Analytical Ltd, Southampton, UK). Cross-calibration was carried out to ensure comparability between methodologies. BOD was measured in the laboratory by incubating 25 ml of unfiltered sample at 20ºC for five days using a calibrated Hanna dissolved oxygen meter (Hanna Instruments Ltd, Leighton Buzzard, UK). Dark glass bottles were used to avoid the contribution of photosynthesis from autotrophic organisms at the sample. The difference in the level of dissolved oxygen at time 0 and five days is the level of BOD (in mg/L).

With regard to an accurate determination of suspended solids, specific 500ml collected in prewashed polyethylene bottles by rinsing 24 hours in acid bath and then distilled water was used to ensure removing any residues might interfere in post sampling. This approach depends on using filtration through an ignited pre-weighed Whatman GF/C filter paper (0.45um) technique that is used to determine both total suspended solids (TTS) and organic matter (OM). This method depends on filtering above mentioned and specific amount of water from each site immediately once the samples reached and arranged in wet lab. First ignition is for the filter paper at 500 to ensure that any moisture in the paper is removed prior to the filtration. The second weighing of filter paper is used to determine the difference which addresses the TSS concentration by using normal oven that provides temperature between 100-250C^0 as temperature needed is 105. The third weighing is used to a certain amount of OM in each sample by using high performance oven that provides 500C^0 to ensure that all organic matter has removed from the filter paper.

2.2.3 Trace metal analysis

Dissolved concentrations of trace metals were measured at the surface and the bottom of each site. Firstly, water samples were filtered by using 0.45µm Whatman cellulose acetate filter paper. Then filtered samples are kept in dark place after adding two drops of nitric acid (2% HNO_3); this acidification (pH<2) keeps the metal within each sample in solution until analysis by using ICP-MS (Agilent 7500cx, Agilent Technologies, Santa Clara, USA). Calibration was by matrix-matched standards. This method is able to detect very small concentrations of metals to 0.04ppm. The group of heavy metals analysed were Mn, Fe, Cu, Zn, As and Pb.

2.2.4 Chlorophyll a

Chlorophyll-*a* is considered as the main photosynthetic compounds in most plants, including phytoplankton. This pigment can therefore be used as an indicator to address phytoplankton biomass. The procedure depends on extracting this pigment by filtering specific amount of water sample from each site at from the surface and the bottom through a 0.45 filter paper. Then these papers are placed into plastic tubes filled with pure ethanol (70%) and kept overnight in the dark. Measurement of samples used a spectrophotometer at wavelengths of 665nm and 750nm (the latter to correct for turbidity) to calculate chlorophyll-a concentration by using the equation:

$$Chl\text{-}a \ \ (\mathrm{g/L}) = \frac{V_e \times f \times \mathrm{A}}{\mathrm{V}_s \times l}$$

Where:

V_e = total volume of solvent (ml)

$$f = \frac{1}{\textit{Specific extraction coefficient} \times 1000}$$

A = Absorbance at 665nm - absorbance at 750nm
V_s= Total volume of sample filtered (litres)
L = Cell path length (cm)
The specific extraction coefficient for chlorophyll-a in ethanol is 83.41 g^{-1} cm^{-1}.

2.3 Biological monitoring

2.3.1 Phytoplankton

The main technique used for phytoplankton is a standard sweeping method, by using 53 µm mesh net. The obtained sample then is then filtered through a 230 µm mesh (zooplankton net) to remove any grazers that might reduce phytoplankton species and numbers. Samples are preserved in distilled water after adding around three drops of 1% Lugol's iodine. This solution functions as a preservative and stains most microorganisms found in freshwater samples. Lugol's iodine solution may decrease due to photic degradation; therefore, samples have to be checked regularly in case they need a top-up to keep the samples workable over the long term. The majority of phytoplankton is identified to the genus level while the density is obtained by bio-volume per litre. Identification was carried out by compound microscope and using of Sedgewick-Rafter slide with chamber divided into 1000 squares. This chamber allows quantification of 1ml from the total of 30mls. Counting was carried out on up to 100 squares to identify the freshwater microorganisms, more counts were required during flash flood and high flows which dilute the phytoplankton and hence decreases the numbers of individuals per sample.

2.3.2 Zooplankton

Zooplankton determination is carried out by a technique that again relies on the 3m sweep method using a zooplankton net with a mesh of 230 µm. The samples are preserved in pure ethanol on site. Identification for species and density were the same as for phytoplankton. The procedure requires dissecting microscope and Bogorof slide. This slide can be filled with distilled water up to 5ml. The main container of 30ml sample was sieved by suitable mesh and all the residues were driven to Bogorof slide with distilled water.

2.3.3 Benthic invertebrates

Benthic invertebrates are considered as one of the most important parameters of water quality and river's health. Colonisation Samplers (Figure 3) were used to collect data by choosing four stations (Mark Addy, Regent Road Bridge, Old Trafford Road Bridge and Turning Basin) along the upper reaches of the lower river Irwell and upper MSC as the water was of course too deep to allow sampling using a nick-net. Samples were collected monthly, two times of warm season during August and September 2014, and the last two samples were collected respectively during the cold season of November and December of 2014. Samples were sieved, and the amount of sediment quantified to assess the effect of sediment deposition on the invertebrate populations. Pure ethanol was used to preserve the samples and microscopic investigation was carried to identify each individual to the family level.

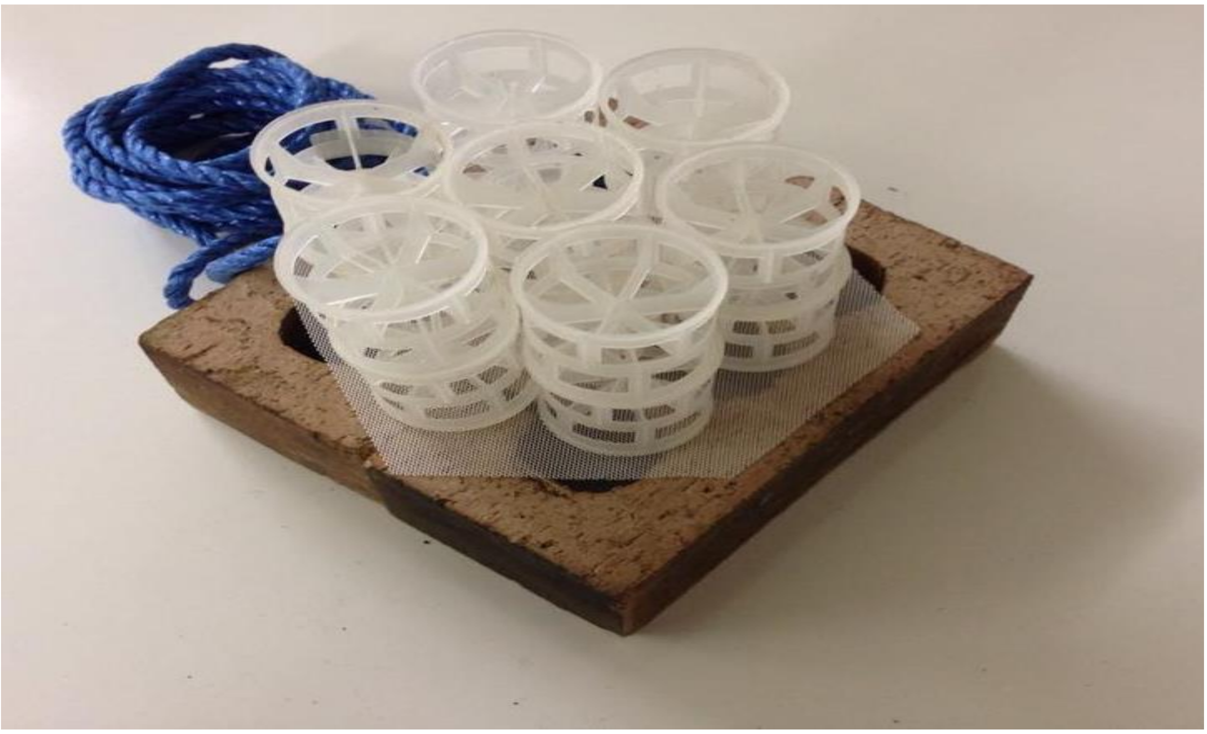

Figure 3- Invertebrate colonisation sampling unit.

Colonisation sampler units were assembled at the lab by following description of Watton & Hawkes (1984). The unit consists of polypropylene rings attached by either glue or plastic ties, and then the plastic rings were attached and fixed to the base of piece of brick to ensure the unit is properly positioned on river's bottom. The unit is tied by proper robe to surface at each station to ensure that the unit will be not dislodged by the flow. In addition, the unit was provided with plastic mesh of size 1mm at the base to ensure the entrapment of the benthic invertebrates as well as the sediment deposited in the unit (Tagliapietra and Sigovini, 2010). With regard to the sampling time scale and collecting of data, there are many differences of time scale depend on the system's nature and hydrology. Periods of 7-14 days are preferred in lotic systems and should not exceed 30 days (F. Richard Haure, 2006). In addition, the scale of 0-30days is considered as appropriate time to sample the most dominant organisms and representative species in the system (Meier et al 1979; Weber 1973).

2.4 Historical Data

To assess the past chemical and biological water quality of the lower River Irwell and upper reaches of the MSC past water quality data from obtained as external sources. Water quality data from Adelphi Weir was obtained from the Environment Agency's water quality archive for the period 2000 to 2012. The Environment Agency do not monitor downstream of this site on the River Irwell. However, the environmental consultancy company APEM Ltd have monitored three sites on the lower Irwell and the MSC and provided data for the MSC plus the open dock basin from 2000 to 2012 for physio-chemical parameters and from 2000 to 2007 for phytoplankton and benthic invertebrates (Figure 4). The following parameters were measured at both the EA and APEM sites: physio-chemical parameters such as pH, dissolved oxygen, conductivity, temperature, ammonia, biological oxygen demand, nutrients (Phosphate and Nitrate), water clarity (Secchi depth). In addition, biological data was provided for zooplankton and phytoplankton by APEM Ltd for their sampling site at Turning Basin.

Discharge data for the River Irwell at Adelphi Weir was obtained from the National Flow Archive from the same period. As stated above I was not able to measure discharge below Adelphi Weir as it was not possible to obtain cross-sectional area measurements of the river channel for safety reasons, discharge at all the sites was therefore calculated by adding discharge for, respectively the two tributaries Irk and Medlock at each site.

Rainfall data was obtained for Manchester City centre from 2010 to 30thDecember 2014.

Figure 4- Aerial photograph of Salford Quays showing APEM Ltd's sampling sites. Data was provided for sites 10 (MSC, see also Figure 2) and 1 (open dock basin), also it was used as site 4 and as station for colonisation sampler at Turning Basin (source: APEM Ltd)

2.5 Statistical analysis

The main statistical packages used were Excel 2013, Graph pad prism version 7 and SPSS. All these packages were provided by the IT department of Manchester University. The first was used for collating data on water quality and ecology in spread sheets plus carrying out basic statistical analysis such as the mean and standard error. Most statistical analysis were done by Prism graph pad including linear regression, correlation, t test, one-way anova and two-way anova. Primer (Clark, K.R and Warwick, 2001) and R (R Core Team, 2016) software were both used to analyse benthic invertebrate data and short term environmental factors that coincided with this ecological sampling.

2.6 Freshwater quality and ecology standards

2.6.1 Water quality standards

General Quality Assessment (GQA)

This scheme is used to assess the water quality in terms of chemical and physical parameters as well as the freshwater ecology with regard to the benthic invertebrate. GQA standards are based on measurements taken since 1988 and the first use of this scheme to assess the rivers in 1995. The basic measurements of this scheme are dissolved oxygen, biological oxygen demand and ammonia.

Table 2: UK river classification scheme based upon chemical parameters (General Quality Assessment (GQA) chemical grading).

Water Quality	Grade	DO (%)	BOD (mg L^{-1}) 90-%ile	NH3/NH4 (mg L^{-1}) 90-%ile
Very good	A	80	2.5	0.25
Good	B	70	4.0	0.6
Fairly good	C	60	6.0	1.3
Fair	D	50	8.0	2.5
Poor	E	20	15.0	9.0
Bad	F	-	-	-

Table 3: Phosphate grade.

Class	Grade limit (mg P L^{-1})	Description
1	<0.02	Very low
2	>0.02 – 0.06	Low
3	>0.060-00.1	Moderate
4	>0.1 – 0.2	High^{+}
5	>0.2 – 1.0	Very high^{+}
6	>1.0	Excessively High^{+}

Table 4: Nitrate grade.

Class	Grade limit (mg NO_3 L^{-1})	Description
1	<5	Very low
2	>5 – 10	Low
3	>10 - 20	Moderately low
4	>20 – 30	Moderate
5	>30 – 40	High
6	>40	Very high

Water Frame Directive (WFD-UK)

As stated in the introduction section 1.2.3, The European water frame directive is the most important piece of legislation that concern with water quality and ecology. The most important priorities of this scheme are to make recommendation and amendments in water quality standards. Therefore, the standards will be more applicable and conservatives of the principle of the natural environment.

The UK Technical Advisory Group (UKTAG)

UK technical Advisory Groups are working groups that include members from different agencies that take responsibility to revise water quality standards and making recommendations and amendments for better improvement and protection of freshwater environments.

Table 5: Classification of river chemistry according to the Water Frame Directive (WFD-2012) ta 90 percentile.

Variable	High	Good	Moderate	Poor
BOD (mgL^{-1}) Altitude<80m	4.0	5.0	6.5	9.0
BOD (mgL^{-1}) Altitude>80m	3.0	4.0	6.0	7.5
Ammonia-N (mgL^{-1}) Altitude<80	0.6	0.6	1.1	2.5
Ammonia-N (mgL^{-1}) Altitude>80	0.2	0.3	0.75	1.1
DO (% saturation)	>80	79	64	50
PO_4-P (mgL^{-1}) Altitude< / >80	0.05	0.12	0.25	1.0

2.6.2. Freshwater Ecology

General Quality Assessment (GQA) scheme took into consideration freshwater ecology and biology since it was established. There is a group of organisms belong to the benthic invertebrate includes mayfly nymphs, snails, shrimps and worms that mainly found in the river bed. This organism groups were studied carefully by the Biological Monitoring Working party and led to establish a method addressed as (BMWB score). These organisms classified into 83 taxa (Families) due to different response to the freshwater pollutants. The scale of this scheme is between 0 (pollution –tolerant taxa) and 10 (pollution sensitive taxa). Whereas the more sensitive taxa are found is the better water quality, otherwise the presence of pollution tolerant taxa reflects polluted environment that needs to be considered and may improve. The BMWP-score depends on the sum of all macroinvertebrate families in the specific environment, and the higher score the better water quality. In addition to BMWP-Index, the biodiversity and species richness were calculated by excel according to Shannon Weiner and Simpson indices.

Table 6: BMWP Score table.

Group	Families (Taxa)	Score
Mayflies, Stoneflies, Riverbug, Caddisflies or Sedgeflies	**22**	**10**
Crayfish, Dragonflies	**8**	**8**
Mayflies, Stoneflies, Caddisflies or Sedge flies	**5**	**7**
Snails, Caddisflies or Sedge flies, Mussels, Gammarids, Dragonflies	**9**	**6**
Bugs, Beetles, Caddisflies or Sedgeflies, Craneflies/Black flies, Flatworms	**24**	**5**
Mayflies, Alderflies, Leeches	**3**	**4**
Snails, Cockles, Leeches, Hog louse	**10**	**3**
Midges	**1**	**2**

On the other hand, there is another scheme that considers the average of tolerance scores of all macroinvertebrates that found in the sample and ranges between 0 to

10, this addressed as ASPT scheme (Average Score Per Taxon), the scheme calculated and outlined in table. The main difference between BMWP-score and ASPT scheme is that ASPT excludes the family richness and do not depend on the abundance of representative species in the sample.

Table 7: BMWP score, ASPT and interpretation.

BMWB Score	ASPT	Category	Interpretation
0-10	≤3.9	Very poor	Heavily polluted
11-40	4.0-4.9	Poor	Polluted or impacted
41-70	5.0-5.9	Moderate	Moderately impacted
71-100	6.0-6.9	Good	Clean but slightly impacted
>100	>9	Very Good	Unpolluted/unimpacted

Furthermore, Lincoln Quality Index (LQI) uses data obtained from the calculating of both BMWP and ASPT. Habitats are divided to tow main sections according to LQI, habitat-rich riffles and Habitat- poor riffles/pools. The overall quality rating and scores are given in table.

The most recent water quality index is Whalley, Hawkes, Paisley and Trigg (WHPT). There are two main difference between this scheme and BMPW-score, firstly it takes numeric abundance into consideration and calculation, secondly it depends on the highest number of taxa (106) therefore it is more accurate and the score values either 1 for natural status and 0 for the more affected environment (UKTAG-2014).

Table 8: EQR for WHPT-ASPT and WHPT-NTAXA.

	whpt	
	ASPT-EQR	NTAXA EQR
High/Good	0.97	0.80
Good/Moderate	0.87	0.68
Moderate/Poor	0.72	0.56
Poor/Bad	0.59	0.47

2.6.3 Assessing Trophic Status

One of the most the most patterns of standing water is Eutrophication in case the system is subject to nutrient enrichment especially phosphorous and nitrate.

Trophic state Indices (TSIs) is a crucial method that is used for lake classification and ranking due to water quality standards and parameters of fresh water which includes Secchi depth, chlorophyll-a and total phosphorous level. The scheme ranges between 0 – 100, the scale depends on the relationship between water clarity figures (Secchi depth) and concentration of both chlorophyll-a and phosphorous (Carlson, 1977). This apparent paradox is illustrated by calculating the trophic state index according to the following formula:

$$\text{TSI-P} = 14.42 \times \text{Ln}\,[\text{TP in } \mu\text{g/L}] + 4.15$$

$$\text{TSI-C} = 30.6 + 9.81\ \text{Ln}\,[\text{Chlor-a in } \mu\text{g/L}]$$

$$\text{TSI-S} = 60 - 14.41 \times \text{Ln}\,[\text{Secchi in metres}]$$

The average TSI is then mean of the three values:

$$\text{Average TSI} = \frac{(\text{TSI-P} + \text{TSI-C} + \text{TSI-S})}{3}$$

Table 9: Relationship between trophic state index (TSI) and water quality (from Carlson, 1977).

	Water Quality
< 30	Oligotrophic; clear water; high DO throughout the year in the entire hypolin
30-40	Oligotrophic; clear water; possible periods of limited hypolimnetic anoxia (I
40-50	Moderately clear water; increasing chance of hypolimnetic anoxia in su supports all swimmable/ aesthetic uses
50-60	Mildly eutrophic; decreased transparency; anoxic hypolimnion; macro problems; largely coarse fisheries; supports all swimmable/ aesthetic us "threatened"
60-70	Blue-green algae dominance; scums possible; extensive macrophyte probl
70-80	Heavy phytoplankton blooms possible throughout summer; dense macro beds; hypereutrophic
> 80	Cyanobacterial scums; summer fish kills; few macrophytes due to shadi phytoplankton; coarse fish dominant

3 RESULTS

3.1 Historical changes in water quality and ecology

3.1.1 Discharge and suspended solids

The discharge at Adelphi Weir and Pomona Docks was measured during the period from 2001 to 2013. Data was accessed from the National River Flow Archive for Adelphi weir on the River Irwell, and from the rivers Medlock and Irk immediately above their confluence with the Irwell (see Section 3.4). The downstream discharge of the system at Pomona Docks (one of the sites sampled during the 2013-2014 seasonal survey) or the Turning Basin opposite Salford Quays was derived from the above three sites as there are no point-sources that will significant contribute to discharge at Pomona Dock or at the Turning Basin (APEM Ltd pers. comm.).

Figure 5 shows the average discharge rates at Adelphi Weir and Turning Basin during 2001-2013. At both sites discharge fluctuates between 2 and 4 m^3/sec at Adelphi Weir and 4 and 8 m^3/sec at Pomona Docks. Increased discharge was observed at both sites in 2003, 2008 and 2012. During the whole period of 13 years, the discharge rate of Pomona Docks was more than double that of Adelphi Weir, with a discharge of 5.61±0.32 m^3/s for the former and 2.44±0.13 m^3/s for the latter (Figure 5B); hence the two tributaries in aggregate make a significant contribution to discharge and hence potentially to water quality of the lower Irwell/upper MSC (Figure 4). As there is no significant input between Pomona Docks and the Turning Basin, discharge is assumed to be the same at this latter site.

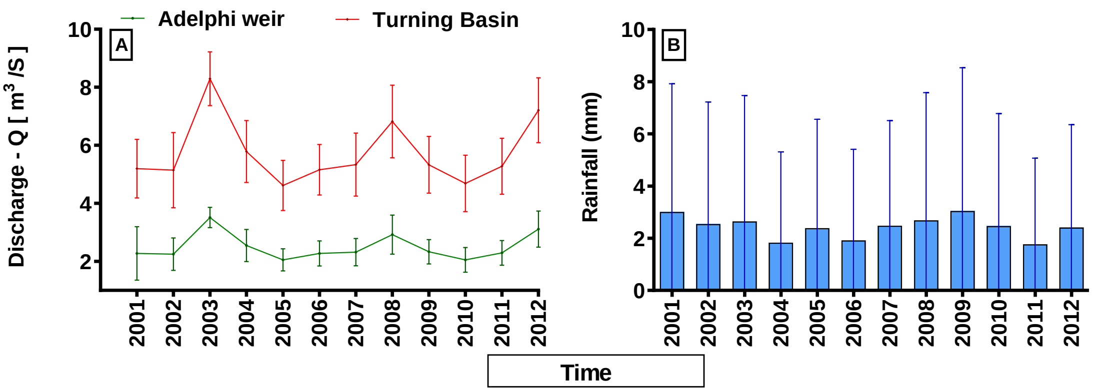

Figure 5- The annual average discharge at Adelphi Weir and Turning Basin between 2000 and 2012 recorded on a monthly basis from 2001 to 2012. Data shown as mean ± SE. Unpaired t test, $p<0.0001$ indicates significant difference between two sites. n=13.

Flow rate is considered as one of the most physio-mechanical factors that affects discharge and suspended solids in the system. No flow rate was available from the archives for 2003-2012 but flow rate measurements were taken by float method during 2013 and by a flow-meter during 2014 during the course of this study. The average flow rate was low at 0.146, 0.075, and 0.137 cm/sec upstream at the Mark Addy site, the middle of the system above confluence of the River Medlock at Pomona Docks and the downstream Turning Basin site respectively, Table 10. Flow varied greatly at all sites, from 0.009-0.5 cm/sec at Mark Addy, 0.025-0.223 cm/sec at Pomona Docks and 0.009-0.22 cm/sec at the Turning Basin. As expected, flow increased during periods of heavy rainfall.

Table10 : Flow rate (cm/sec) at monthly intervals between March 2013 and June 2014 at three sites on the Irwell/upper MSC. The mean flow rate and SE (in parenthesis) for each site is also shown.

Time	Mark Addy	Pomona Docks	Turning Basin
Mar-13	0.111	0.068	0.058
Apr-13	0.106	0.092	0.125
May-13	0.163	0.047	0.133
Jun-13	0.055	0.011	0.022
Jul-13	0.066	0.033	0.041
Aug-13	0.222	0.066	0.055
Sep-13	0.090	0.142	0.147
Oct-13	0.480	0.111	0.111
Nov-13	0.500	0.037	0.041
Dec-13	0.222	0.025	0.013
Jan-14	0.058	0.010	0.014
Feb-14	0.054	0.012	0.009
Mar-14	0.025	0.223	0.164
Apr-14	0.009	0.039	0.025
May-14	0.020	0.072	0.072
Jun-14	0.115	0.220	0.220
Average	0.146	0.075	0.137

Flow rate is a key influence on the level of suspended solids. It was not possible to collect sediment from the river bed due to the depth of the water column but an indication of the particle size distribution of deposited particulates was obtained from the invertebrate colonisation samplers and is shown in Table11.

Table11, the level of suspended solids (gm) collected by Colonisation sampler

Site	Time	Suspended solids size		
		1mm	250um	150um
Mark Addy	10/09/2014	21.95	105.77	90.76
	15/10/2014	62.75	225.47	339.43
	15/11/2014	50.3	178.68	280.93
	20/12/2014	48.9	93.15	280.94
Regent Road Bridge	10/09/2014	0	83.99	80.05
	15/10/2014	0	0	90.17
	15/11/2014	0	0	90.11
	20/12/2014	0	0	90.1
Trafford Road Bridge	10/09/2014	23.77	21.81	0
	15/10/2014	0	0	105.81
	15/11/2014	0	0	281.62
	20/12/2014	0	0	184.72
Turing Basin	10/09/2014	0	253.43	90.1
	15/10/2014	0	0	23.7213
	15/11/2014	0	0	105.92
	20/12/2014	0	0	105.74

It is very clear that there is an obvious difference in the size of suspended solids deposited in the colonisation samplers with a greater percentage of finer material deposited at the downstream sites. The particle size distribution at Adelphi Weir was markedly different again. The system was too shallow to position colonisation samplers, so no measure of deposited solids was obtained. Visual observation recorded a substrate consisting predominantly of gravel (40%), sand (40%) and silt (20%). Following rainfall, the percentage of gravel increased to 70%; followed by sand (20%) and slit (10%). According to the Hjulstorm-Sundborg diagram that shows the relationship between particle size and fall velocity (Figure 6) and the flow rate (velocity) in Table 10, it is very clear that the sediment of all the sites from the Mark Addy will consist predominantly of sand, silt and clay. Only during the occasional high flows of up to 0.5 cm/sec will erosion of the coarser particulates occur. This contrasts with gravel and a significant proportion of sand at Adelphi Weir.

Up to 100% of the suspended solids consisted of organic matter (Table 12). The percentage tended to decrease with total amount of suspended solids, perhaps reflecting increased contribution of surface run-off lower in organic content.

Table 12: Suspended solids and organic matter components of colonisation sampler substratum at Mark Addy [M.A], Regent Road Bridge [R.R.B], Trafford Road Bridge [T.R.B] and Turning Basin [T.B]. Figures in parentheses is the percentage organic content. *erroneous result

		S.S-mg/L			O.M-mg/L		
Site	**Time**	**1mm**	**250um**	**150um**	**1mm**	**250um**	**150um**
M.A	**Sep-14**	1.47	11.53	0.07	0.49 (33%)	5.13 (45%)	0.39 (*)
	Oct-14	62.75	225.47	4.79	0 (0%)	0 (0%)	1.07 (22%)
	Nov-14	50.3	178.68	0.92	0 (0%)	0 (0%)	0.27 (29%)
	Dec-14	0	0	618.78	0	0	18.63 (3%)
R.R.B	**Sep-14**	0	0.04	0.02	0	0.03 (75%)	0.02 (100%)
	Oct-14	0	0	0.67	0	0	0.18 (27%)
	Nov-14	0	0	0.1	0	0	0.04 (40%)
	Dec-14	0	0	31.98	0	0	4.21 (13%)
T.R.B	**Sep-14**	0.43	0.11	0.05	0.39 (90%)	0 (0%)	0.05 (100%)
	Oct-14	0	0	5.18	0	0	0.93 (18%)
	Nov-14	0	0	64.83	0	0	43.78 (68%)
	Dec-14	0	0	129.1	0	0	6.88 (5.3%)
T.B	**Sep-14**	0	10.72	0.7	0	0.17 (16%)	0.242 (34%)
	Oct-14	0	0	0.0704	0	0	0.0253 (36%)
	Nov-14	0	0	2.11	0	0	0.49 (23%)
	Dec-14	0	0	0.27	0	0	0.07 (25%)

In summary, discharge did not show an overall change between 2000 and 2012 although annual highs in 2003 and 2008 presumably reflect extreme rainfall events. The two tributaries make a significant contribution of discharge of study reach of the Irwell/MSC between Adelphi Weir and the MSC Turning Basin. It is also apparent from data on substrate composition that the system is changing from a more erosional system at Adelphi Weir (and above: e.g. National Rivers Authority, 1993) to an increasingly depositional system between Manchester city centre and the upper reaches of the MSC down to the Turning Basin. The depositional nature of the system is also reflected in the very low water velocity that favours deposition of suspended particulates that often contains a large amount of organic matter from upstream. As discharge has not markedly changed over the period 2000-2012 (Figure 5) and no re-engineering works have been carried out over the same period (APEM Ltd pers. comm.) it is suggested that substrate characteristics of the study area have likewise not changed.

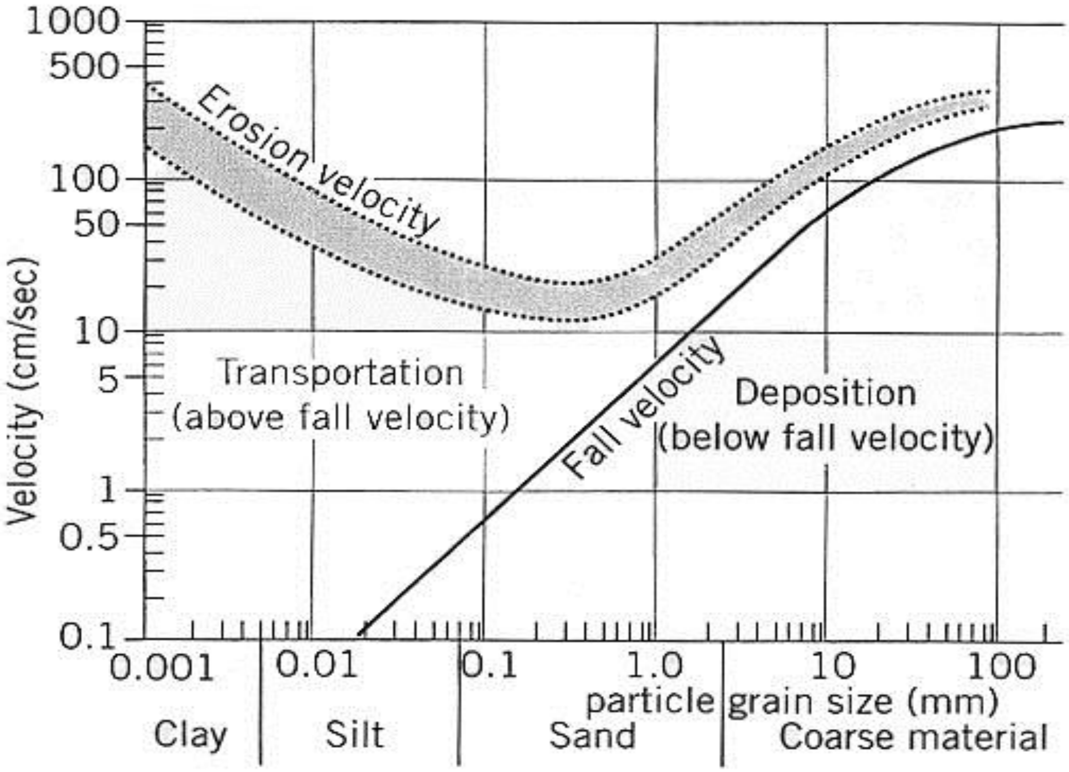

Figure 6- Hjulstorm-Sundborg diagram explaining the relationship between flow velocity and solid particle size. Also the graph outlines the change from an erosion to depositional system according to the effect of flow on erosion and transportation of sediments (Keylock, 2004).

3.1.2 Pollutant Flux

There was a noticeable change in BOD, suspended solids, nitrate and phosphate over the period 2000-2012. Changes in the flux of these parameters were examined in the Irwell (Adelphi Weir) and MSC (Turning Basin). Both the surface and bottom of the water column was examined at the latter site.

3.1.2.1 Biological oxygen demand

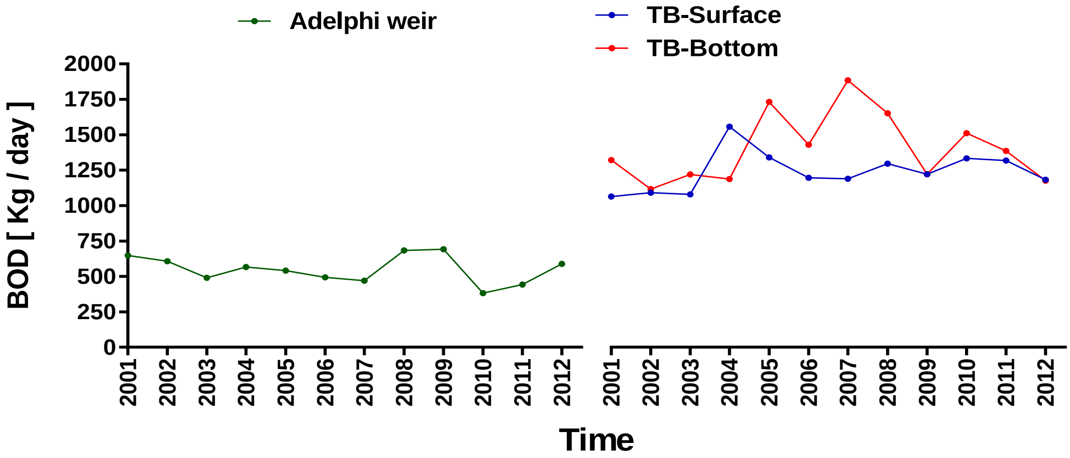

Figure 7- Seasonal changes in biological oxygen demand [BOD] flux at Adelphi Weir [AW], Turning Basin [TB] surface and bottom between 2000 and 2012. R^2 = 0.117, 0.863, 0.842 for Adelphi Weir, Turning Basin surface and respectively. P<0.05 showed no effect of discharge on the level of BOD at AW, but there was a highly significant (P<0.0001) temporal change at both the surface and bottom in the Turning Basin. N: AW = 12, TB =13.

Biological oxygen demand at Adelphi weir showed a slight decrease between 2001 and 2010 as the level decreased from 750 Kg/day to less than 500 Kg/day, Figure 7. BOD flux was greatest in 2008 and 2009. Statistical analysis by (ANOVA) showed no overall change in BOD upstream at Adelphi weir. The BOD flux was more changeable downstream at the Turning Basin with number of peaks, including again in 2005 and 2007, and was also slightly higher at the bottom in the Turning Basin with an average over 1400 Kg/day compared to 1200 Kg/day at the surface. There was significant overall increase in BOD flux between Adelphi Weir and the Turning Basin which is indicative of an increase in water column contamination by organic material between the two sites.

3.1.2.2 Suspended solids

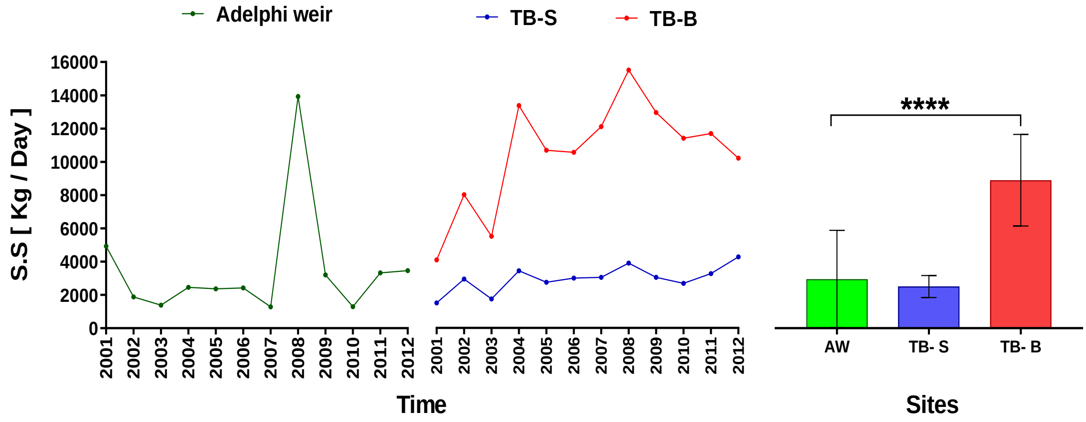

Figure 8- The effect of discharge on suspended solids flux at Adelphi Weir [AW], Turning Basin [TB] surface and bottom between 2000 and 2012. R^2= 0.341, 0.810, 0.490 for Adelphi Weir, Turning Basin surface and bottom respectively. P<0.0001 significant for AW and TB at surface and bottom. n= [AW=12, TB = 13].

The annual flux of suspended solids was generally steady at both Adelphi weir and the Turning Basin surface with an average of 5000 Kg/day between 2000 and 2012 (Figure 8). There was significant ($P<0.0001$) overall difference between the two sites as well as positive trend ($p<0.05$) between surface and bottom at the Turning Basin. A single peak was observed at Adelphi Weir in 2008 and elevated flux was noted at the Turning Basin in 2000. Suspended solid flux was slightly higher with an average of 10000 Kg/day at the bottom of the water column in the Turning Basin between 2001 and 2012, Figure 8. On the basis of a flux of 5000 Kg/day, yearly suspended solid load entering the Turning basin equates to nearly 1,600 tons/yr with an unknown amount being deposited in the sediments.

3.1.2.3 Nutrients

Nitrogen flux was very constant between 2001 and 2012 with an average of 500 Kg/day and a range of 150-200 Kg/day at Adelphi Weir, whereas nitrogen flux steadily increased downstream at the Turning Basin as the level was below 50 Kg/day in 2001 but reached more than 300Kg/day in 2012 (Figure 9). Discharge did not markedly increase over the same period (see Figure 5) which indicates increased input of nitrate with time. In contrast to nitrate, phosphorous flux was higher upstream at

Adelphi weir, at just over 50 Kg/day in 2001 and the level increased steadily reaching an average of 300 Kg/day in 2012. At the other end of the system downstream at the Turning Basin, phosphorous flux was similar to Adelphi Weir in 2001 at just over 50 Kg/day with a similarly stead increase, but only to around 200 Kg/day in 2012. Statistical analysis showed positive trends for both nitrate ($p<0.05$, $r2=0.5$) and phosphate ($p<0.05$,$r2=0.55$). The smaller amounts of phosphorus downstream may indicate deposition to the sediment.

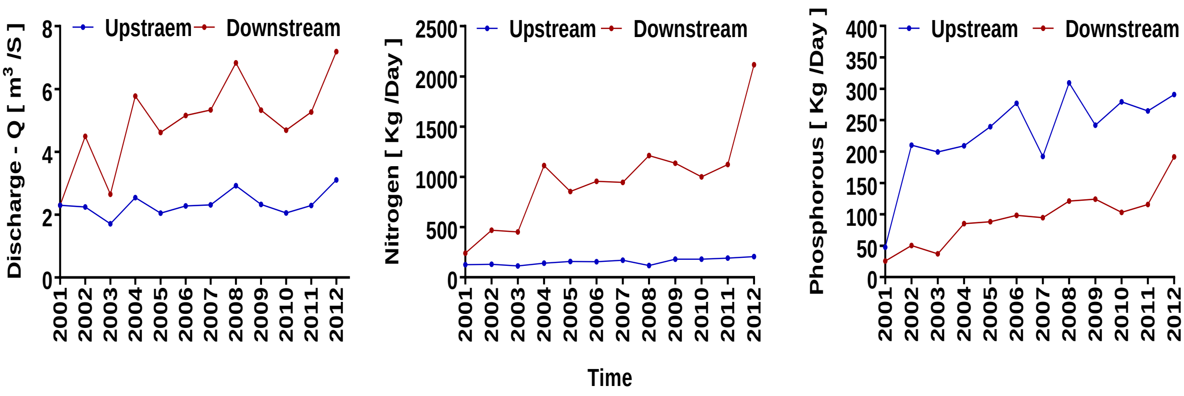

Figure 9 -The effect of discharge on nutrient flux at Adelphi Weir, and Turning Basin between 2000 and 2012. R^2= {0.078 for nitrate, 0.116 for phosphate at Adelphi Weir} and {0.483 for nitrogen, 0.578 for phosphorous at Turning Basin}. P<0.05 shows no significant change at Adelphi Weir, but a significant increase in the Turning basin for both nitrogen and phosphorous flux. n= [AW=12, TB = 13].

In summary, although discharge did not show an overall change between 2000 and 2012 suspended solid and nitrate flux increased. It is clear that the downstream site has a higher organic load on the basis of the BOD flux. Although the River Irwell is the major source of suspended solids, organic material and nitrate, in the absence of significant point source inputs (APEM Ltd pers. comm.) these increases reflect a significant additional contribution from the Irk and the Medlock. The exception was phosphorous which was much higher upstream at Adelphi Weir, presumably due to reduced input from one or both of the two tributaries and/or deposition in the sediment. The increase nitrogen flux with time in the Turning Basin may reflect increased nitrogen load from the tributaries whereas the increase in phosphorus flux between 2001 and 2012 points to an increase load in the Irwell as well as the Irk and Medlock.

3.1.3. Temporal change in water quality parameters

A number of physical and chemical parameters were recorded at Adelphi Weir by the EA and the Turning Basin of the upper MSC by APEM Ltd over the selected historical period of 2000-2012. In addition, data is presented for the open dock basin (Basin 6, see Figure 2) to allow comparison between the Irwell/MSC and a backwater with very little water exchange with the MSC.

3.1.3.1 pH

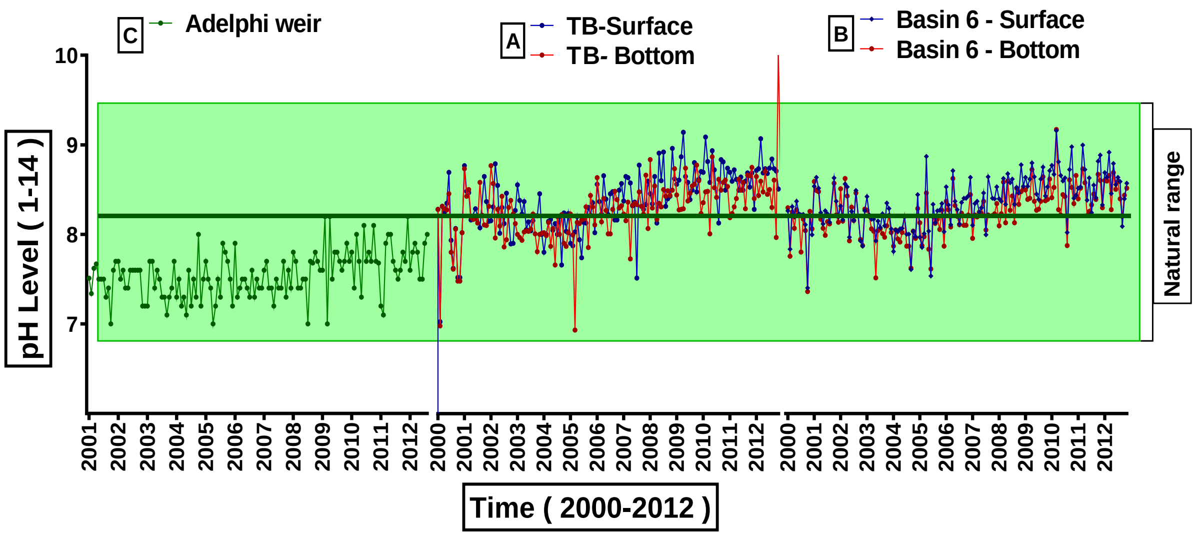

Figure 10- Historical pH changes in Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012 in (A) Adelphi Weir during 2000-2012, (B) Turning Basin (C) Basin-6. Values are the individual approximately monthly measurements taken on site by the EA and APEM.

According to figure 10 which shows the monthly measurements, and which displays the annual means, the pH at all three sites is mostly near-neutral with occasional peaks that indicate either slightly acidic or, more commonly, alkaline conditions in the Turning Basin and Basin 6. There appears to be a bigger variation in pH at the bottom of the MSC with surface pH as high as 8.25 being recorded (Figure 11 9A). Mean pH in the Turning Basin and Dock 6 was subject to a decrease to 7.0 in 2005 and then increased up to a maximum of 8.0 in 2010 (Figure 11A, B). There was small difference of around 0.5 in pH between surface and bottom in the Turning Basin and Basin-6. Upstream at Adelphi weir, the pH is stable most the first half of the last decade with slight increase of less than 1 degree up to 2012. The pH at Adelphi Weir was slightly more acidic than the system downstream and fluctuated between 5.5 and 7.3 (Figure 11 9C). The pH was stable over most of the first half of the last decade with slight increase thereafter but of less than 1 unit up to 2012 (Figure 10C).

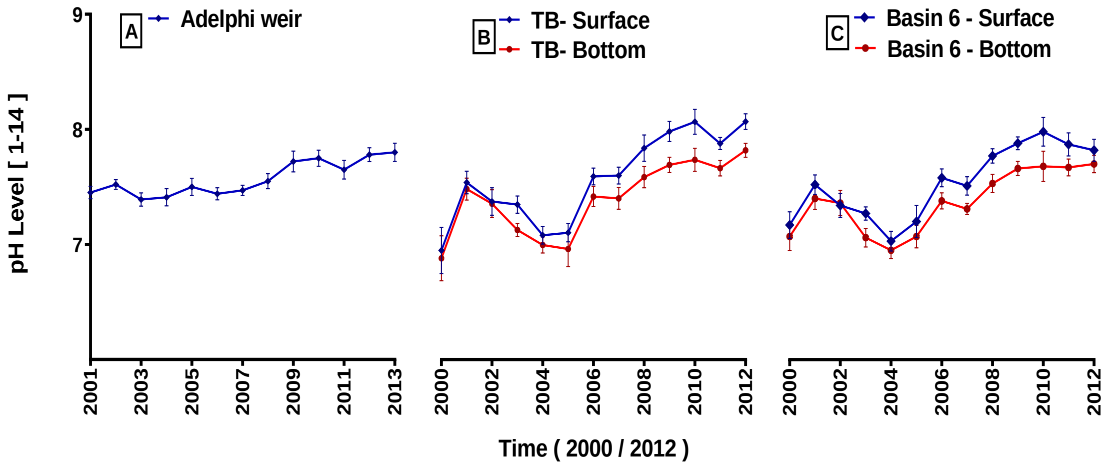

Figure 11- Annual mean pH in Adelphi Weir Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. Values are the mean of whole set of data provided by EA and APEM. Data shown as mean ±SEM. N = [Adelphi weir = 154, Turning Basin and Basin-6=168].

The monthly average showed that there was no significant (P <0.05) differences in pH during winter, spring and summer (see Figure 12). The analysis showed significant difference in the system between October and November, specifically in March at the two MSC sites.

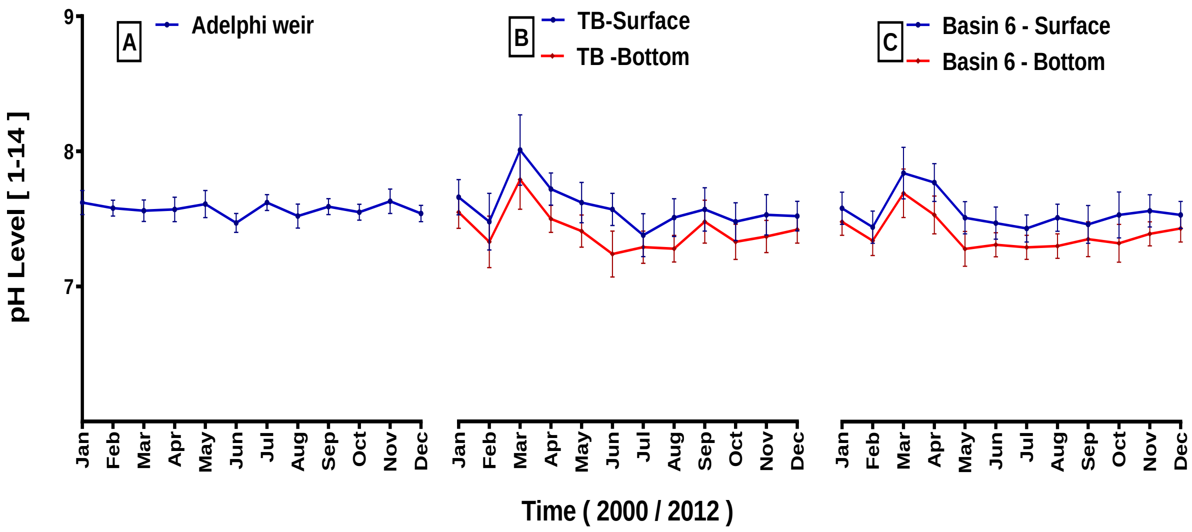

Figure 12- Seasonal changes in pH in Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. Values are the mean of a whole set of data provided by EA and APEM. Data shown as mean± SEM. N = [Adelphi weir = 154, Turning Basin and Basin-6=168].

The average pH during a whole year did not reflect any marked seasonal change except a slight increase between February and April and a peak during March, (Figure 12A, B). Statistical analysis showed significant differences between upstream (Adelphi weir) and downstream at MSC-Surface and bottom but only at the surface in Basin 6 which was higher in comparison with the MSC. ANOVA shows there was no significant seasonal trend from 2000 to 2012 (Figure13). Overall the whole system was slightly alkaline.

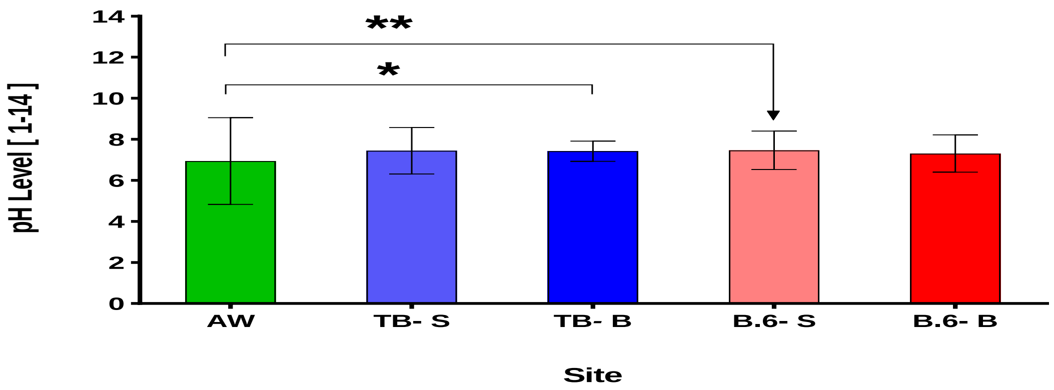

Figure 13- Statistical analysis of seasonal changes in pH in Adelphi Weir, Turning Basin and Basin-6 and Adelphi Weir [AW] between 2000 and 2012. Adelphi Weir, Turning Basin [TB] and Basin-6. Statistical analysis ($P < 0.05$) showed significant difference between Adelphi weir and other sites. There were no significant differences with time. Errors bar ± SEM. N = [Adelphi weir = 154, MSC and Basin-6=168].

3*.1.3.2 Temperature*

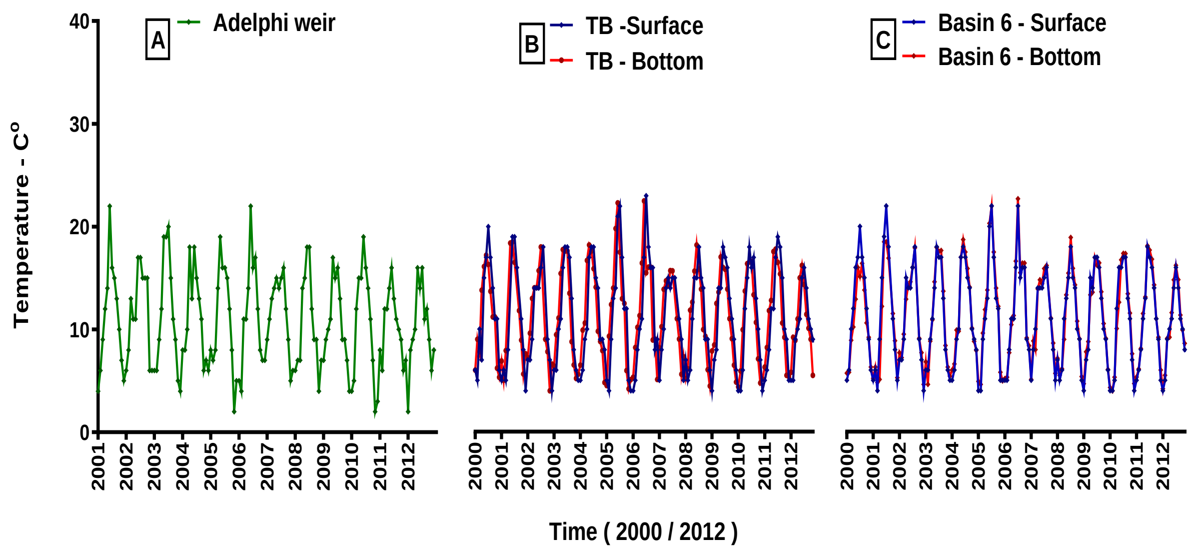

Figure 14- Historical temperature (C^0) changes in Adelphi Weir Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. The values of surface and bottom temperature were shown as blue and red lines, respectively. (C) Temperature at Adelphi Weir. Values are the individual measurements taken on site by the EA and APEM.

Water temperature at each site (Figure 14A) is typical of UK seasonal temperatures that vary between 5ºC during the winter, 10⁰C during autumn and spring and 20⁰C in the summer. There is a slight increase in temperature upstream at Adelphi Weir compared to the Turning Basin (Figure14B). The system is completely mixed as there are no differences between surface and bottom, and hence no evidence of stratification (Figures 15 and 16).

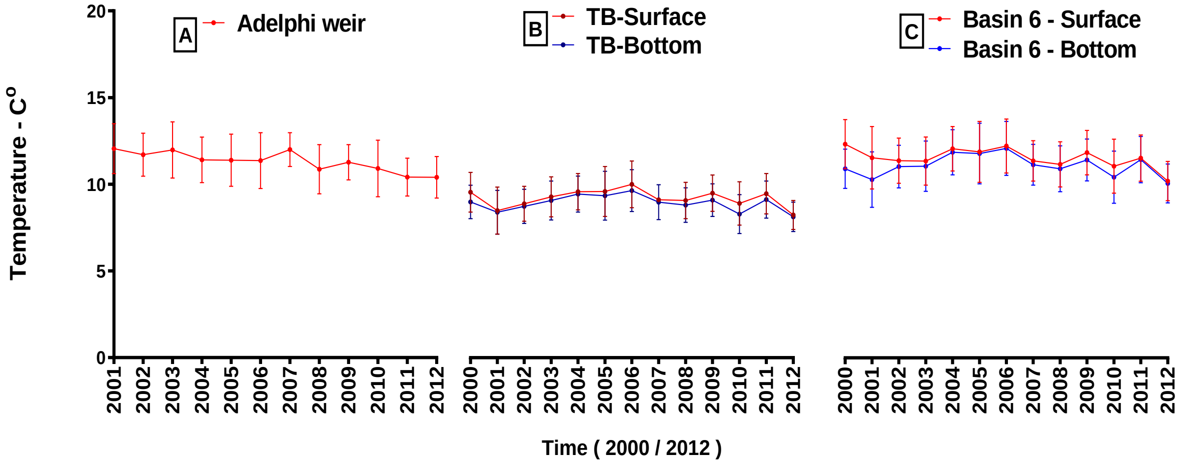

Figure 15- The annual mean of temperature (C⁰) at Adelphi Weir Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. Values are the mean of a whole set of data provided by EA and APEM. Data shown as mean ±SEM. n=13.

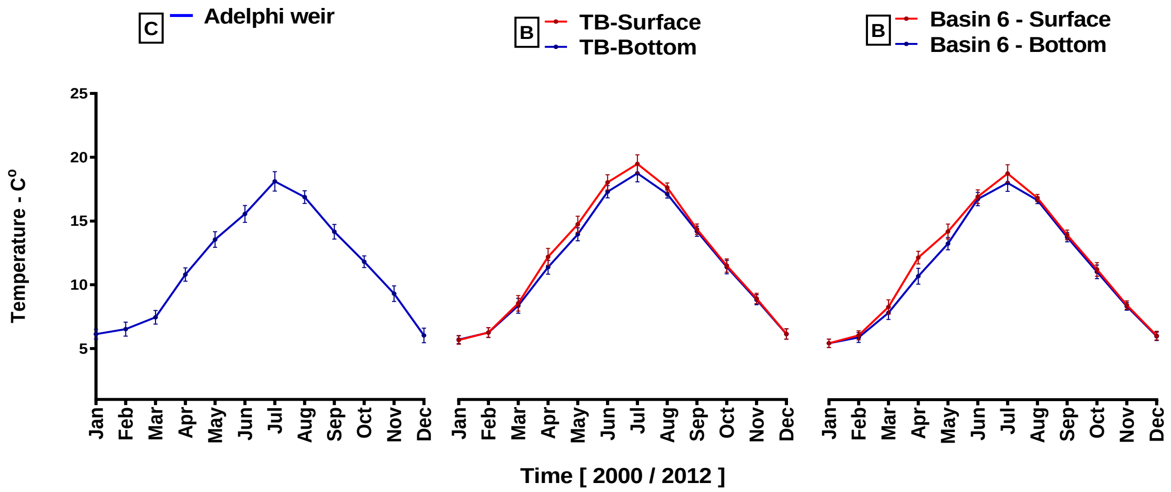

Figure 16- Seasonal changes of temperature (C⁰) at Adelphi Weir Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. Values are the mean of a whole set of data provided by EA and APEM. Data shown as mean± SEM. (n=13).

There is no difference in average seasonal temperature over the study period of 2000-2013 (Figure 17) with distance down-stream or with depth at the two lower sites with a depth <1m. Moreover, the average remains the same between 5^{0}Cin the winter cold season and 20^{0}C during summer from 2000 to 2012. Statistically there were no significance differences either between sites or during different seasons (Figure 17)

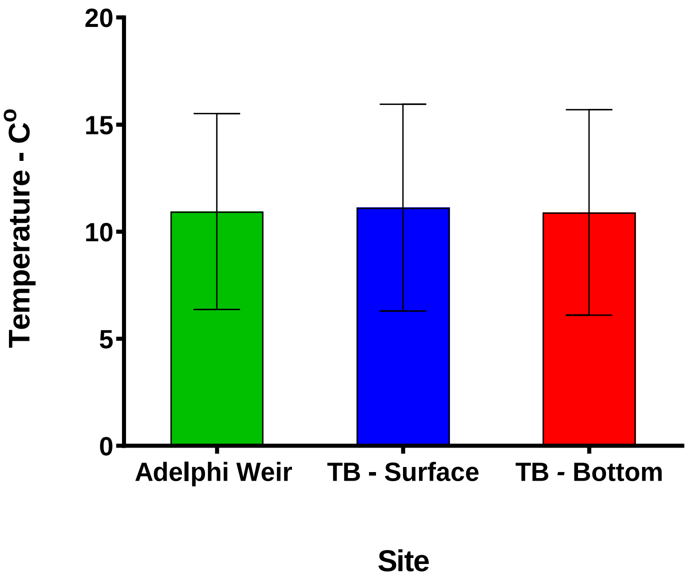

Figure17- Seasonal changes in temperature (C^{0}) for at Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. $P < 0.05$ showed there were no significant differences between sites (n=3) and time. Errors bar ± SEM.

3.1.3.3 Conductivity

Conductivity fluctuated most of the time between 300µS and 700µS (Figure 18) which is generally above the normal range in unpolluted waterbodies (>70 - 250 µS) according to the UK-WFD/2015. Yearly average values of conductivity are mostly stable at around 500µS, (Figure 18A and C). The figures also show that there is no difference between surface and bottom where the trend is identical at both MSC, and the semi-enclosed Basin 6.

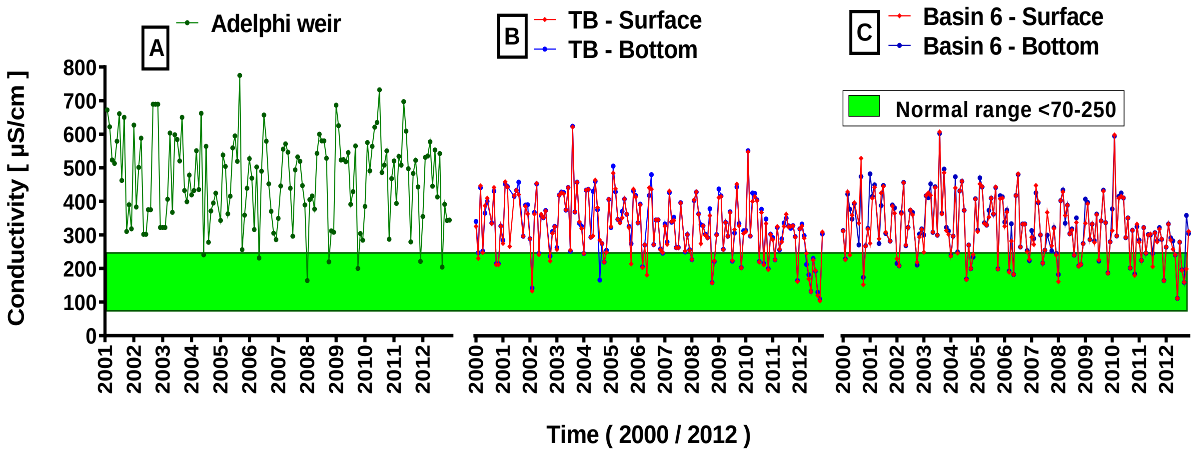

Figure 18- Historical changes in conductivity at Adelphi Weir Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. The values of surface and bottom conductivity are shown as blue and red lines, respectively. (C) Conductivity at Adelphi Weir. Values are the individual measurements taken on site by the EA and APEM

Although the yearly average values (Figure 19, A and C) are mostly stable at around 500µS, conductivity shows a marked decrease to 300-350 µS in 2012. The system was also quite changeable upstream at Adelphi Weir but conductivity does not mirror the changes in the Turning Basin and Basin 6.

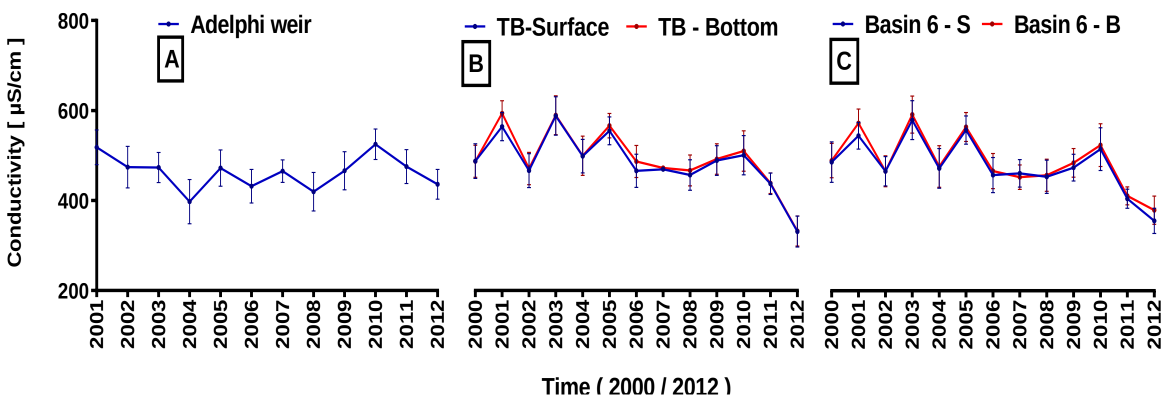

Figure 19- The annual mean of conductivity at Adelphi Weir Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. Values are the mean of a whole set of data provided by EA and APEM. Data shown as mean ±SEM. n=13.

The monthly average conductivity shows the system is subject to more seasonal changes downstream, Figure 20A and B, compared to upstream which is more stable but shows a more pronounced decrease, from 550 to 350µS, during the autumn (Figure 20C). Conductivity is slightly higher in the summer, perhaps as a result of reduced dilution from rainfall.

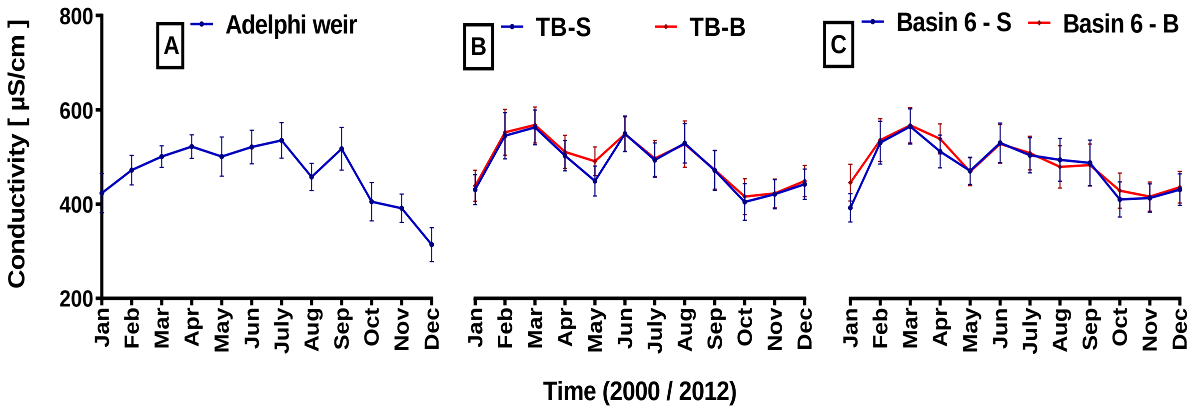

Figure 20- Seasonal changes of conductivity at Adelphi Weir Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. Values are the mean of a whole set of data provided by EA and APEM. Data shown as mean± SEM. (n=13).

There was no significant (P<0.05) overall difference observed in the average conductivity between sites (2000-201), Figure 21 suggesting that the two tributaries have a similarly high conductivity.

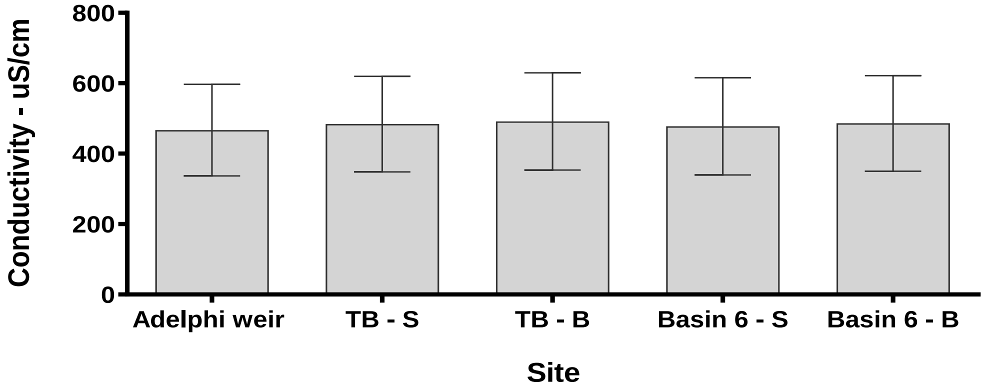

Figure 21- Statistical analysis of seasonal changes of conductivity at Adelphi Weir Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. (P < 0.05) showed there were no significant differences between sites (n=5). (n=13). Errors bar ± SEM.

3.1.3.4 Dissolved oxygen

Dissolved oxygen fluctuated most of the time around 75 % saturation with a slight variation between surface and bottom, Figure 23. Hence the sites are classified as ‘Very Good’ or ‘Good’ according to the Chemical GQA. Occasional low values were recorded, down to ‘Fair’ or ‘Poor’ according to the GQA. There is also a coincidence between sites in the system with a relatively low variation upstream at Adelphi Weir. The average DO is more changeable in the lower reaches of the study area, perhaps reflecting the effect of the slower flow rate and hence less re-aeration (Figure 24 A and C). DO in Basin-6 is a little more stable. Noticeable differences between surface and bottom were observed at both sites despite the installation the Helixor mixing devices; sometimes values fell below 20% saturation in the Turning Basin and Basin 6 (Figure 23). There is a progressive rise in DO from 2004 at Adelphi Weir and the Turning Basin (Figure 24) which followed a deterioration from 2001.

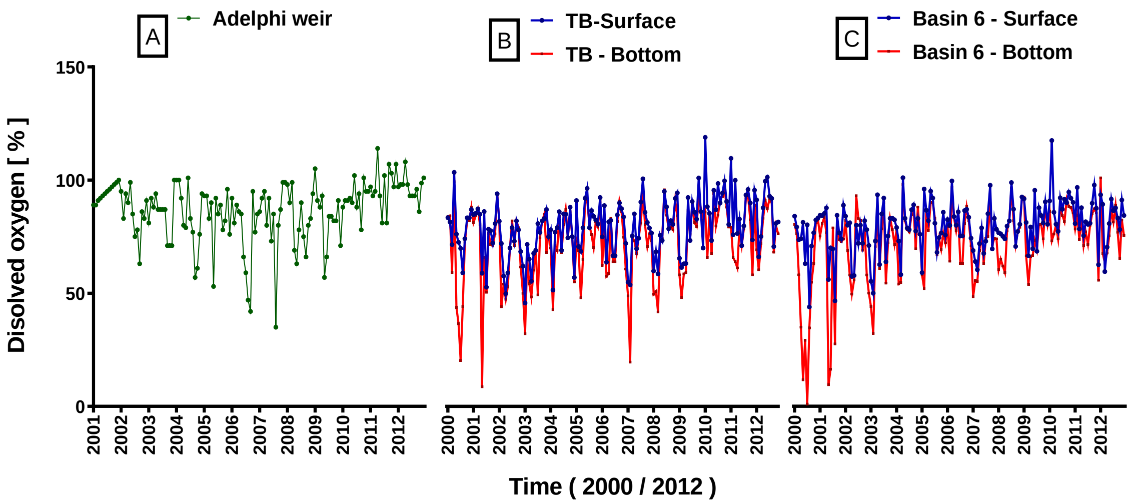

Figure 22- Historical changes in dissolved oxygen at Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. The values of surface and bottom dissolved oxygen are shown as blue and red lines, respectively. Values are the individual measurements taken on site by the EA and APEM.

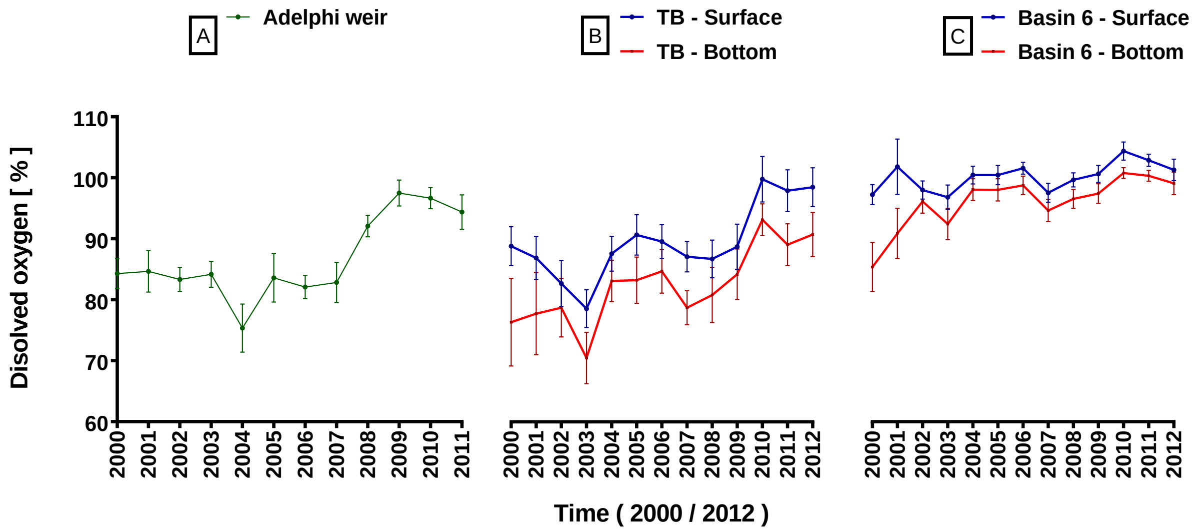

Figure23- Annual mean dissolved oxygen [DO] at Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. Values are the mean of a whole set of data provided by EA and APEM. Data shown as mean ±SEM. n=13.

Dissolved oxygen decreases during the summer which reflects the effects of high temperature in decreasing the solubility of DO and perhaps less dilution of organic inputs. This is very clear from Figure 25 that shows all sites on the system, including the shallow and well-mixed Adelphi Weir site; it is therefore to be expected that the average DO over the study period was markedly higher (P<0.05) at Adelphi Weir than the downstream sites of Turning Basin or the adjacent and interconnected Basin 6 (Figure26).

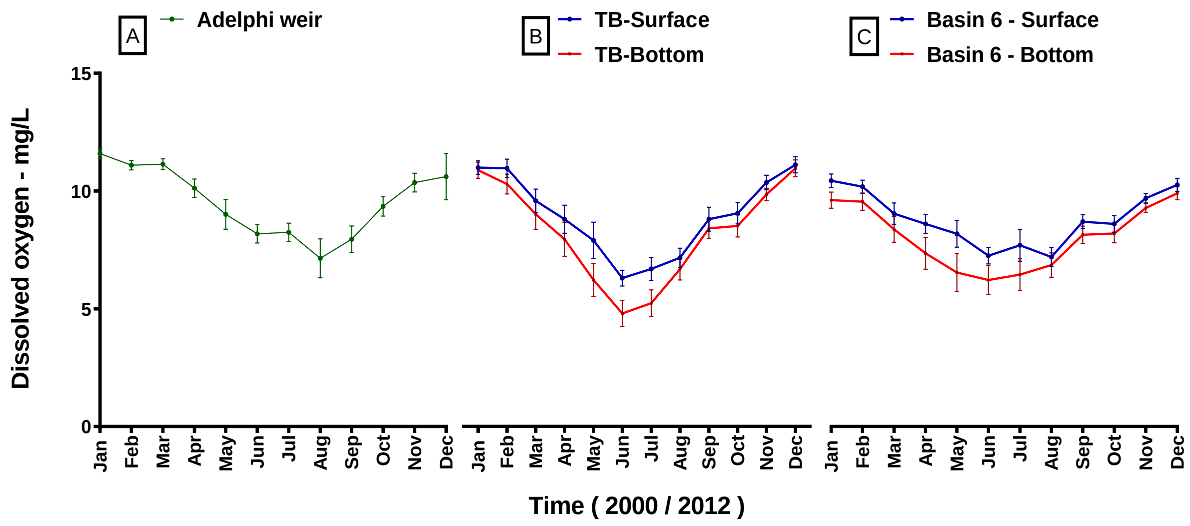

Figure24: Seasonal changes of dissolved oxygen [DO] at Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. Values are the mean of a whole set of data provided by EA and APEM. Data shown as mean± SEM. (n=13).

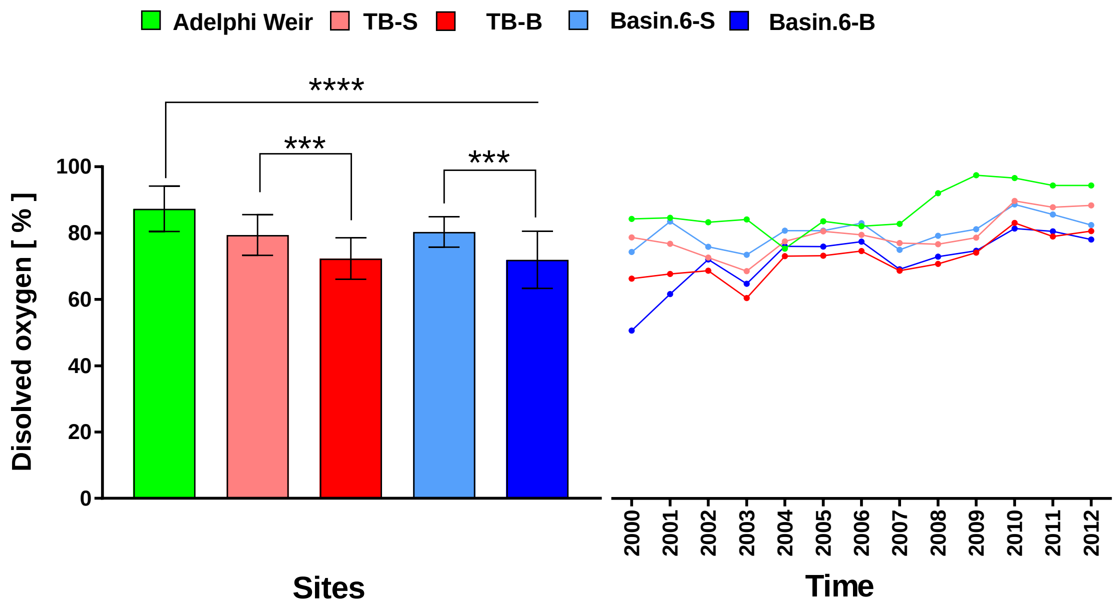

Figure 25 - Statistical analysis of seasonal changes of dissolved oxygen [DO] for Adelphi Weir, Turning Basin [TB] and Basin-6 between 2000 and 2012. The asterisks show there were significant ($P < 0.05$) differences between site and during seasonal variations. N= (sites=5, years=12). Errors bar ± SEM.

3.1.3.5 Biological oxygen demand (BOD)

Biological oxygen demand fluctuated around normal range (GQA 2.5 Very Good, 4 Good) at Adelphi Weir and Basin-6 with no significant differences between surface and bottom (figure 27). BOD is much higher downstream at the Turning Basin (Figure 27B, C, Figure 26) compared to the upstream site (Figure 27A) and falls within the range ‘Fairly Good’ to ‘Fair’. Adelphi Weir is also slightly more stable and with less large variation although occasional spikes in BOD are also observed at this site.

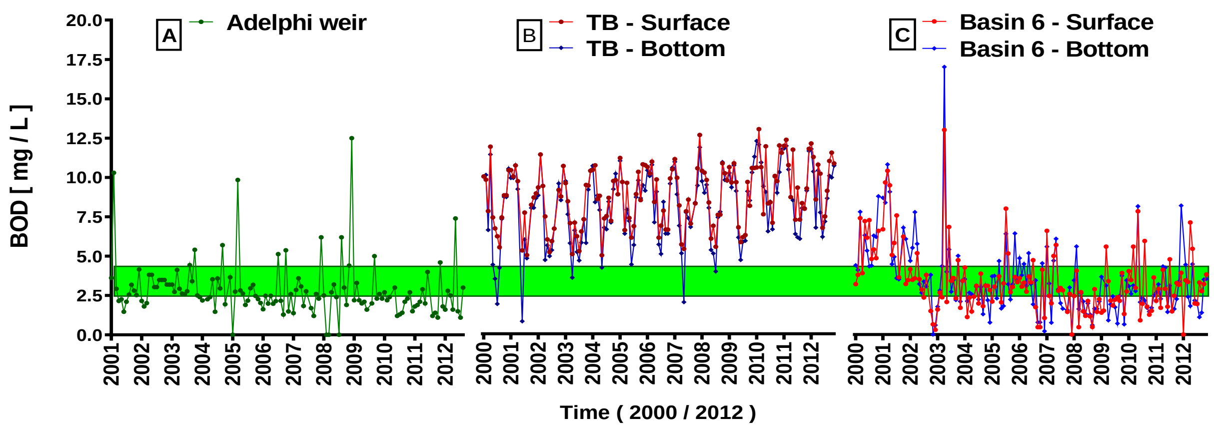

Figure 26- Historical changes in Biological oxygen demand (BOD) Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. The values of surface and bottom biological oxygen demand are shown as blue and red lines. The green bar shows the GQA @Very Good’ designation. Values are the individual measurements by the EA and APEM.

The annual average BOD shows a noticeable decrease over the time scale of the historical data-set, particularly from 2002 to 2012 at the lower sites (Figure 28A, B). BOD dropped from nearly 8mg/L to just under the 4 mg/L over this period, although there was a slight increase between 2005 and 2009 in the MSC. There was no trend in BOD at Adelphi Weir.

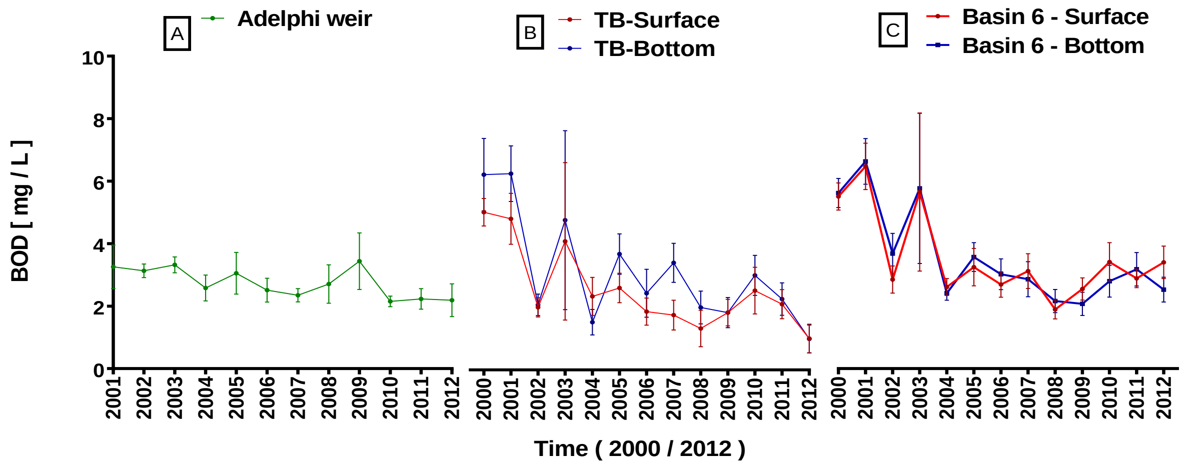

Figure 27- The annual mean of Biological oxygen demand (BOD) at Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. The values of surface and bottom biological oxygen demand are shown as blue and red lines Values are the mean of a whole set of data provided by EA and APEM. Data shown as mean ±SEM. n=14.

The average seasonal change over 2000-2012 reveals that the BOD is generally more upstream at Adelphi weir, Figure 28 C. Downstream the system was slightly more variable with a significant peak in BOD during April. Basin-6 experienced a significantly higher BOD at the surface compared to the bottom of the basin during the summer (Figure 28B). There were significant differences between sites with Adelphi Weir having a lower BOD over the study period compared to the Adelphi Weir (Figure29). Interestingly the BOD in Turning Basin at the surface was significantly lower than at the bottom of the water column whereas in the adjacent Basin.6 the converse was the case (Figure29 A, B). The apparent fall in BOD with time was confirmed on the basis of a significant (P<0.0001) difference in mean BOD at all sites from 2000 to 2012, figure30.

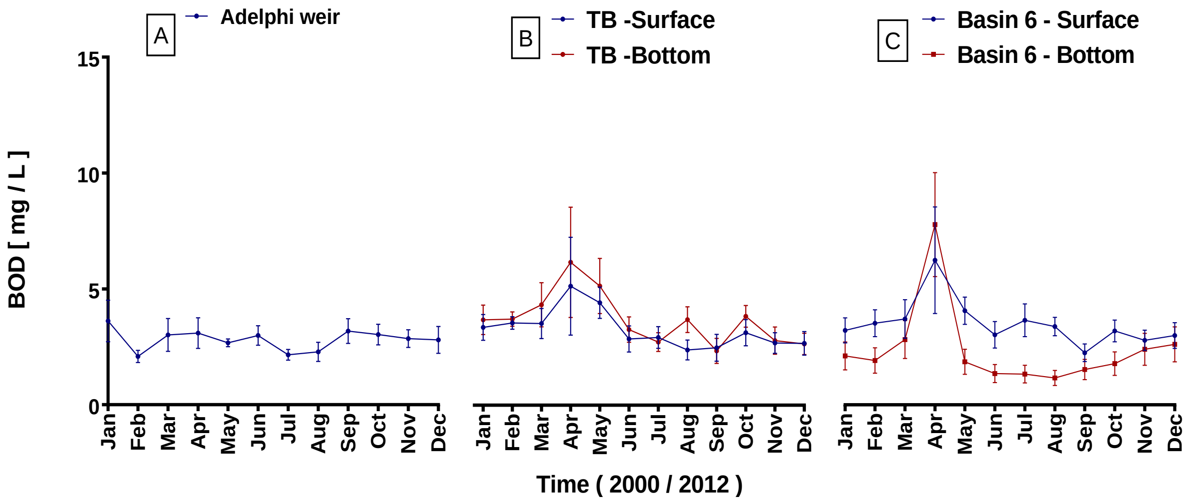

Figure 28- Seasonal changes of Biological oxygen demand (BOD) at Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. Values are the mean of a whole set of data provided by EA and APEM. Data shown as mean± SEM. (n=14).

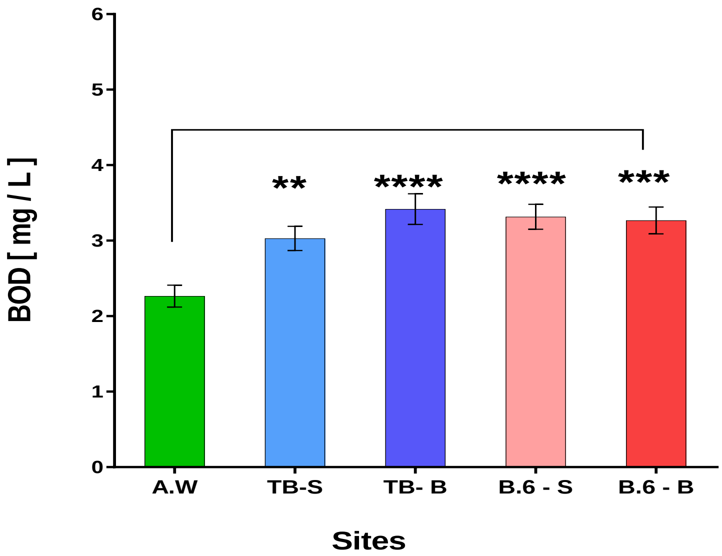

Figure29- Statistical analysis of seasonal changes of Biological oxygen demand [BOD] for Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (P < 0.05) showed there were significant differences between sites (n=5) . Errors bar ± SEM.

3.1.3.6 Chlorophyll-a

Changes in chlorophyll-a concentration over the study period is shown in Figure 30. The peaks of course reflect increases during the spring and summer growth periods which were usually of the order of 10-20µg/l but exceeded 30µg/l in 2001 and 2011 downstream at Adelphi Weir. Peaks exceeding 30µg/l were also observed at the downstream site and the semi-enclosed Basin 6 but do not always correspond to the upstream site (Figure31). Chlorophyll-a concentrations are generally within the mesotrophic range of 2.6-20µg/l according to Carlson 1996 although occasional values are indicative of eutrophic conditions.

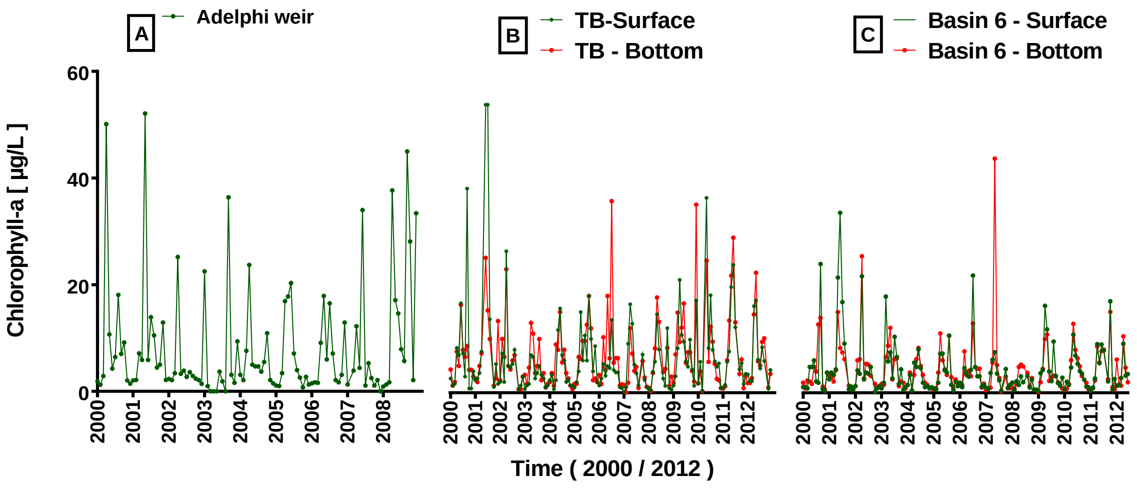

Figure30- Historical changes in Chlorophyll-a at Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. The values of surface and bottom biological oxygen demand were shown as blue and red lines, respectively. Values are the individual measurements taken by the EA and APEM.

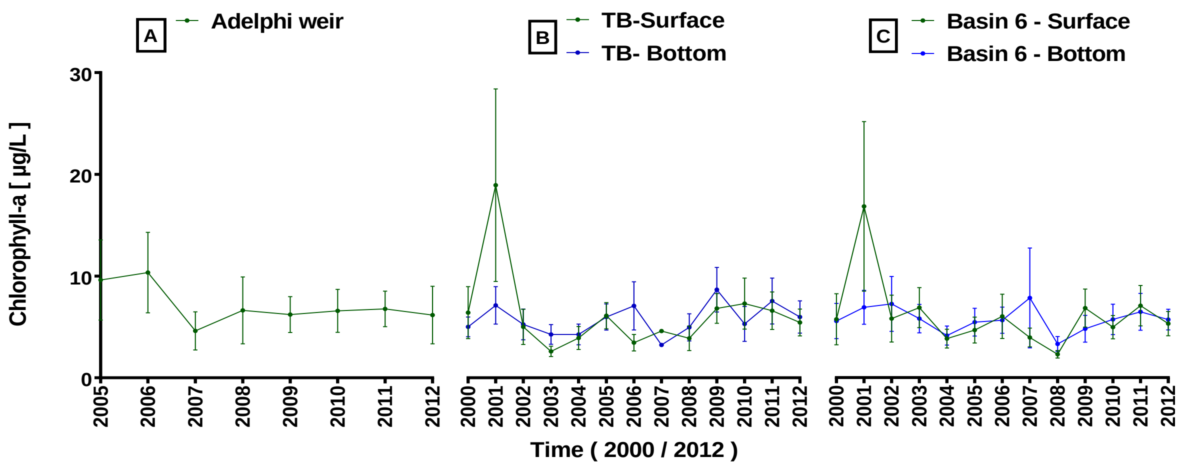

Figure 31- The annual mean of Chlorophyll-a at Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. Values are the mean of a whole set of data provided by EA and APEM. Data shown as mean ±SEM. n=14.

The annual mean chlorophyll-a was mostly around 5 µg/l (Figure 32). As expected, the monthly mean figure shows the concentration was significantly higher during summer, at just less than 20 µg/l (Figure33). There were no significant differences between sites between 2000 and 2012 (Figure34).

The relationship between chlorophyll-a and light extinction as measured by Secchi depth was examined in the MSC and Basin 6. There is no significant relationship between Secchi depth and chlorophyll-a levels in the MSC, Figure35. Secchi depth varied from 50cm to 100cm but was not correlated with chlorophyll suggesting that other particulates in the water column significantly contributed to changes in transparency.

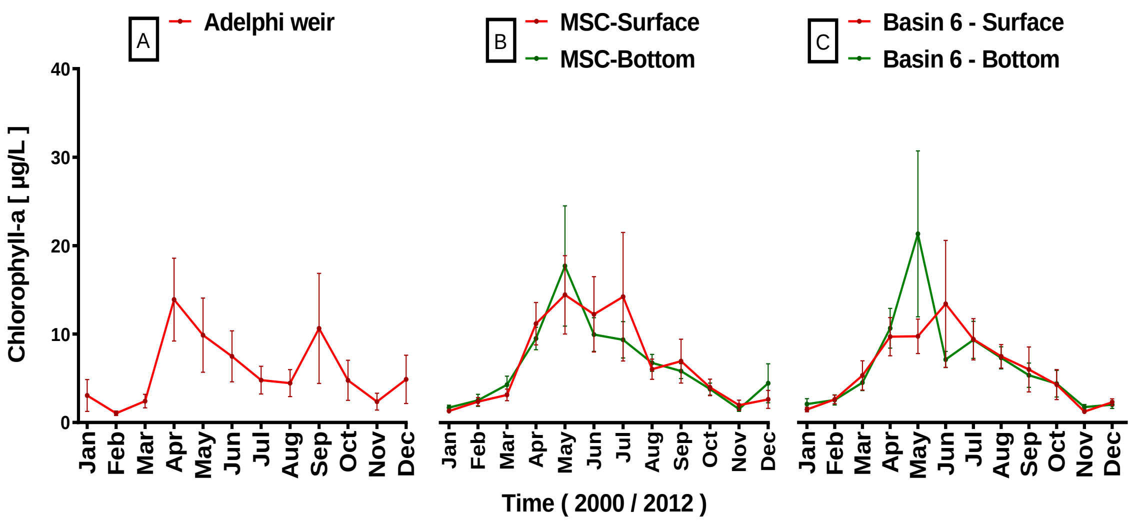

Figure 32- Seasonal changes of Chlorophyll-a at Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. Values are the mean of a whole set of data provided by EA and APEM. Data shown as mean± SEM. (n=14).

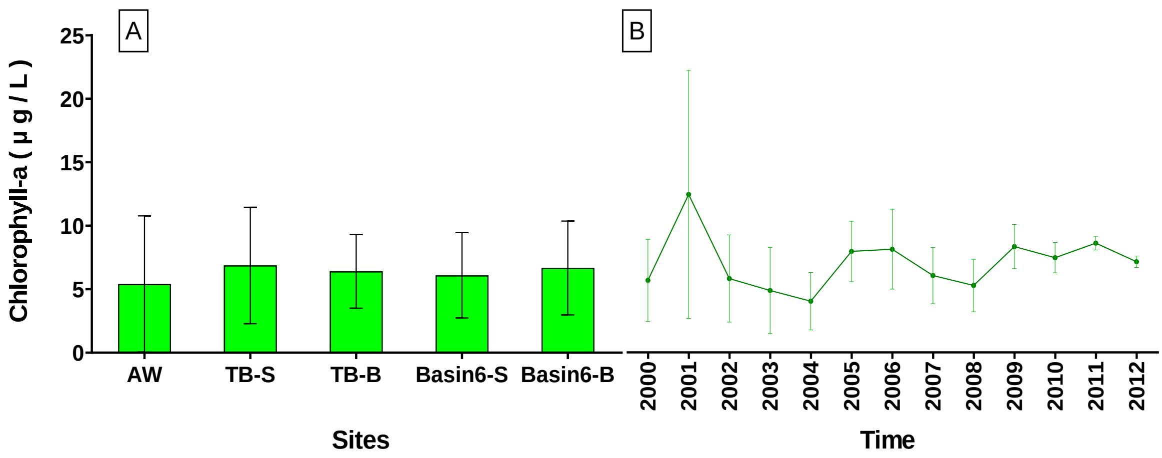

Figure 33- Statistical analysis of seasonal changes of Chlorophyll-a at Adelphi Weir, Turning Basin and Basin-6 between 2000 and 2012. (A) Adelphi Weir, (B) Turning Basin [TB] and (C) Basin-6. ($P < 0.05$) showed there were no significant differences between sites (n=5) and along time (n=14). Errors bar ± SEM.

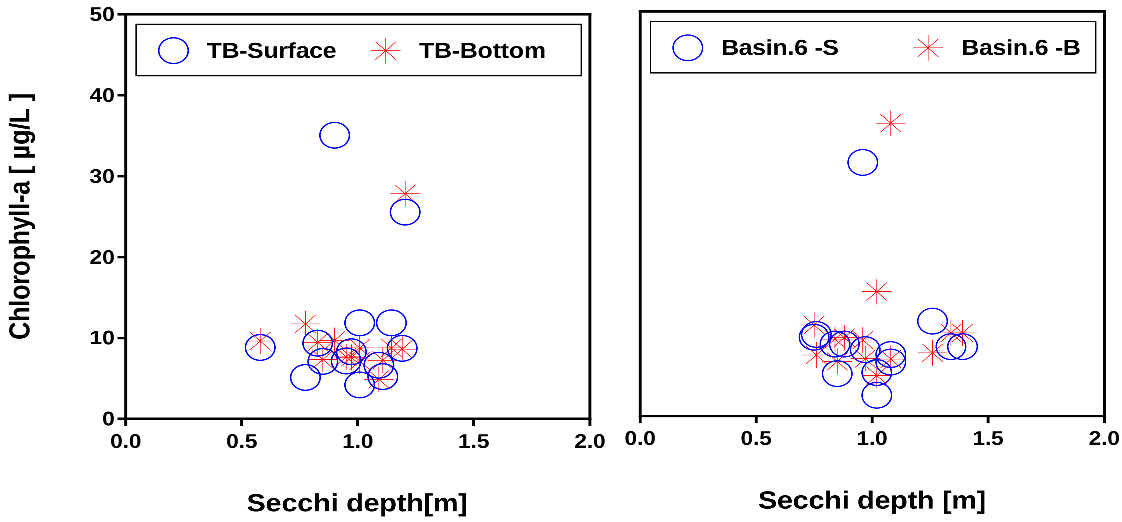

Figure 34- Relationship betweenSecchi depth and Chlorophyll-a in the Turning Basin and Basin-6 between 2000 and 2012. (A) Turning Basin [TB], (B) Basin-6. Values at the surface presented as circles with dark blue colour and bottom values presented as stars with light red colour. R^2 values (Turning basin 0.6558 and 0.4432 for surface and bottom respectively/ Basin-6. 0.00279 and 0.0129 for surface and bottom respectively) were generally very low and hence revealed no correlation between parameters.

Figure 36 demonstrates the relationship between phosphorous and chlorophyll concentration in the lower part of MSC. It can be seen that phosphorous level is positively related to chlorophyll concentration, although it might be due to the single high chlorophyll value.

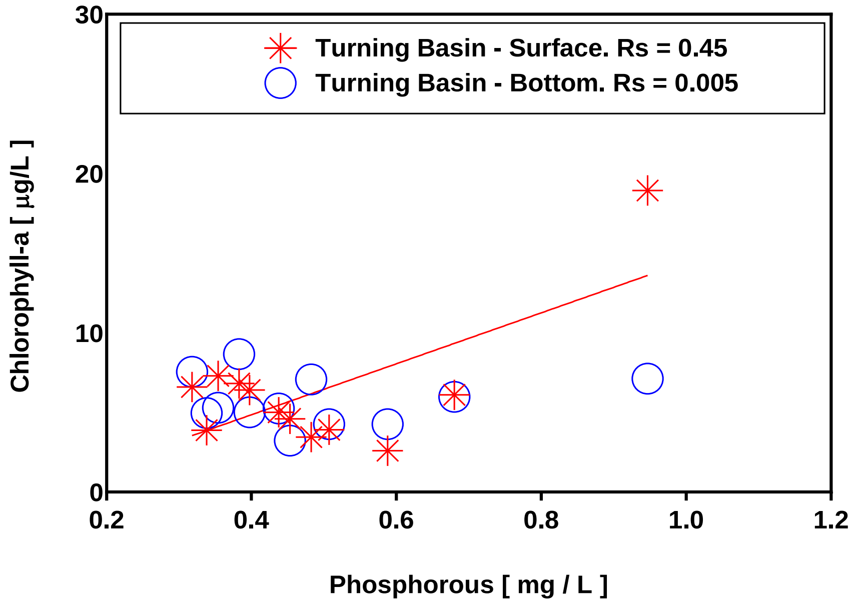

Figure 35- Relationship between phosphorous and Chlorophyll-a in the Turning Basin at the surface and bottom between 2000 and 2012. Turning Basin surface level measurements presented in red stars and blue circles for bottom. R^2values generally are low but surface level values were positive and showed significant relationship between chlorophyll-a and phosphorous with a R^2 value of 0.45.

3.1.3 Nutrients (Nitrogen [NO_3-N], Phosphorous [PO_4-P] and Ammonia

Nitrate varied mostly between 2 and 6 mg/l, but in some years dropped to below detection limits while in others reached 8mg/l and concentrations of this nutrient were therefore highly variable, Figure37A. Generally, concentrations were classed as Very Low according to the GQA nitrate grading and never exceed the 'Low' grade (Figure 37A). When the data was averaged for each year the trend was for a significant decline through the time in the Turning basin as NO_3-N was just over 7 mg/l in 2002 but fell to close to 1 mg/l in 2012 (figure37B). In contrast, the trend at Adelphi Weir was for an increase from a mean to 3 to 5 mg/l from 2001 until 2006; then a slight decrease up to 2012, and afterward an increase again by the end of the study period (2011 2012). Statistical analysis showed significant differences between the upstream site at Adelphi weir and downstream site at Turning Basin on the MSC, Figure38. The monthly average increased in late winter/early spring in the MSC which may reflect mineralisation following decline and death of phytoplankton at this time.

In contrast a slight increase was observed in Adelphi Weir over the summer and autumn. The general seasonal average nitrate was between 3-6 mg/l (Figure37C).

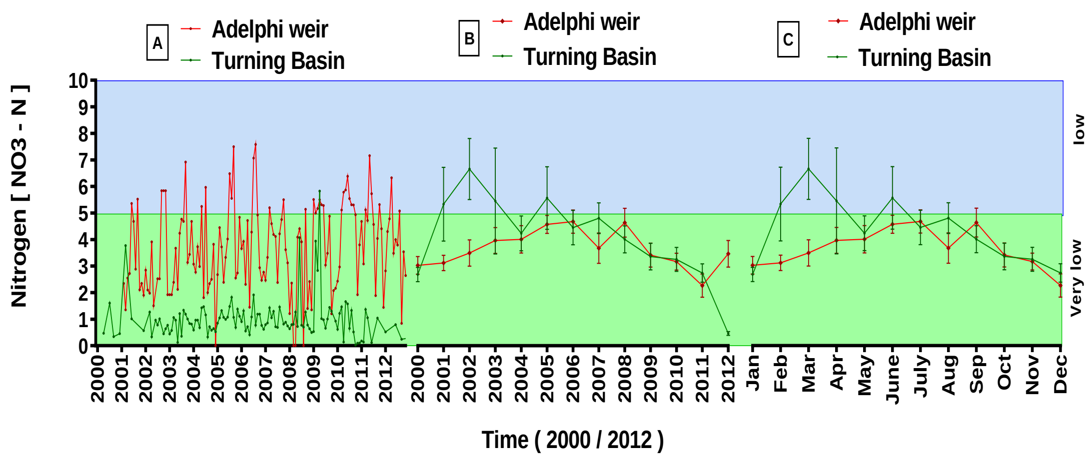

Figure 36- Historical changes in Nitrogen [NO_3–N] (A) (A) at Adelphi weir and Turning Basin [TB] between 2000 and 2012. (B) The annual mean of [NO_3–N] between 2000 and 2012. (C) Seasonal changes in Nitrogen [NO_3–N] from January to December during 2000 and 2013. The values of Adelphi weir and Manchester Ship Canal were shown as red and green lines, respectively. Values are the actual measurements taken by EA and APEM.

Statistical analysis showed significant differences between the upstream site at Adelphi weir and downstream site at Turning Basin along Manchester ship Canal, figure38.

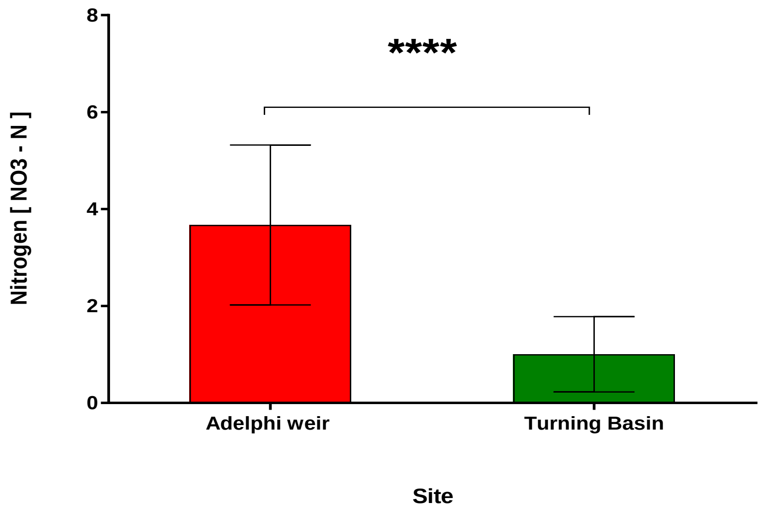

Figure 37- Mean nitrogen [NO_3–N], 2000 and 2012 at Adelphi Weir and the Turning Basin. $P < 0.05$ shows there were significant differences between sites (n=2). Errors bar ± SEM.

The historical data showed that phosphate concentrations varied greatly in time and space (Figure 39), ranging from ‘Very Low’ to ‘Moderate’ according to the nutrient GQA. On several occasions concentrations in the Turning Basin were classified as ‘High’ or ‘Very, to Excessively High’. Under the Carlson (1996) classification all sites would be classed as eutrophic or hypereutrophic. Overall, concentrations of PO_4-P were markedly higher in the MSC compared to Adelphi Weir (Figure39C, Figure 40). A largely continuous decline in phosphate occurred through the whole period of the data-set (Figure 39) and the annual mean showed that total phosphorous declined from 0.8 mg/l in 2001 to nearly 0.2 mg/l in 2012, figure39B. Nevertheless, some readings were relatively high (up to 1.4 mg/l) in the first quarter of the last decade although most fluctuated around 0.2 mg/l upstream at Adelphi Weir. The majority of the concentrations of PO_4-P in the MSC were at the high end of the range, fluctuating between 0.4 mg/l and 1.4 mg/l mg/l from 2011 2006 where the trend declined steadily to just 0.4 mg/l from 2006 to 2013.

The monthly average PO_4-P indicates an increase during the summer, reaching more than 0.8 mg/l compared to early spring and winter where the values fluctuated around 0.4 mg/l, Figure 38C.

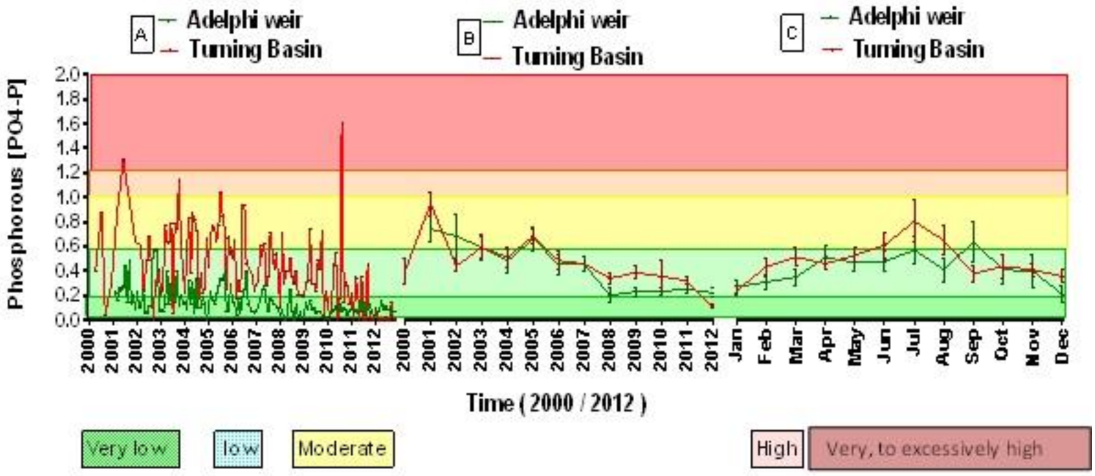

Figure 38- Historical changes in phosphorous [PO4-P]. (A) Phosphate [PO4-P] level at Adelphi weir and Turning Basin during 2000 and 2012. (B) The annual mean of [PO4-P] between 2000 and 2012. (C) Seasonal changes in Nitrogen [PO4-P] level of water from January to December during 2000 and 2013. The values of Adelphi weir and Manchester Ship Canal were shown as red and green lines, respectively. Values are the actual measurements taken on site by EA and APEM.

There was a significant relationship between phosphorous and nitrogen upstream at Adelphi Weir suggesting similar source or sources of the two nutrients, whereas no significant relationship was found between these nutrients downstream at the Turning Basin, Figure 41.

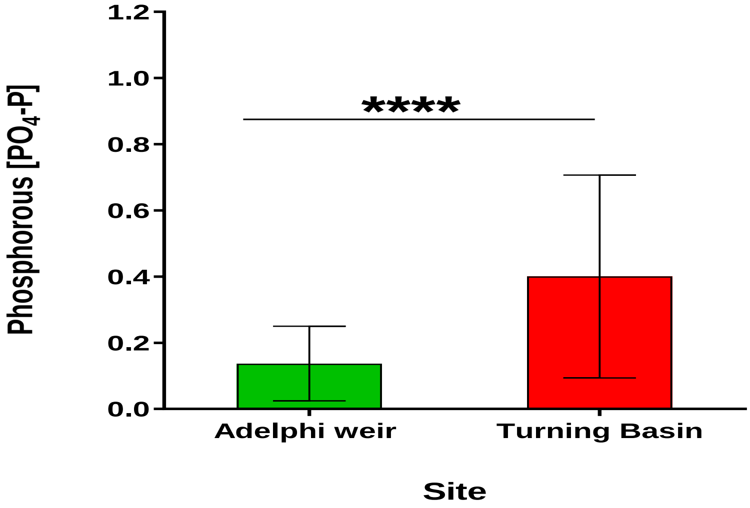

Figure 39- Statistical analysis of Phosphorous [PO4 – P] between 2000 and 2012 at Adelphi weir and the Turning Basin. P < 0.05) showed there were significant differences between sites. Errors bar ± SEM.

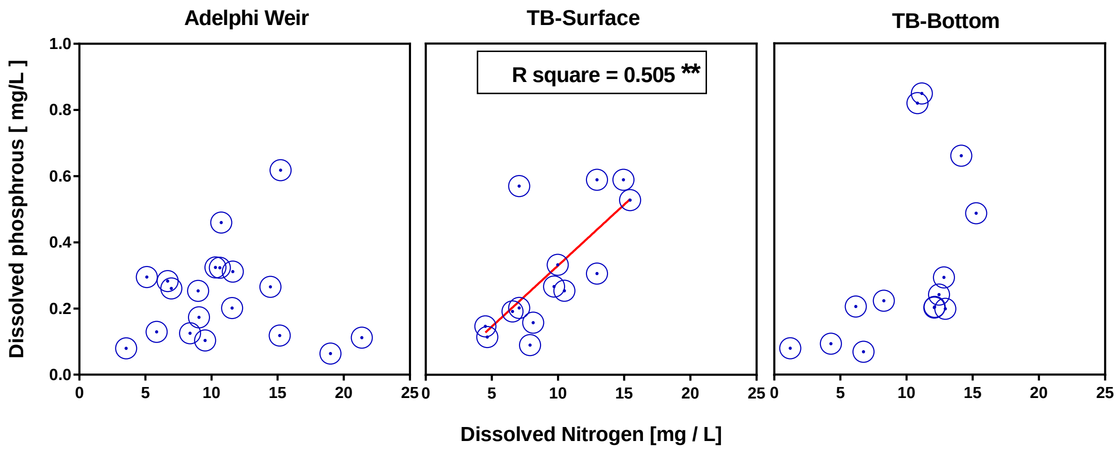

Figure 40- Linear Regression defining the relationship between nutrient components (nitrate [NO3 – N], and phosphate [PO4 – P]). Values presented in blue circles, R squared is 0.505 for Turning Basin surface showing slight positive relationship. Time period [2000 - 2012]. n = 12.

Ammonia levels varied between 0-5 mg/l in the Turning Basin, Figure 42A, and from 0-10 mg/l at Adelphi weir, figure42B. The trend was for a gradual decline over the whole period of the study from ‘Poor’ according to the Chemical GQA to ‘Good/Very Good’. However the frequent peaks, particularly between 2003 and 2009 suggests that there may have been some failure in the treatment of ammonia by the Sewage

Treatment Works (STWS) upstream of Adelphi Weir until 2009 that is reflected in elevated concentrations both at this site and downstream in the Turning Basin.

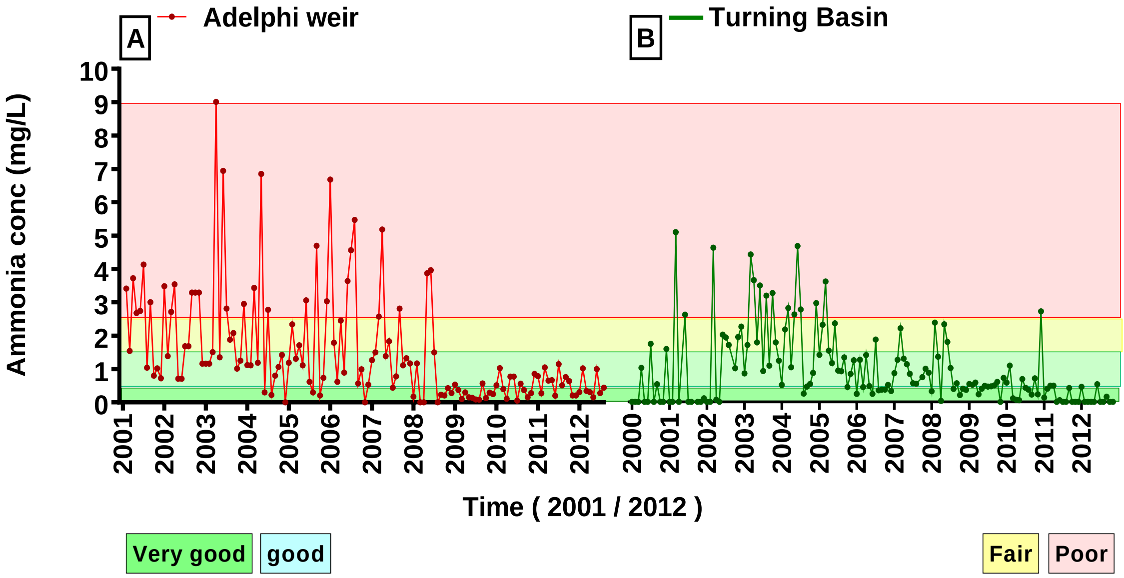

Figure 41- Historical changes in ammonia at (A) Adelphi Weir and the Turning Basin (B) between 2000 and 2012. Values are the actual measurements taken by EA and APEM.

The large overall decrease in ammonia with time is particularly apparent from the annual mean values at both sites, from 2.5 mg/l to just under 0.5 mg/l in the last three years of the study (Figure43A). The monthly mean values showed a slightly higher level of ammonia during the summer where the mean was around 2.5 mg/l. On the other hand during either winter or early spring, the mean fluctuated around 1 mg/l, figure43B). Statistical analysis showed that there were significant higher concentrations of ammonia at Adelphi Weir compared to the MSC (Figure44).

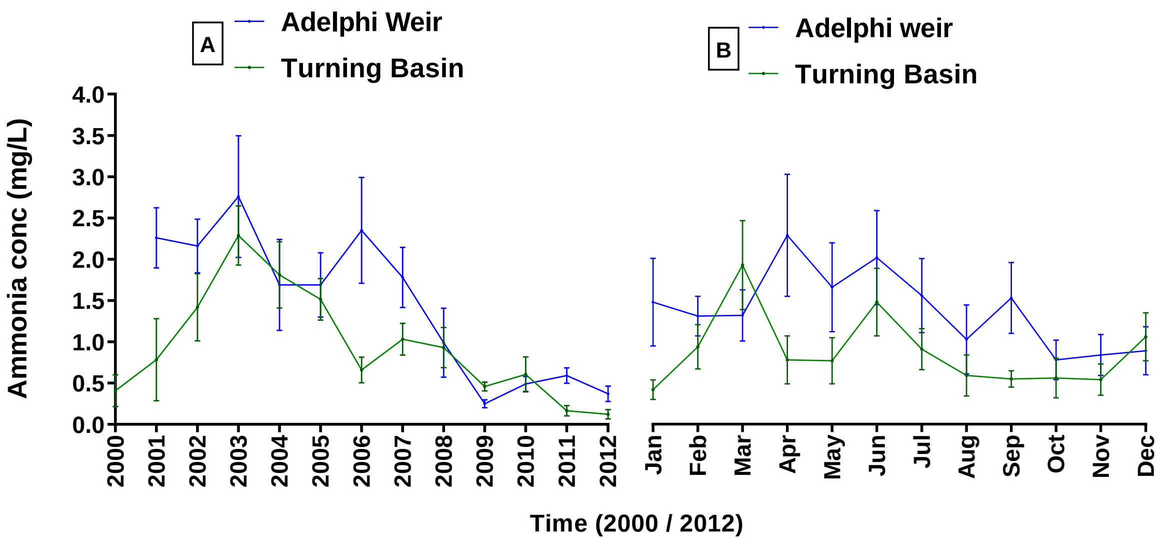

Figure 42- (A) the annual mean of Ammonia. (B) Seasonal changes of ammonia at Turning Basin and Adelphi weir between 2000 and 2012. Measurements described as red lines for Adelphi weir and green lines for Turning Basin. Values are the mean of a whole set of data provided by EA and APEM. Data shown as mean ±SEM. n=14.

With regard to the possible conversion of Ammonia to Nitrogen, there was no significant relationship between ammonia and nitrate at either ends of the system.

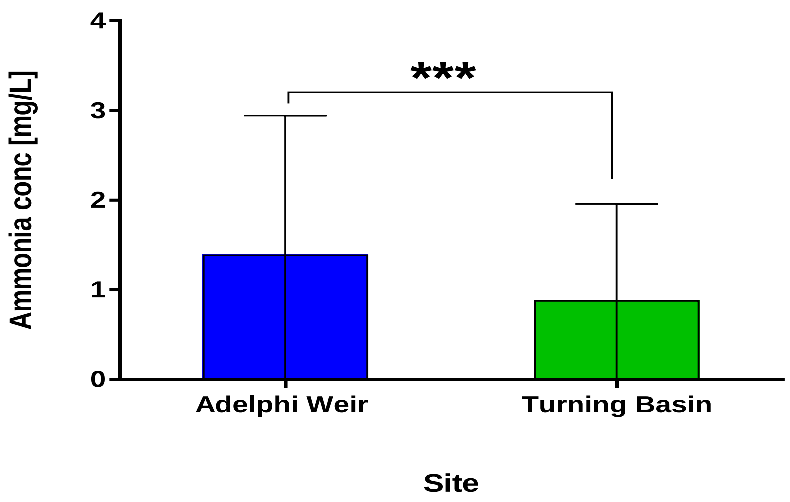

Figure 43- Mean ammonia concentration in the surface of the water column of the Turning Basin and at Adelphi Weir between 2000 and 2013. Showed There was a significant difference (P < 0.001) between sites (n=12). Error bar ± SEM.

3.1.4 Transparency and water clarity

The system is oligotrophic as most values during the period of study (2000 – 2012) fluctuated around 1m depth at both sites, figure45.

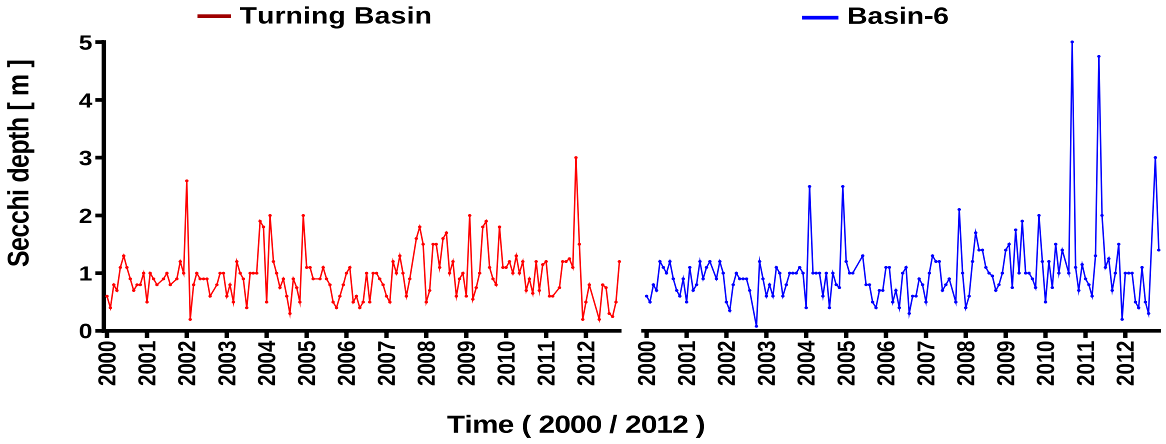

Figure 44- Historical changes in Secchi depth[m], 2000 and 2012 at (A) Measurement Turning Basin and (B) Basin-6. Values are the actual measurements taken on site by the EA and APEM.

The mean values of Secchi depth shows that water clarity at both sites are just over 1m from 2000 to 2008; afterwards transparency in the MSC decreased slightly from 2009 to 2011 where it can be seen that there was a sharp decline in 2012 when the values were just over 0.5 m, Figure 45. Secchi depth in Basin-6 also decreased in 2012 but only slightly and remained at around 1 m from 2008 until 2012. The site also showed larger variations over this period compared to the Turning Basin with periods of improved water clarity. Mean yearly values revealed a slight increase in transparency during spring whereas the system experienced a slight decrease during summer (Figure 46). Statistical analysis showed that there was no significant overall difference in Secchi depth between the Turning Basin and Basin6.

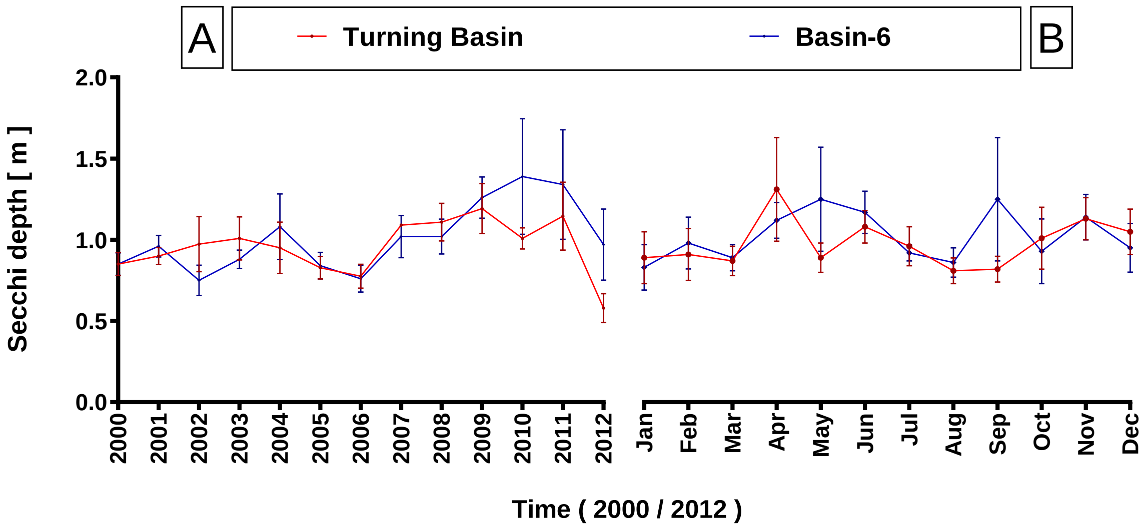

Figure 45- (A) the annual mean and (B) Monthly average of Secchi depth, 2000 and 2012. Data shown as mean ±SEM. n=14. Values are the mean of a whole set of data taken onsite and provided by the EA and APEM.

Total Suspended Solids (TSS)

Total suspended solids (TSS) were generally within the normal range of up to 20 mg/L (WFD standard) but showed regular increases at both sites that markedly exceeded this threshold, commonly reaching concentrations of 80mg/L (Figure 47) and with a maximum value of 436mg/L. Mean TSS concentrations were 17 mg/l, 708 mg/l, 16.5 mg/l for Adelphi weir, Turning Basin surface and Basin 6-surface respectively. The mean at the bottom of the water column was 20.6 mg/l and 16.5 mg/l at Turning Basin and Basin-6. Baseline TSS fluctuated between 5 – 10 mg at the surface in the MSC and Basin-6 and were relatively low compared to Adelphi Weir as the flow rate is slow; therefore, TSS showed larger fluctuations at this upper site, reaching the very high values greater than 400mg/l several time as shown in Figure 46A. In spite of the generally slow flow rate, we can see that TSS is relatively high at the bottom in both MSC and Basin-6 compared to the surface, where the values varied between 5 – 50 mg/l most of the time (Figure 47A, B).

The annual mean (Figure 48) shows no trend with time and TSS is mostly stable at the surface at all sites at around 10 mg/l. Bottom values fluctuated slightly around 20 gm/l most of the time. The peaks suggest that the level of TSS may be affected by high flow rate during high level of discharge or by run-off during high rainfall. In addition, dredging work takes place in the Turning Basin regularly upstream in the MSC.

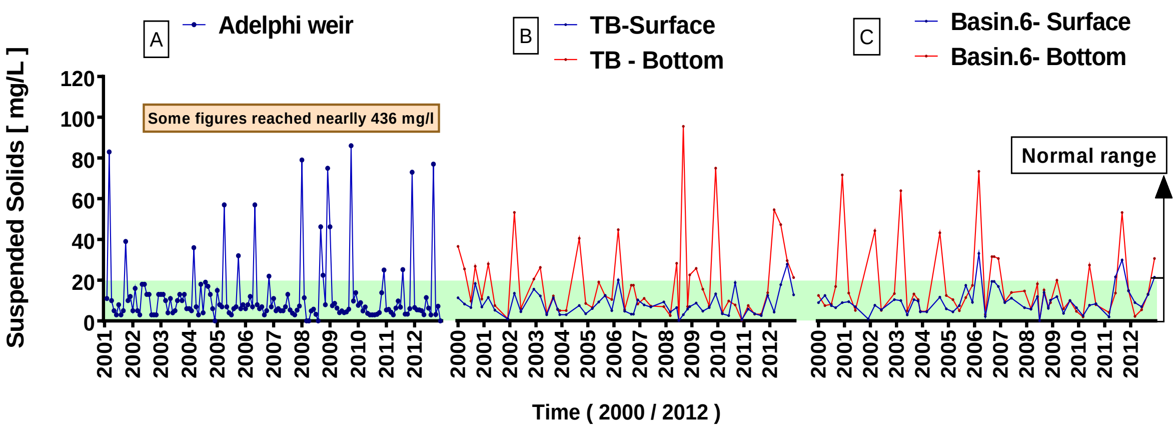

Figure 46- Historical changes in total suspended solids at (A) Adelphi Weir, (B) Turning Basin during 2000-2012. (C) Suspended solids level of water in Basin-6 during 2001 and 2013. The values of surface and bottom suspended solids were shown as blue and red lines, respectively. Values are the actual measurements taken by the EA and APEM.

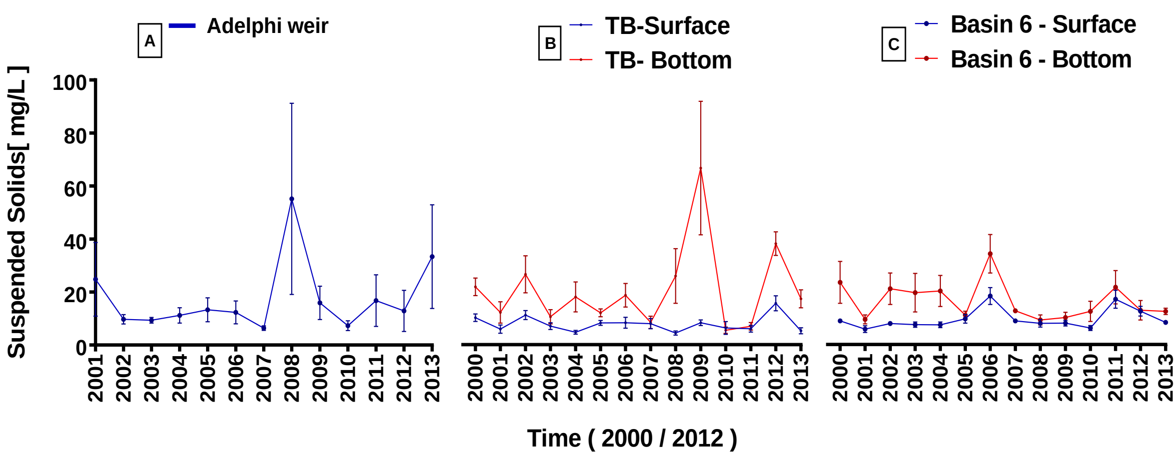

Figure 47- The annual mean of total spends solids, 2000 and 2012. (A) Adelphi Weir (B) Turning Basin [TB], and (C) Basin-6. Data shown as mean ±SEM. n=14. Values are a set of data provided by the EA and APEM.

There is an increase in the level of TSS during the autumn and winter at Adelphi Weir and at the bottom of the MSC and Basin-6. Generally, there was no specific trend in suspended solids at any site or depth in the water column during the different seasons, Figure 49. Note that the very large peak in TSS at Adelphi Weir during January, and it is indicated by the very large error bars (figure49C). Relatively low TSS was observed between June and August and may reflect low rainfall during these months during 2000 to 2012.

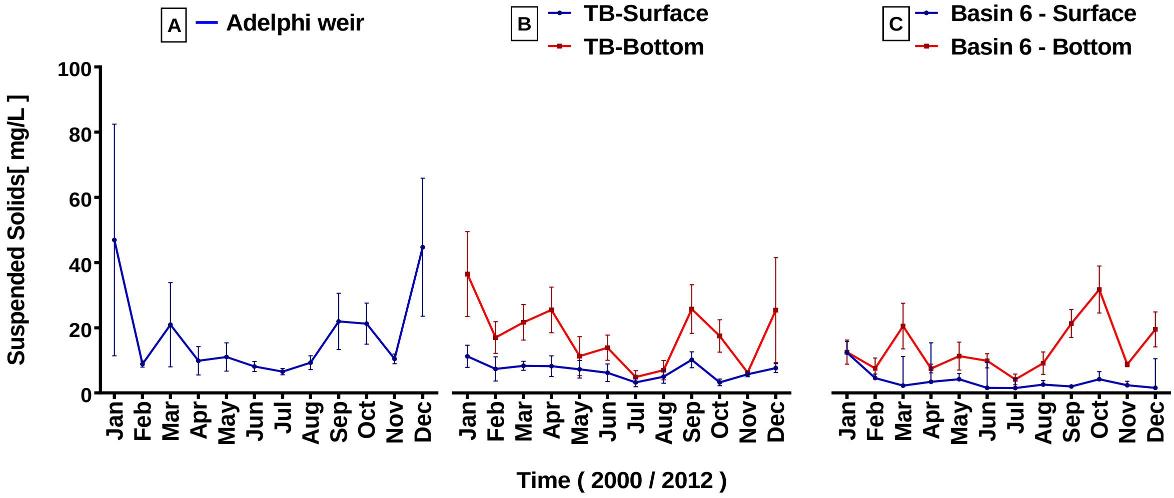

Figure 48- Seasonal changes of Suspended solids (TSS) between 2000 and 2012. (A)Adelphi weir, (B) Turning Basin [TB] (C) Basin-6 and. Data shown as mean± SEM. (n=14). Values are a set of data provided by the EA and APEM.

Statistics showed that there were slight but significant differences between surface and bottom at Turning Basin, and between Turning Basin surface and Basin-6 bottom, figure50.

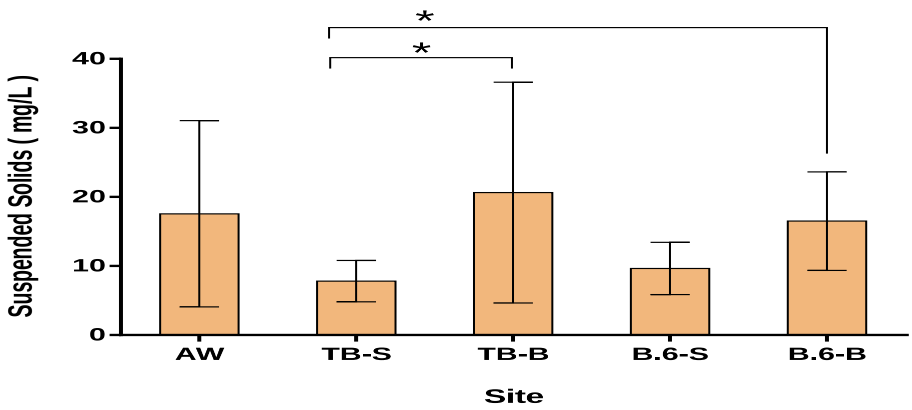

Figure 49- Seasonal average of suspended solids at Adelphi Weir [AW], Turning Basin surface [TB-S], Turning Basin bottom [TB-B], Basin.6-surface [B.6-S], Basin.6-bottom [B.6-B] between 2000 and 2013. Statistics (P>0.05) shows the difference between Turning Basin surface and bottom and between Turning Basin surface and Basin.6-B. Errors bar ± SEM. N=14.

The high concentrations and episodicity of suspended solids suggested that TSS is one of the most important and parameter in the Irwell/MSC. The relationship with transparency as measured by Secchi depth was therefore examined in the MSC and Basin-6, figure 51. It is clear that suspended solids are a significant barrier to light penetration at Manchester Ship Canal Turning Basin site,

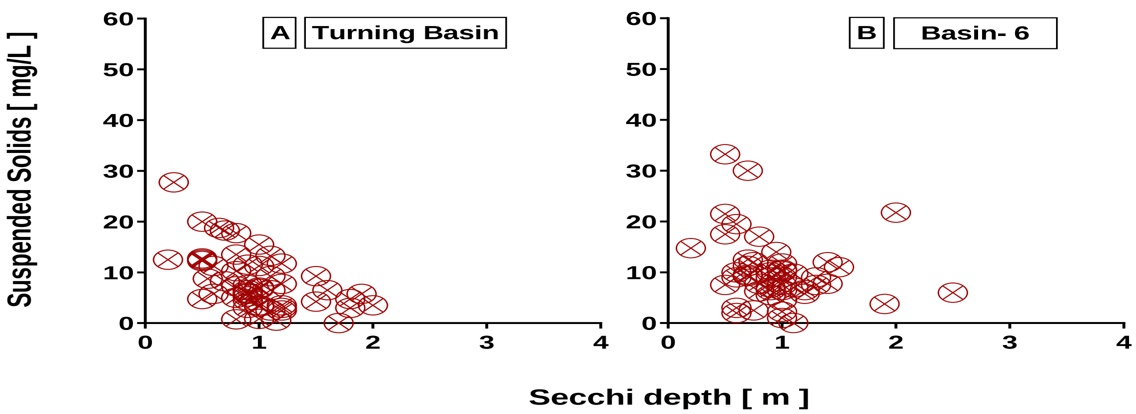

Figure 50- Relationship between suspended [TSS] (Correlation coefficient) and Secchi Depth in the Manchester Ship Canal Turning Basin (A) and Basin-6 (B) between 2000 and 2012. P value for A is <0.05 showing statistically significant between TSS and Secchi depth. [r^2=0.278] and p<0.05 for B indicate that there was no effect of TSS [r^2=0.027] on Secchi depth in Basin 6. n = 55.

In contrast to (TSS) there was no significant relationship between water clarity and chlorophyll-a concentration in the MSC or Basin-6, Figure 52**.**

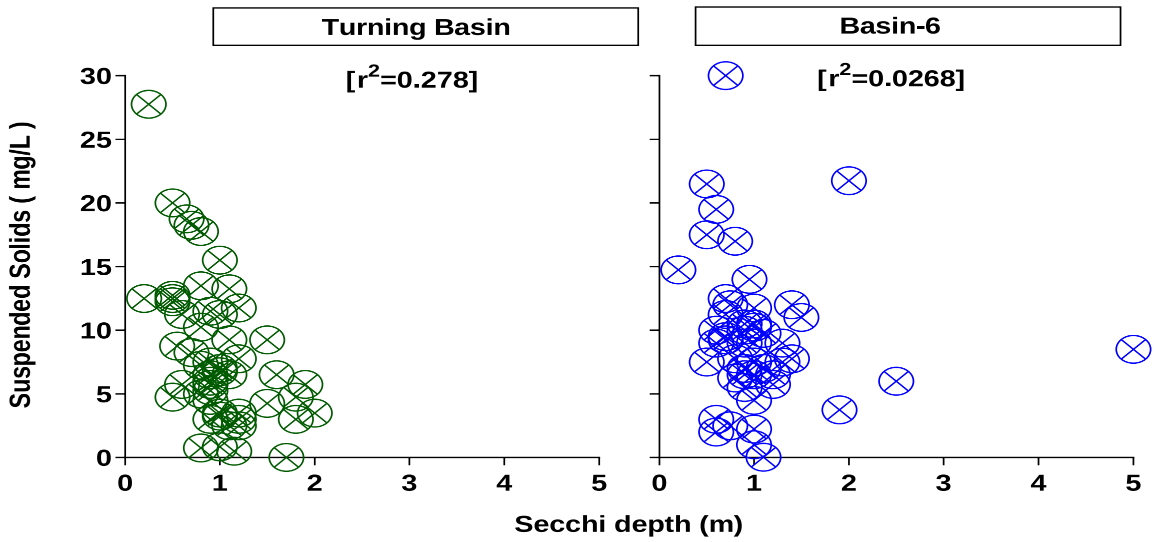

Figure 51- Relationship between chlorophyll-a concentration on water clarity (Secchi depth) in the Turning Basin [TB] and is Basin-6 between 2000 and 2012. p >0.05 indicates that there was significance effect in the TB [r^2=0.278] and Basin-6[r^2=0.0268]. N=55.

In summary possibly due to the effect of discharge inputs from the Irk and Medlock, the pH was relatively low at the upstream site with compared to the Turning Basin and Basin-6 but was still near-neutral and hence within the normal range. The slightly alkaline pH in the Turning Basin may be due to the shift in the carbonate-bicarbonate equilibrium arising from phytoplankton production. Temperature was in the normal seasonal range and hence does indicate any sources of thermal pollution. Conductivity was consistently high reflecting run-off of dissolved salts although there is some indication of s slight decrease over the present decade. DO was generally above 70% saturation and has shown a progressive increase with time but occasional long values of below 50% were noted, particularly although not exclusively in the Turning Basin and the open dock (Basin 6). The sometimes-low DO may reflect the often relatively high BOD, particularly at the lower two sites which are often designated as ‘Fairly Good’ to ‘Fair’ under the Chemical GQA. The overall increase in DO may reflect an improvement in BOD. There was little evidence of stratification at the lower two sites in the basis of differences in temperature or DO. Ammonia levels were low throughout the study and decreased with time. Decreases in BOD and ammonia both point to an improvement in the wastewater treatment Works (WwTWs) upstream of Adelphi Weir. Although overall total suspended solids where classified as low a number peaks where observed, including very high values of over 400mg/L. Moreover, and as will be discussed further in Section 5.4, this standard is negotiable and probably will be amended in the very close future as recommended by the recent researches carried out by the EA. Nevertheless, the system is subject to episodes of high suspended solids, presumably due to bed resuspension and episodic discharges from point sources and run-off. The high TSS are a key contributor to the low Secchi depth in the deep lower site on the MSC plus the associated open dock basin, and also probably also account to the lack of a relationship between Secchi depth and chlorophyll-a at these sites. A significant proportion of the TSS was of organic origin pointing to anthropogenic source or sources. Nitrate was classed as low although concentrations of phosphate were high, classifying all sites as eutrophic or hypereutrophic despite the marked improvement in phosphate contamination over the study period. Higher concentrations of phosphate were present in the Turning basin compared to Adelphi Weir which may indicate mobilisation from the sediments whereas nitrate levels were lower.

3.1.6 Biological parameters

Phytoplankton community

Phytoplankton distribution for the most major groups for the MSC and Basin-6 is shown in Figure 53. No data was available from the EA for Adelphi Weir and this is likely because the resident phytoplankton community will be very low due to the high flow and slow water depth. It is noticeable that total diatoms and chlorophytes are the most common groups within the two sites. Note that total chlorophytes are significantly lower in Basin-6 where the numbers for most of the period of study were below 100/L whereas reached nearly 1400 individuals/L in the Turning Basin. Numbers of Cyanobacteria for most of the time fluctuated and remained below 40/L at both sites. Mean numbers of phytoplankton were highly variable over the study period with peaks in 2004, 2005 and 2006 depending on the group.

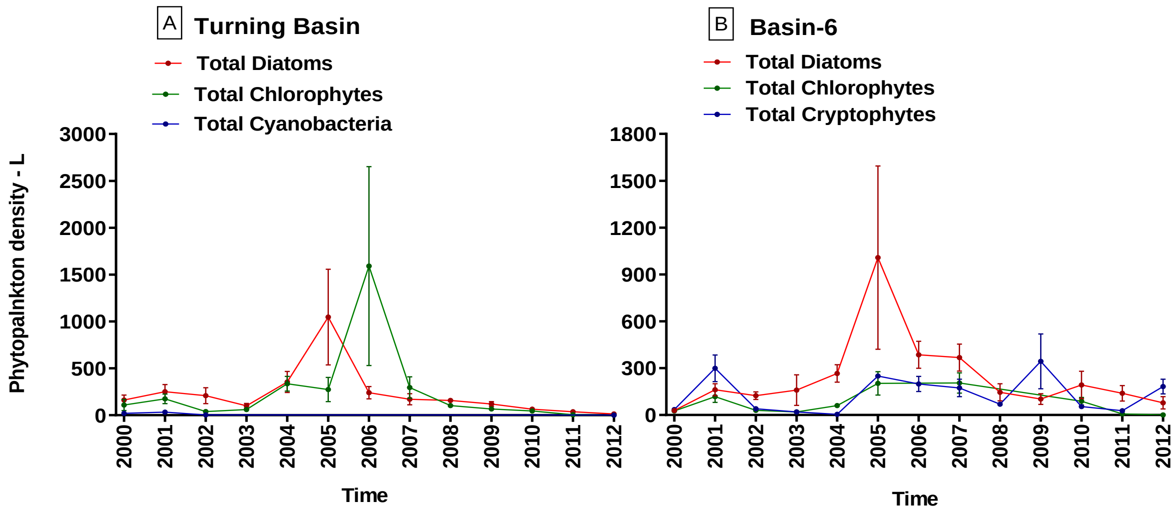

Figure 52- Mean densities of the most important phytoplankton populations in the Turning Basin (A) and Basin-6 (B) between 2000 and 2012. Data shown as mean ± SE, n = 14.

As expected most genera of diatoms showed seasonal changes, specifically *Synedra sp*, *Navicula* sp, *Oscillatoria sp* and *Closterium sp,* Figure 54. Generally, the largest numbers were present in spring although cyanobacteria were relatively common throughout the whole season in the Turning Basin. *Synedra sp* was found at slightly higher numbers in the MSC and reached almost 100/L; however, *Navicula sp* were present at even higher numbers of around 500/L in Basin 6. *Closterium sp* was found is almost exactly the same number at the previous site as *Navicula sp* but it was markedly lower in the MSC at just under 20/L. The cyanobacterium *Osciallotoria sp* was low at both sites where it fluctuated at just under 20/L. There were significance differences between the MSC and Basin 6 in terms of phytoplankton population distribution; for instance, *Synedra* and *Osciallotoria* were common in the MSC, reaching between 40 and 80 /L during summer whereas *Navicula* and *Closterium* were most common in Basin 6 with average numbers between 200 and 400 /L during a short period of time in early spring.

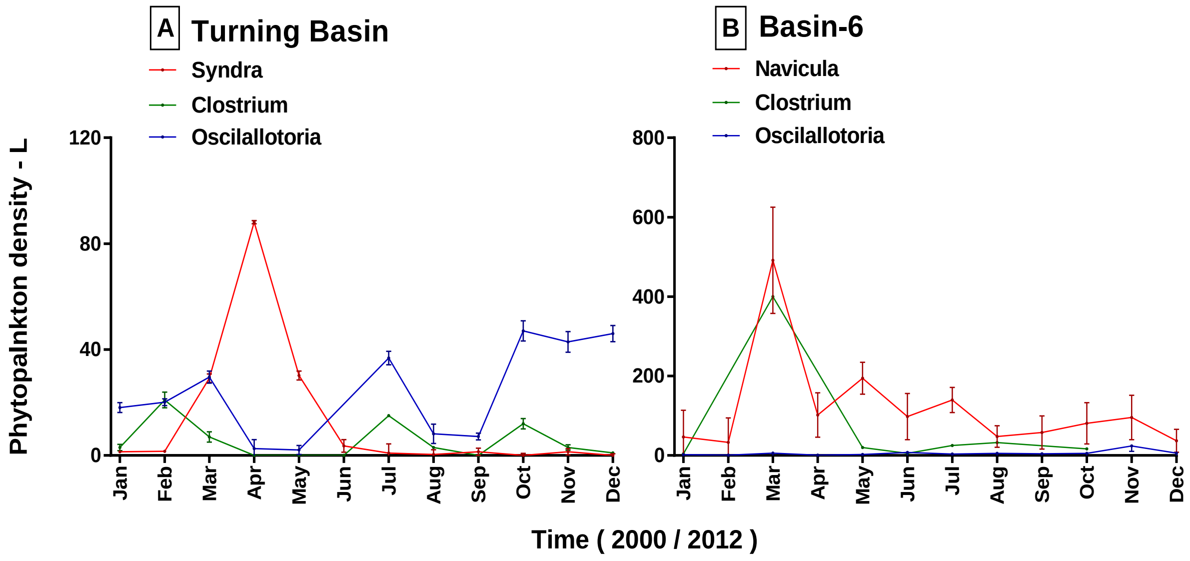

Figure 53- Monthly mean of the most common diatoms in the Turning Basin (A) and Basin-6 between (B) 2000 and 2012. Data shown as mean ± SE, n = 14.

Statistical analysis showed no significant variation between two sites in terms of total diatoms, whereas there was a significant variation between MSC and Basin 6 with regard to Cyanobacteria later in autumn and early winter, specifically around November, and total in chlorophytes in the middle of summer around July, Figure 55.

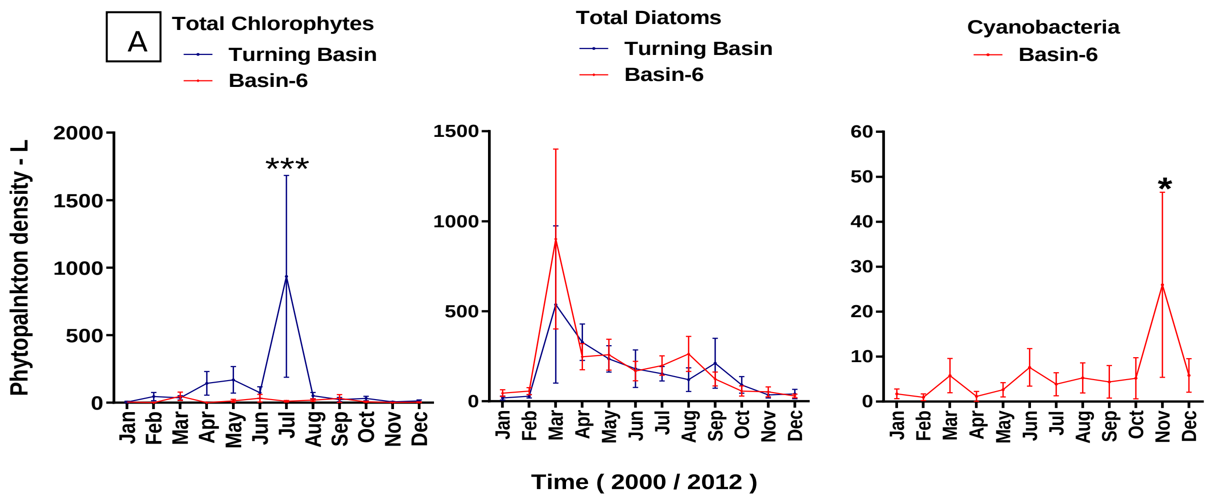

Figure 54- The monthly mean of the most common phytoplankton groups in the Turning Basin (A) and Basin-6 (B) between 2000 and 2012. p<0.05 indicates significant difference between Turning Basin and Basin-6 for total Chlorophytes only. Data shown as mean ± SE, n = 14.

Table 13 shows the TSI index from the long-term survey data at the Turning Basin site between 2000 and 2012. The average TBI was between 60 and 70 and with a mean of 67.9 indicating that the system is eutrophic with potential for blue-green algae dominance and that surface algal scums are possible (Carlson, 1996). In 2001 and 2005 scores above 70 indicates hypereutrophy and hence the potential for heavy phytoplankton blooms throughout the summer. As alluded to above, that such blooms are not in fact observed despite the high concentrations of phosphate is probably due to the high TSS entering the system. Thus, the trophic status indicated by chlorophyll concentrations is lower at an average of 51.9 with only one year exceeding 60 whereas the phosphorus TSI is commonly >80.

Table 13: TSI index from long term survey data at the Turning Basin site between 2000 and 2012.

	TSI-P	TSI-Chlorophyll	TSI-SS	Average
2000	90.46	51.86	62.34	68.22
2001	102.96	62.68	61.51	75.72
2002	91.83	50.34	60.39	67.52
2003	96.10	46.04	59.88	67.34
2004	93.96	48.48	60.73	67.72
2005	98.19	51.96	62.73	70.96
2006	93.25	50.08	63.67	69.00
2007	92.34	47.18	58.74	66.09
2008	88.12	52.05	58.51	66.23
2009	89.92	50.93	57.47	66.11
2010	88.78	52.49	59.88	67.05
2011	87.23	56.39	58.04	67.22
2012	71.76	51.34	67.84	63.65
Average	91.15	51.68	60.90	67.91

In summary, a diverse phytoplankton community is present in the deeper sites, consisting of diatoms, chlorophytes and cyanobacteria. Seasonal fluctuations occur although it is unclear to what extend these are influenced by predation ('top-down' control') by zooplankton as these were not recorded by APEM Ltd. 'Bottom-up control by nutrient limitation is unlikely in this eutrophic system. The presence of cyanobacteria is of concern given their potentially toxic nature. It is apparent the suspended solids are a key control on phytoplankton productivity and such control has important implications for the future management, as will be discussed later.

Benthic invertebrates

Benthic invertebrates are an important indicator of water quality. Indeed, a lack of benthic invertebrate diversity and a high level of dominance commonly indicates poor water quality. In this study some taxa fluctuated between 500 - 4000 individuals per colonization sampler, Figure 56, resulting in a high degree of dominance. Biological diversity in terms of benthic invertebrate taxa is very low in the system. The most common taxa were the isopod crustacean *Asellus aquaticus;* Oligochaeta (worms), midges of the family Chironomidae, the leeches *Helobdella stagnalis* and *Erpobdella sp,* and the amphipod crustacean *Gammarus sp*. *Asellus aquaticus* Oligochaeta and Chironomidae were found in high densities at both sites where they fluctuated around 500 and 1000 individuals per colonization sampler in Basin-6 and MSC respectively over the period of study 2000 to 2007. *Gammarus sp, Erpobdella sp* and *Helobdella stagnalis* were found in low numbers and fluctuated around 100 and 20 individuals per colonization sampler at the two sites.

Generally, benthic invertebrates were more common and reached high densities in Basin-6 compared to the MSC Turning Basin site. In addition, there were significant differences between *Asellus aquaticus* and other species in the MSC with an average of 700 per colonization sampler, Figure 57. The other common taxa were present at average densities of less than 100 individuals per colonization sampler. *Asellus aquaticus* occurred at an average density of less than 400 individuals in Basin 6 and the other common taxa were relatively high with an average of 200 per colonization sampler respectively. Thus, the Basin-6 invertebrate community is much more diverse and with a higher population density than the MSC, especially during the summer months.

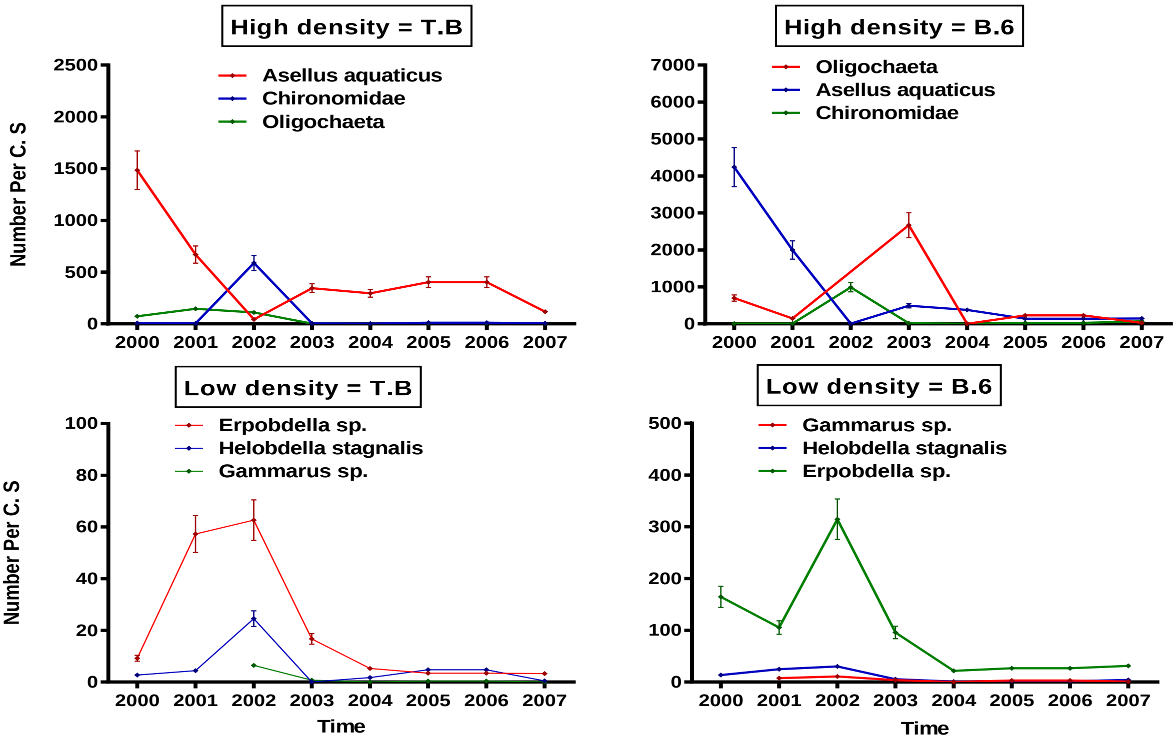

Figure 55- Number per colonisation sampler of major high density benthic invertebrate taxa (*Asellus aquaticus, Chironomidae* and *Oligochaeta*) and low-density taxa (*Helobdella stagnalis, Erpobdella sp* and *Gammarus sp*) in the Turning Basin [T.B] and Basin-6 [B.6] between 2000 and 2007. The first two graphs represent high density populations and the lower two represent low density populations. Data shown as mean ± SE, n=8.

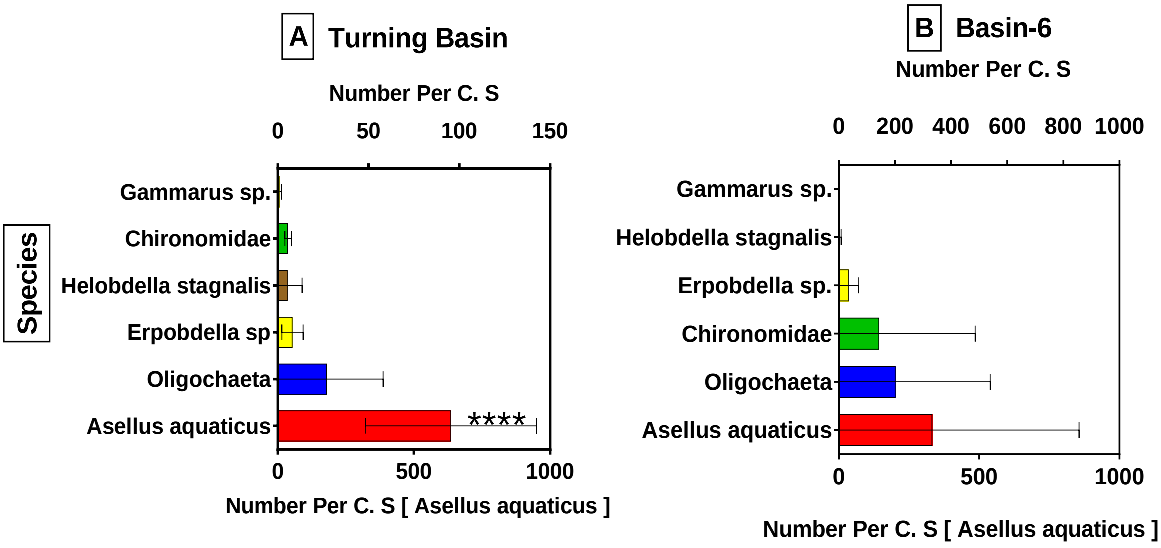

Figure 56- Average density of benthic invertebrates in colonisation sampler between 2000 and 2013 in the Turning Basin (A) and Basin-6 (B). The asterisk indicates significant difference between taxa. Asellus aquaticus values described on y axis (0 to 1000), p<0.05 showed significant difference at MSC. Data shown as mean ± SE, n=8.

There was wide degree of variation in benthic invertebrate density according to the season; changes in the high density genera are illustrated in Figure 58, and low density genera is shown in Figure 59 from the colonisation sampler located in the MSC Turning Basin and Basin 6. It is very clear that *Asellus aquatus* density is very high with an average of nearly 1000 individuals per sampler in the Turning Basin and an average between 2000-1000 in Basin-6 between January and May with a peak that reached more than 3000 individuals in March. The trend remains the same at both sites between August and December. With regard to Oligochaeta, although the average was more than 2000 in January, the general trend was for a decline and low numbers over the rest of the study period with densities of less than 200 individuals per colonisation sampler and with numbers never exceeding 500 per sampler. Numbers of *Erpobdella* increased in Basin-6 between January and May where the densities reached more than 200 per colonised sampler; thereafter numbers declined rapidly reaching as low 5 individuals in early winter. The low-density genera were Chironomidae, *Helobdella stagnalis* and *Gammarus sp*. Densities of *Gammarus sp* was very low at both sites while numbers of Chironomidae and *Helobdella stagnalis* was between 5 and 15 per colonisation sampler in the Turning Basin, and between 10 and 40 per sampler in Basin 6.

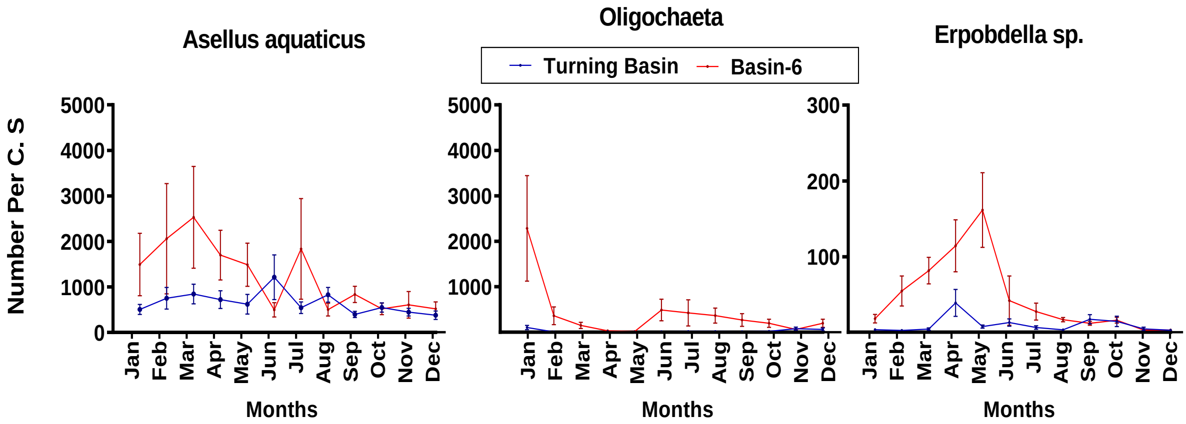

Figure 57- Seasonal changes in high density populations of the benthic invertebrates *Asellus aquaticus, Oligochaeta* and *Erpobdella* from 2000 and 2007. Data shown as mean ± SE, n=8.

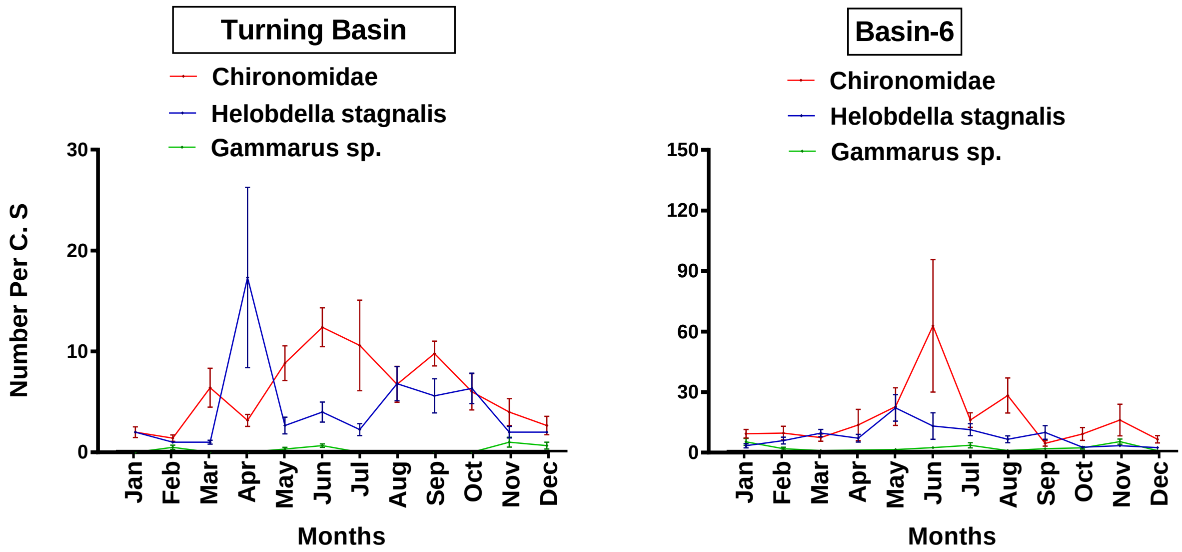

Figure 58- Seasonal changes in low density populations of benthic the invertebrates *Chironomidae, Helobdella stagnalis* and *Gammarus* from 2000 to 2007. Data shown as mean ± SE, n=8.

In summary, it is apparent from the benthic invertebrate community recorded in the colonisation samples that the system is subject to severe pollution stress characterised by a high degree of dominance and low diversity. None of the organisms score more than 6 according to the BMWP score and with the exception of Gammaridae only score between 1 and 3.

3.2 Seasonal changes in water quality

3.2.1 Discharge and flux

In common with the historical data, and for the reasons outlined above discharge at river Irwell, Irk and Medlock was obtained from data from Environment Agency. The values of upstream discharge were obtained directly from the gauging station at Adelphi weir, while the discharge downstream was predicted by adding the values of the three different rivers at the same point of time, Figure 60. The average discharge at Pomona Docks was double the upstream site at Adelphi weir and statistical analysis confirmed a significant increase in discharge between the two sites, reflecting the significant inputs from the rivers Irk and Medlock (Figure 61). A number of physical and chemical parameters where measured at a number of sites on the River Irwell specifically the Mark Addy, Regent Road Bridge, Pomona Docks and the Turing Basin between March 2013 and September 2014. Additionally, flux at each site was calculated for BOD, nutrients and total suspended solids. The location of each site is shown in figure 2 above and details can be found in Table 1.

3.2.1.1 Discharge and precipitation

Discharge at Pomona docks was slightly lower over the summer from the middle of spring of 2013 to September 2014. Discharge, then increased at both sites during the early autumn of 2013 and reached a peak of more than 15 m^3/sec. Afterward, discharge decreased gradually from November 2013 to early Spring of 2014 where the overall of discharge was just under 10 m^3/Sec at Adelphi weir and around 15 m^3/Sec at Pomona Docks during January 2014. The highest discharge was just over 20 m3/s at both sites during February, and after that fluctuated to around 5 m^3/sec between June and July 2014, Figure 60.

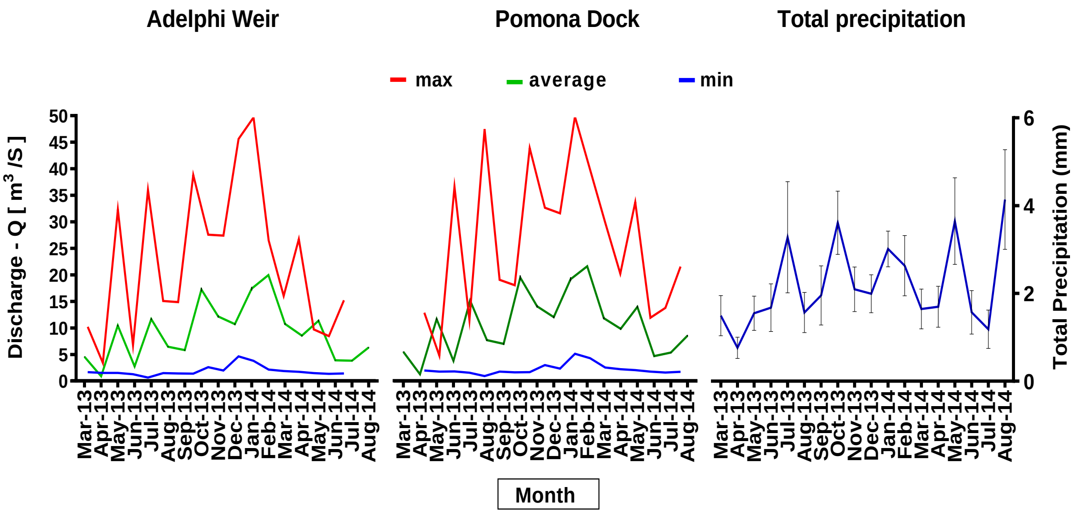

Figure 59 -Monthly average, maximum and minimum discharge at Adelphi Weir and Pomona Docks between March 2013 and August 2014. Discharge rates were recorded at fifteen minutes intervals from 1st March 2013 to 30th August 2014. Precipitation data shown as mean ± SE. P<0.05 showed that discharge at the two sites are statistically different, n=2978 for each Month.

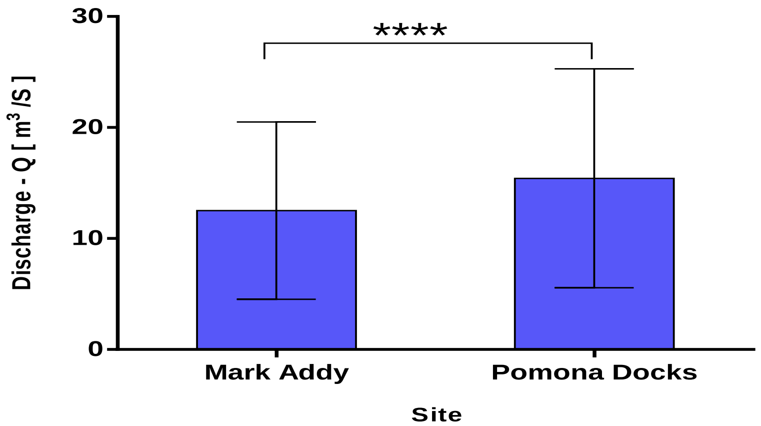

Figure 60 - Annual average and differences in discharge at Adelphi Weir and Pomona Docks between March 2013 and August 2014. P<0.05 show that the two sites are statistically different, n=2978 for each Month and the average data, n = 33.

Overall, discharge was higher during the seasonal survey than from the historical data. This cannot be due to differences in data source as the National Flow archive data was sourced in both cases.

3.2.1.2 Flux

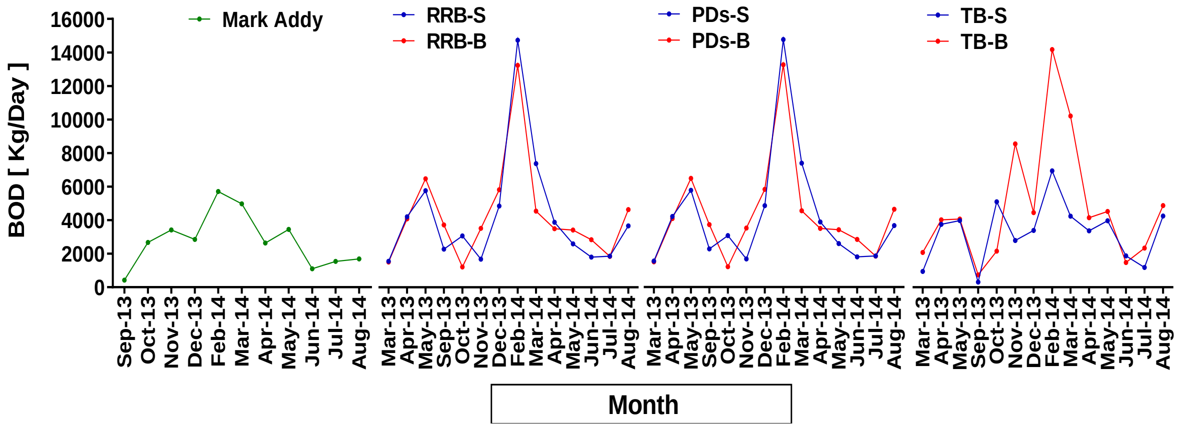

Figure 61- Monthly BOD flux at Mark Addy, Regent Road Bridge [RRB] surface and bottom, Pomona Docks [PDs] surface and bottom, Turning Basin [TB] surface and bottom between March 2013 and August 2014. The Mark Addy site was only analysed from September 2013. Correlation analysis showed R^2 values as 0.639, 0.501, 0.523, 0.453, 0.393, 0.503, 0.607 for Mark Addy, Regent Road Bridge surface and bottom, Pomona Docks surface and bottom, Turning Basin surface and bottom respectively. P<0.05 showed significance variation between all sites and different depths. N Mark Addy=11, RRB, PDs and TB = 14.

The average BOD flux was around 3000Kg/Day along the whole system, Figure 62. Generally, although the time scale is different between Mark Addy upstream where the measurements took place by September 2013 and the other three sites were measured from March 2013, the overall trend was an increase in BOD flux from September 2013 until February 2014 where it peaked, reaching more than 14000 Kg/Day but only at the downstream sites, not at Adelphi Weir where the peak was a more modest 6000 Kg/Day. After this time, BOD flux decreased significantly to just over 1000 Kg/Day in June 2014. Statistical analysis showed significant difference in flux of BOD at all sites. The overall seasonal BOD flux was higher than that estimated from the historical data due to the greater discharge.

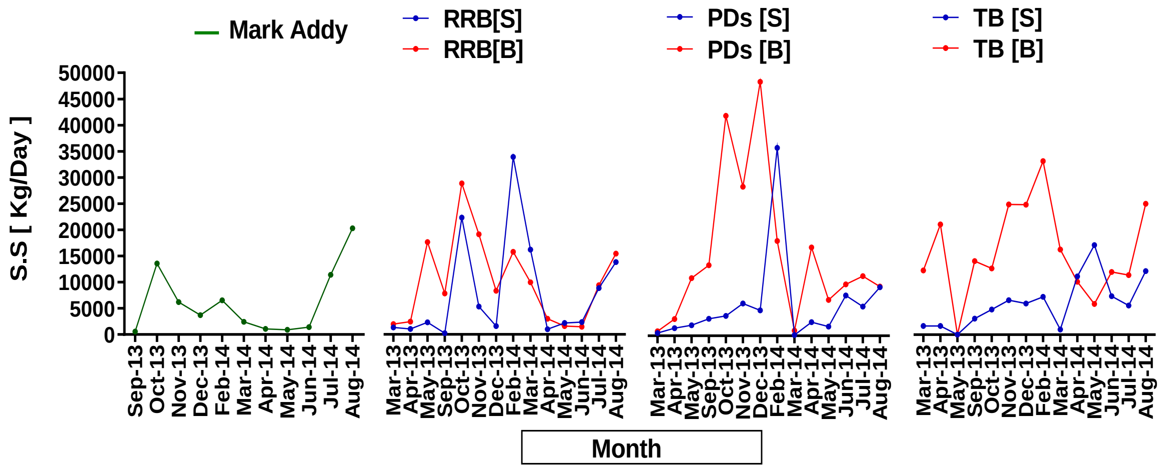

Figure 62- Monthly suspended solids flux at Mark Addy, Regent Road Bridge [RRB] surface and bottom, Pomona Docks [PDs] surface and bottom, Turning Basin [TB] surface and bottom between March 2013 and August 2014. R^2= 0.0264, 0.537, 0.302,0.458, 0.297, 0.0282, 0.351 for Mark Addy, Regent Road Bridge surface and bottom, Pomona Docks surface and bottom, Turning Basin surface and bottom respectively. P<0.05-Significant for RRB, PDs, TB at surface [**] and bottom [*]. n= 14

Suspended solids flux was on average lower upstream at the Mark Addy site and also at the surface downstream at Pomona Docks and the Turning basin.At Regent Road Bridge where the surface level of suspended solids was as high as at the bottom of the water column, with an average that is around 10000 Kg/Day, Figure 63. On the other hand, the level of suspended solids was much higher and changeable at the bottom at Pomona Docks and the Turning Basin with an average flux of 15000 Kg/Day. The more variable flux of suspended solids at the lower sites at Regent Road Bridge, Pomona Docks and the Turning Basin may reflect inputs from the tributaries and possible sediment resuspension and deposition. The overall SS flux was higher than that estimated from the historical data, once again, due to the greater discharge.

The flux of nitrogen as nitrate was generally slightly lower upstream at both Mark Addy and Regent Road Bridge, compared with downstream at Pomona Docks and the Turning Basin. Upstream average was below 5000 Kg/Day, and with only small changes with time. On the other side of the system, downstream, nitrogen was much higher and more changeable with an average more than 65000 Kg/Day and a range of 1000– 10000 Kg/Day (Figure 64). Nitrogen flux tended to be higher in the winter.

Phosphorous flux was under 2000 mg/sec at Mark Addy with little evidence of seasonality. The flux at all the downstream sites was more variable compared to Mark Addy for most of the time with an average below 200Kg/Day except at Regent Road Bridge which is below to the River Irk and were the flux of phosphorous was mostly over 200 Kg/Day. Statistical analysis showed a slight but significantly lower nitrogen flux only at Mark Addy. No spatial differences were recorded in phosphate flux.

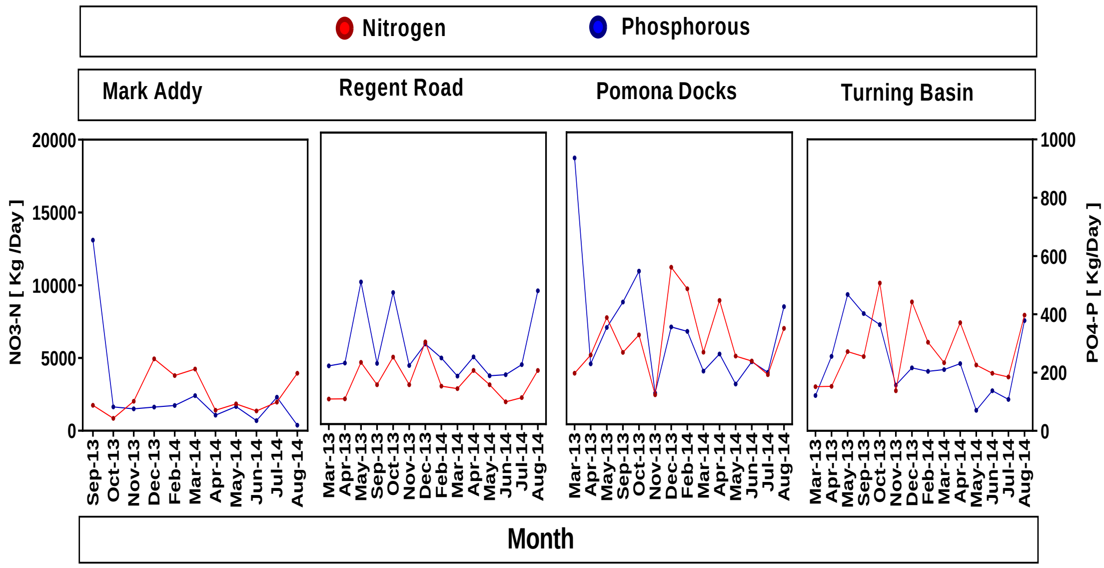

Figure 63- Seasonal nutrient flux at Mark Addy, Regent Road Bridge, Pomona and Turning Basin between March 2013 and August 2014. The R^2 were (Mark Addy N= 0.370, P= 0.0340), (Regent Road Bridge N=0.132, P-0.003), (Pomona Docks N=0.177, P=0.010), (Turning Basin N=0.147, P=0.003). Significant (P<0.05) difference was observed only for nitrogen at the Mark Addy compared to the other three sites [*]. n= 11 and the Mark Addy; all other sites are 14.

3.2.2 Seasonal changes in physical and chemical parameters

Physical and chemical parameters where measured at four sites on the system at the Mark Addy below Adelphi Weir to the Turning Basin on the MSC monthly between March 2013 and August 2014; see Figure 2, Table 1 above. In addition, some physio-chemical (Dissolved oxygen, pH, conductivity and temperature) parameters were measured at Adelphi Weir at three stations over the river width and at the same period of time. The cross-sectional survey was to examine the horizontal special variation of samples that were taken from near-side, middle and far side at the Mark Addy site. Near-side was defined as the side in which the river flows from left to right.

3.2.2.1 pH

The pH of the Irwell/MSC over the 18-month period of study from March 2013 to August 2014 was around near-neutral as it varied around 7.5 at all sites and at surface and bottom of the water column, (Figure 65).

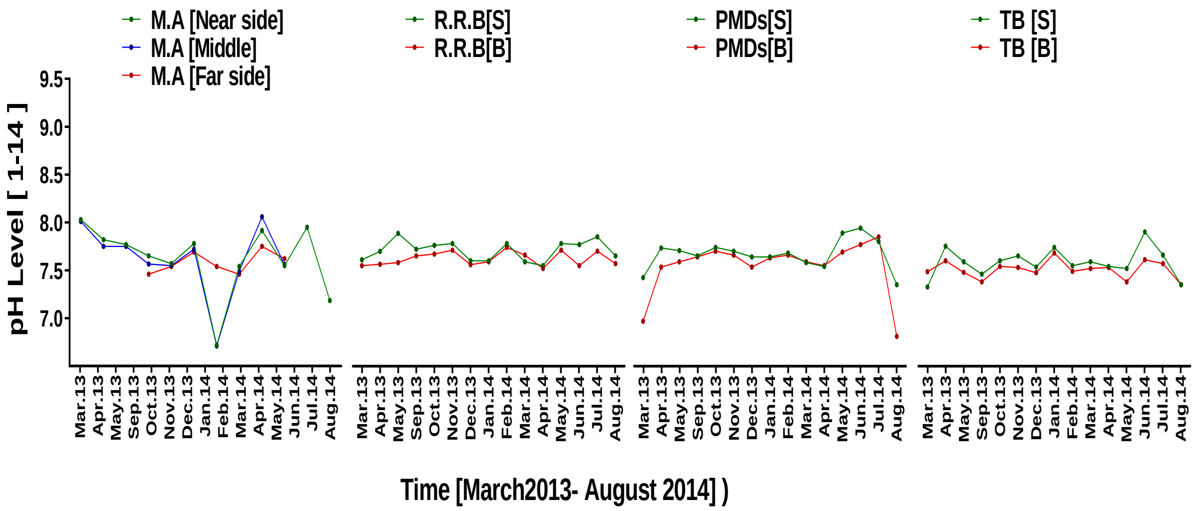

Figure 64- Seasonal changes in pH at Mark Addy [A], Reagent Road [B], Pomona Docks [C] and Turning Basin [D], between March 2013 - August 2014.S and B refer to surface and bottom of the water column respectively. n=13 for Mark Addy and 15 for other sites.

There is no significant difference in pH between the four sites; on the other hand, there were slight but significant differences with time for reasons that are unclear although in any case the changes in pH are very small (Figure 66).

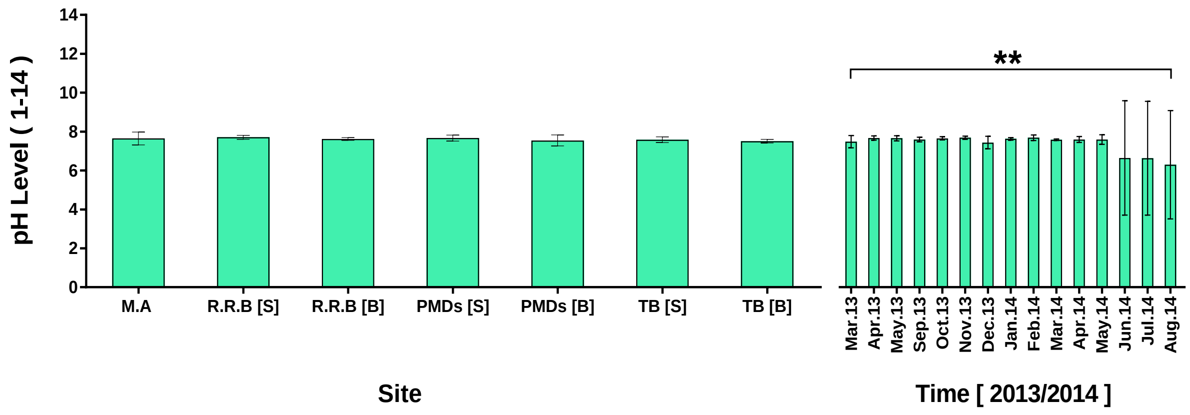

Figure 65- Mean pH at Mark Addy, Reagent Road, Pomona Docks and Turning Basin between March 2013 and August 2014. ANOVA showed no significant difference ($p<0.05$) between sites but significant ($P>0.05$) difference with time. Data shown as mean ± SE, n=15.

3.2.2.2 Temperature

The graphs show a typical UK water temperature that varies between 5^0C during winter and 20^0C during summer time, Figure 67. There is no stratification indicating that the whole system is well mixed as values for surface and bottom of the water column are nearly the same. Statistical analysis shows significant differences across the river upstream between Adelphi Weir and the Mark Addy (Figure 67A), As expected there was a marked increase in the summer (both 2013 and 2014) and the average temperature ranged from 5^0C minimum to 20 C^0 maximum in the whole system, and during the whole period of study from March 2013 to August 2014. In addition, there were noticeable differences of temperature during the spring and summer between the years 2013 and 2014, figure 68. For instance, the average of March 2013 was just under $4C^0$ where the average of March 2014 was over $12C^0$; also the difference was very clear compared to May 2013 as the average temperature was $7C^0$ while during May 2014 the average was just under $15C^0$.

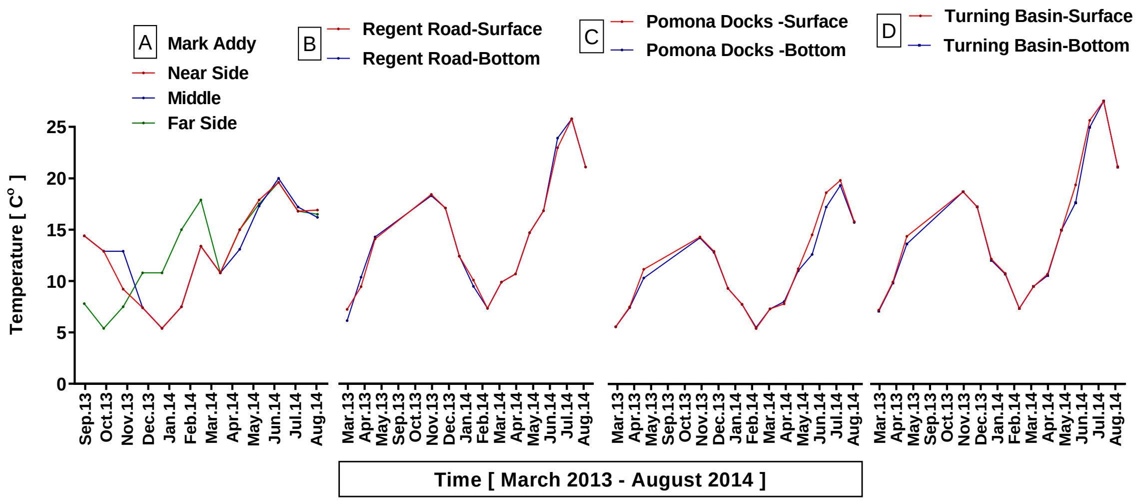

Figure 66- Seasonal changes in temperature (C^0) at the Mark Addy [A], Reagent Road [B], Pomona Docks [C] and Turning Basin [D], between March 2013 and August 2014 S and B refer to surface and bottom of the water column respectively. n=13 for Mark Addy and 15 for other sites.

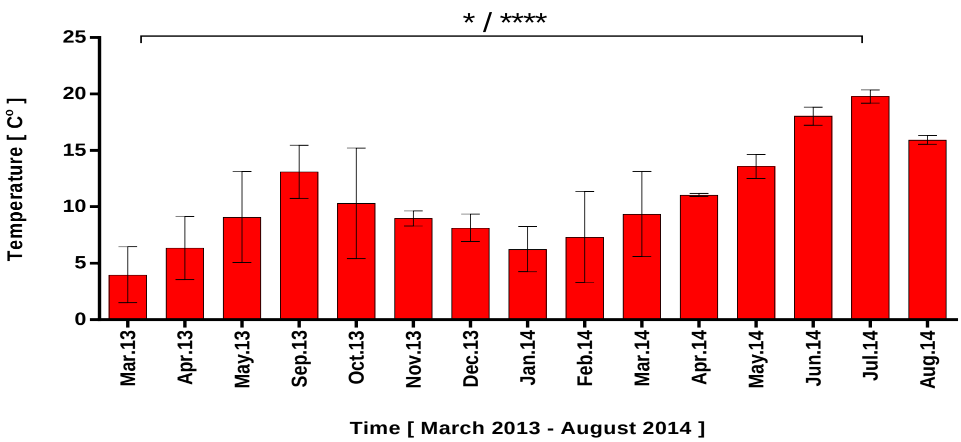

Figure 67- Average temperature (C^0) for all sites between March 2013 and August 2014. p>0.05 indicating significant variance over different seasons (ANOVA). Data shown as mean ± SE, n = 15.

3.2.2.3 Conductivity

Overall, conductivity was around 600 µcS/cm at all sites and at the two depths. Generally, the trend varied between 250 and very high values of 1250µcS/cm with time and with no significance variation between different sites, depth (Regent Road, Pomona Docks, Salford Quays) and across the river (Mark Addy). It is interesting that conductivity at the three downstream sites is very similar with peaks exceeding 1000 µcS/cm in October and December 2013 and with smaller peaks in March and May 2014, figure 69. In common with the historical data, conductivity was consistently very high, exceeding the normal range in unpolluted waterbodies of >70 - 250 µS according to the UK-WFD/2015.

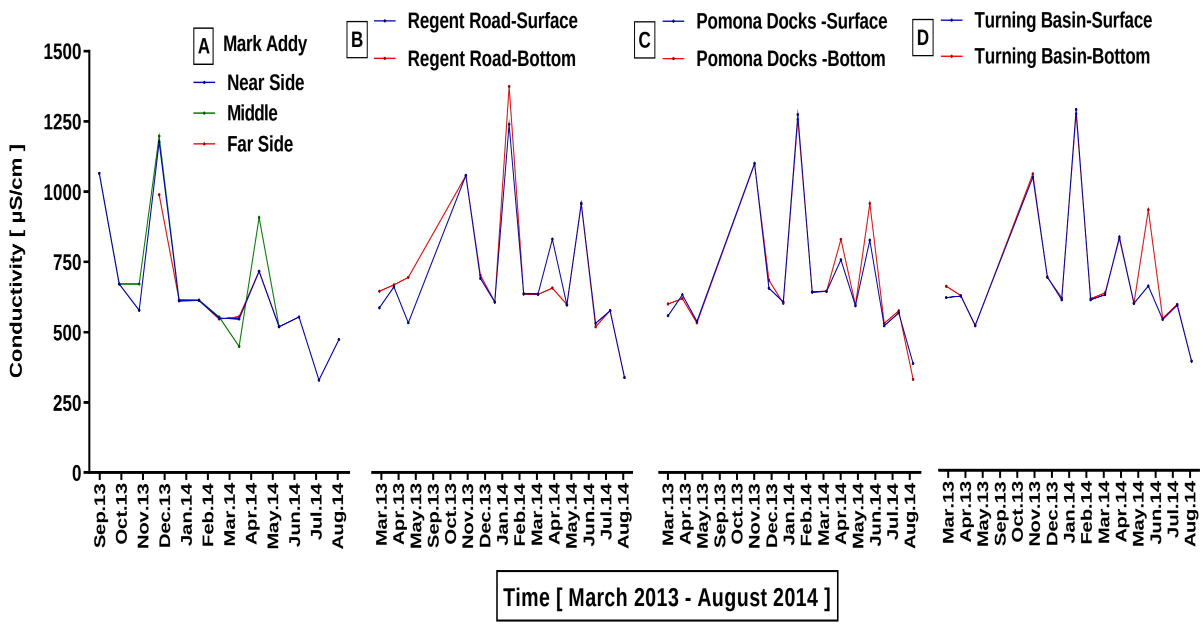

Figure 68- Seasonal changes of conductivity at Mark Addy [A], Reagent Road [B], Pomona Docks [C] and Turning Basin [D], between March 2013 - August 2014.S and B refer to surface and bottom of the water column respectively. n=13 for Mark Addy and 15 for other sites.

3.2.2.4 Dissolved oxygen (DO)

The Irwell/MSC is quite changeable between winter and summer at all sites with a seasonal difference in DO of 4 to 16 mg/l. DO at the upstream site at Adelphi Weir showed the water is reasonably well mixed across the river as only fluctuated by 6-14mg/l around the average of 10 mg/Figure 70. The trend downstream with season shows the level of DO reached 16mg/L during winter especially at Regent Road Bridge whereas there were a significant drop during summer as DO level was just over 8 mg/L except Salford Quays where the decline was more marked and fell to 2 - 4 mg/L at the bottom of the water column and the surface respectively, figure 70D. Statistical analysis (Figure 71) shows significant differences between sites and different seasons with the lowest overall level of DO recorded at Turning Basin bottom. DO at the bottom of the water column was also significantly lower than at the top sites at Mark Addy and Regent Road Bridge.

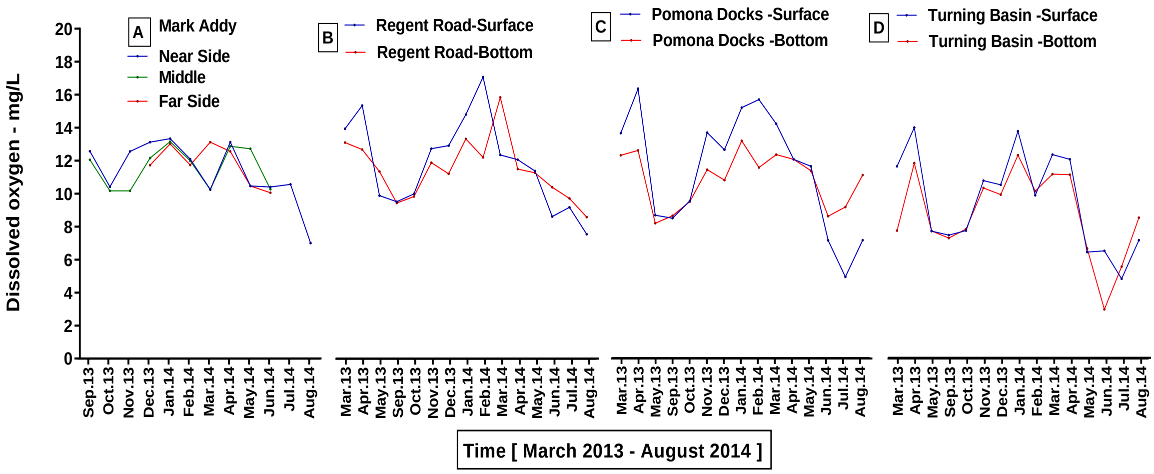

Figure 69- Seasonal changes of Dissolved oxygen (DO) at Mark Addy [A], Reagent Road [B], Pomona Docks [C] and Turning Basin [D], between March 2013 and August 2014.S and B refer to surface and bottom of the water column respectively. n=13 for Mark Addy and 15 for other sites.

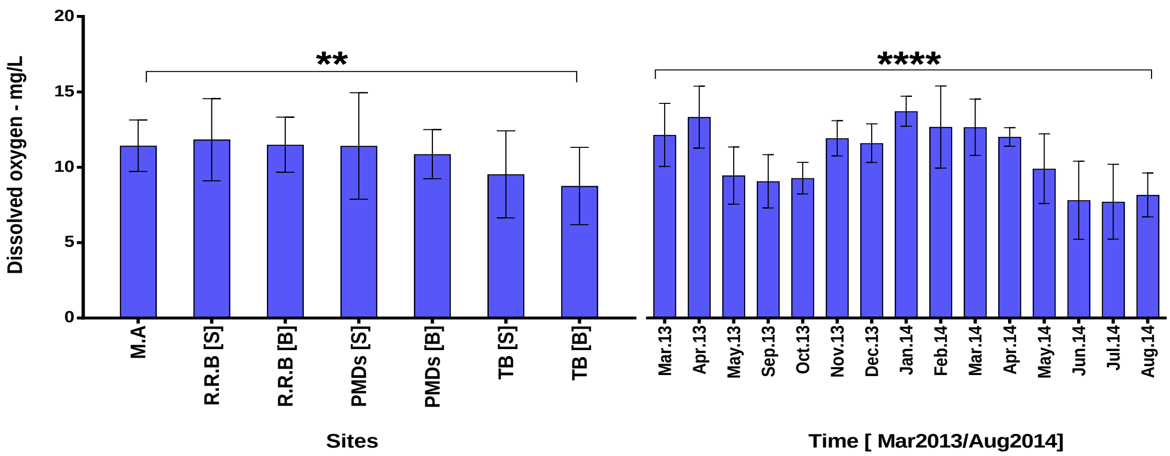

Figure 70- Mean dissolved oxygen [DO] Mark Addy, Reagent Road, Pomona Docks and Turning Basin between March 2013 and August 2014. ANOVA showed significant differences (p>0.05) between sites (P>0.01) and with time (p>0.0001). Data shown as mean ± SE, n=15.

The average BOD between 2013 and 2014 varied generally from 2mg/l to 6mg/l with some peaks at around 8 mg/l in early spring of 2013, figure 72B, C and D. The average BOD at all sites showed that the trend mostly within an accepted range (GQA = 4mg/l; 'Good') between March 2013 and August 2014 with short periods where the BOD declined to levels which are considered as very good according to the GQA and WFD criteria, Figure 72A. BOD is similar to the historical levels recorded during the latter part of the study in 2012. There are statistically significant differences between time and seasonal average during early spring between March and April 2013; also between autumn 2013 and early spring of 2014, Figure 73. Changes in BOD were less marked at the Mark Addy site than downstream where all other sites showed larger fluctuations in BOD with peaks in April 2013, July 2013 and February 2014. BOD at the surface and bottom were similar at all downstream sites except from December 2013 until June 2014 when the BOD of the bottom water was higher at the Turning Basin (compare figure 73B, C and D) this difference is consistent with the historical data from this site.

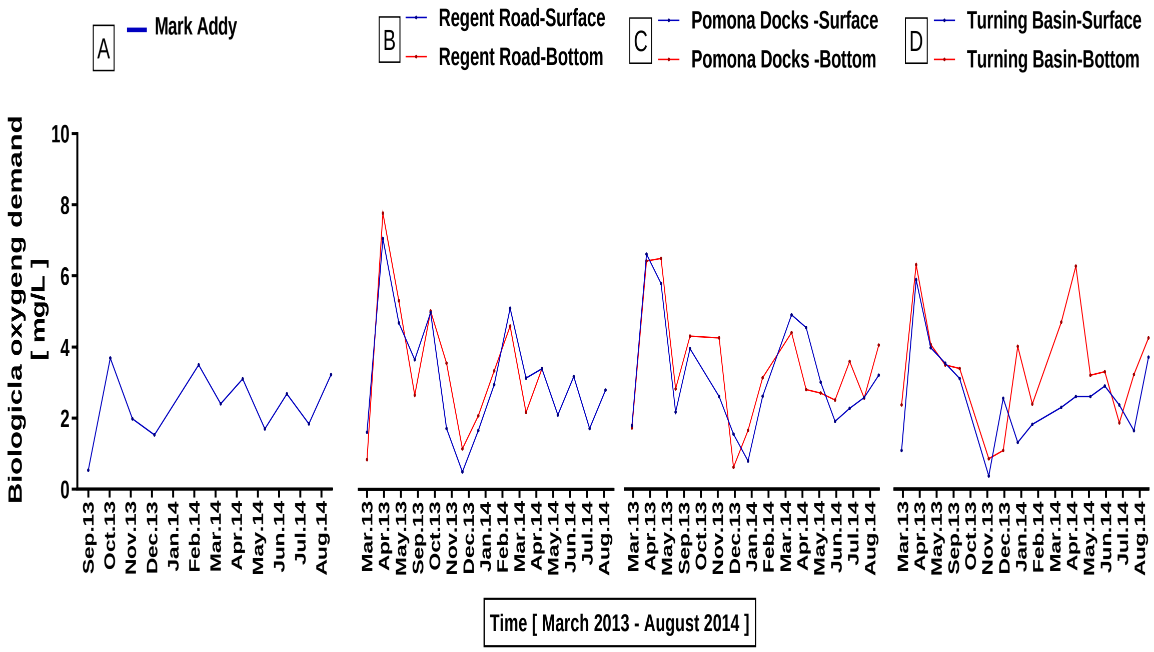

Figure 71- Seasonal changes of Biological Oxygen Demand (BOD) at Mark Addy [A], Reagent Road [B], Pomona Docks [C] and Turning Basin [D], between March 2013 and August 2014.S and B refer to surface and bottom of the water column respectively. n=13 for Mark Addy and 15 for other sites.

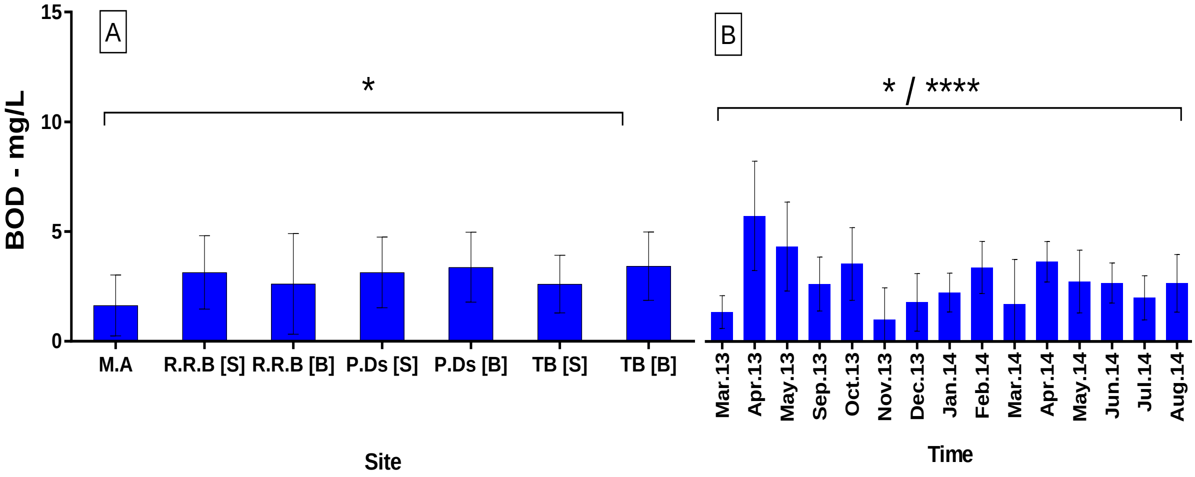

Figure 72- Average of Biological oxygen Demand (BOD) at Mark Addy, Reagent Road, Pomona Docks and Turning Basin between March 2013 and August 2014. [A] p<0.05 showing slight but significant variance between sites, and [B] (p<0.0001) showed significant differences with time. Data shown as mean ± SE, n = 16.

4.2.2.5 Chlorophyll-a

Chlorophyll-a concentration is relatively low at all sites and with different depths, Figure 74. The average concentration is mostly below 5 µg/ except for a few peaks at10µg/l during the summer. Chlorophyll-a concentrations are therefore once again generally within the mesotrophic range of 2.6-20µg/l according to Carlson 1996 with no values indicative of the eutrophic conditions observed in the historical study. There was a significant difference between sites and time (Figure 75) reflecting increased summer primary production and higher primary production in the deeper slower-flowing sites down-stream of the Mark Addy.

There was no significant relationship between either Secchi depth, phosphorous or chlorophyll-a concentration except in the Turning Basin at the bottom where there was a significant apparent effect of phosphorous on the chlorophyll-a concentration, Figure 76.

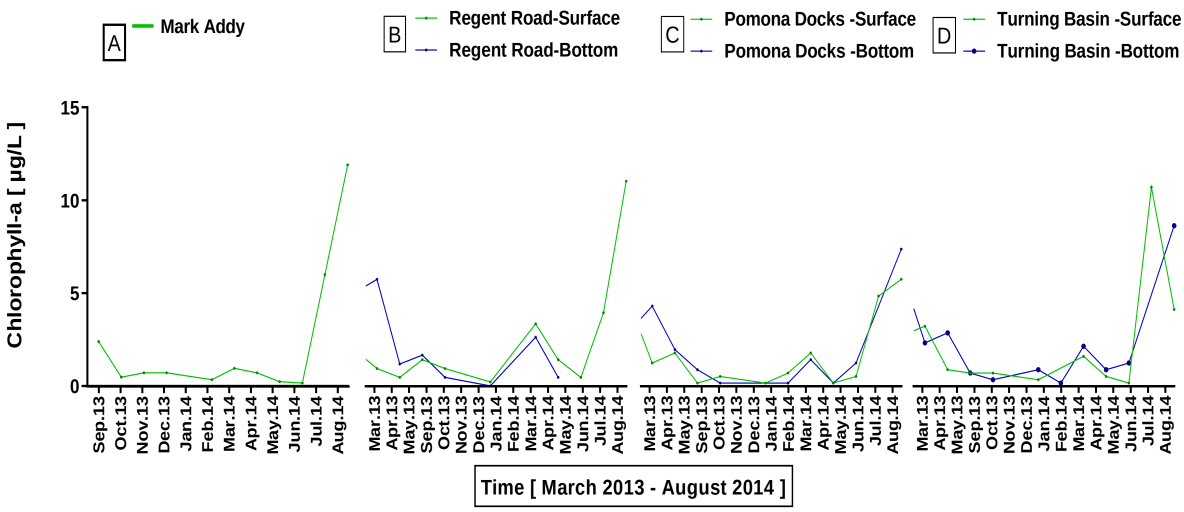

Figure 73- Seasonal changes of Chlorophyll-a concentration at Mark Addy [A], Reagent Road [B], Pomona Docks [C] and Turning Basin [D], between March 2013 and August 2014.S and B refer to surface and bottom of the water column respectively. n=13 for Mark Addy and 15 for other sites.

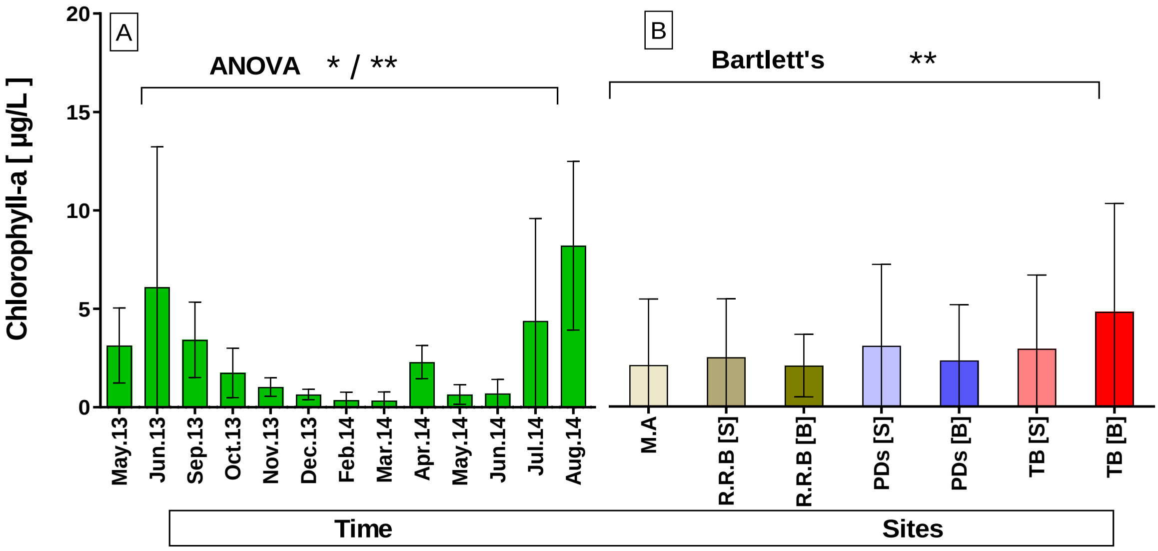

Figure 74- Average of Chlorophyll-a at Mark Addy, Reagent Road, Pomona Docks and Turning Basin between March 2013 and August 2014. (p<0.05) showing significant variance between seasons, but not significant differences between sites using ANOVA. Data shown as mean ± SE, n = 13.

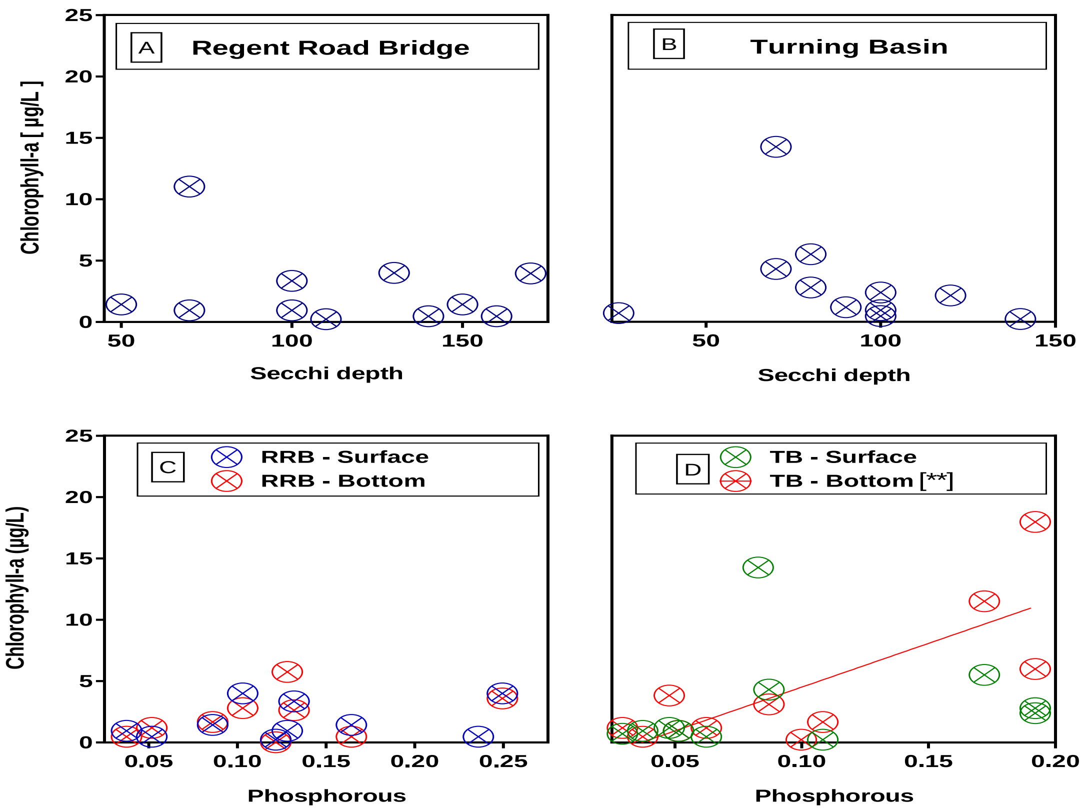

Figure 75- Relationship between Secchi depth [A and B] and phosphorous [C and D] with chlorophyll-a at Regent Road Bridge [RRB] and Turning Basin [TB] between March 2013 and August 2014. Statistical analysis by using both correlation and linear regression showed only significant effect of phosphorous on chlorophyll-a at the bottom of the water column as R^2 =0.6154, n=11.

3.2.3 Nitrogen [NO_3-N], phosphorous [PO_4- P] and ammonia

Nitrate (as N-NO_3) fluctuated around a mean of 6mg/l from March 2013 to August 2014 with just one significant peak between early winter 2013 to early Spring 2014 at all the sites below Mark Addy, Figure 77. Concentrations of nitrate were similar to historical levels in 2011-2012 and hence again classed as Very Low according to the GQA nitrate grading and never exceed the ‘Low’ grade Statistical analysis shows no differences between sites and depths but there was a significant difference with time. In addition, the average nitrate was overall rather low and was only relatively high just only around March or early spring 2014, Figure 78.

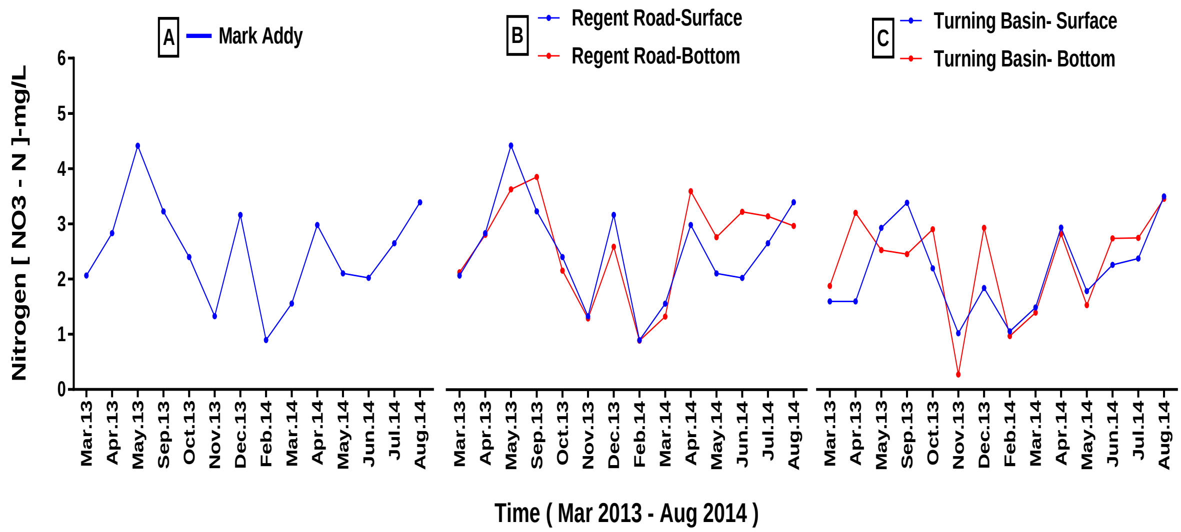

Figure 76- Seasonal changes of Dissolved nitrogen (DN) at Mark Addy [A], Reagent Road [B], Pomona Docks [C] and Turning Basin [D], between March 2013 and August 2014.S and B refer to surface and bottom of the water column respectively. n=14.

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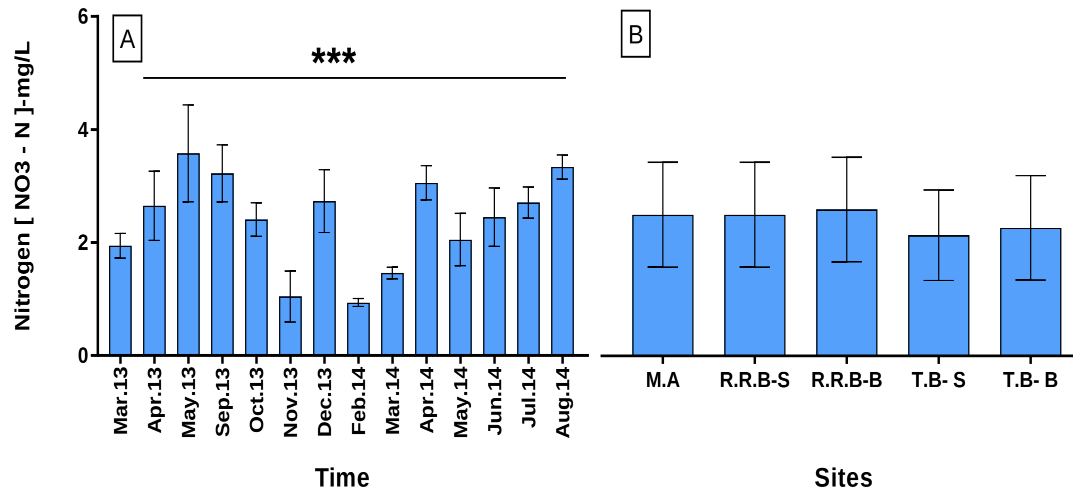

Figure 77-Average of Dissolved nitrogen (DN) at Mark Addy, Reagent Road, Pomona Docks and Turning Basin between March 2013 and August 2014. ANOVA ($p<0.05$) showed no significant variance between site and significant difference for seasons and time. Data shown as mean ± SE, n = 14.

Dissolved phosphorous as phosphate (P-PO_4) showed relatively high concentrations of around 0.5 mg/l. According to the nutrient GQA concentrations ranged of phosphate from ‘Very Low’ to ‘Moderate’. On a few occasions concentrations were classified as 'High' or 'Very, to Excessively High'. Under the Carlson (1996) classification all sites at most times would be classed as eutrophic or hypereutrophic; however sometimes, in particular during the winter concentrations were indicative of meso- or even oligotrophic conditions. There were no differences between sites and depths or with time, figure 79.

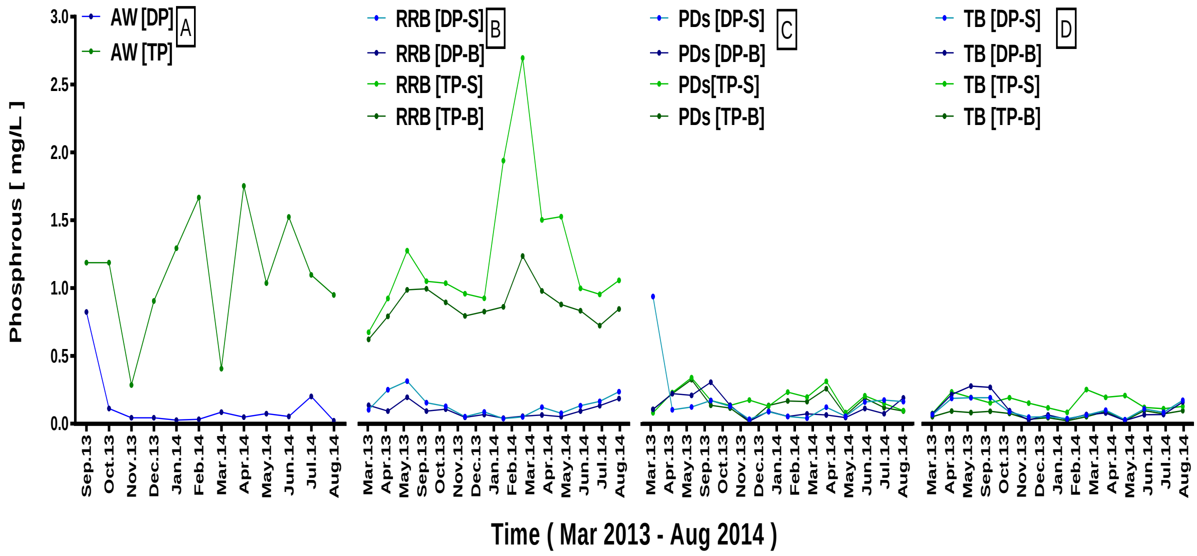

Figure 78- Seasonal changes of phosphate (DP) and total phosphorous (TP) at Mark Addy [A], Reagent Road [B], Pomona Docks [C] and Turning Basin [D], between March 2013 and August 2014.S and B refer to surface and bottom of the water column respectively. n=13 for Mark Addy and 15 for other sites.

Total phosphorous were significantly higher ($P<0.05$) than dissolved phosphorus at the upstream sites, Adelphi weir and Regent Road Bridge as the values reached more than 1.5 mg/l, figure 79A and B. However, TP similar to P-PO_4 at the downstream sites with an average concentration below 0.5 mg/l. There was a significant correlation between dissolved nitrogen and dissolved phosphorous concentration in the system, and the relationship was stronger downstream, Figure 80. Interestingly there was a poorer or no correlation between dissolved P and N at the upstream sites, and the only significant relationship was at surface level of the water column downstream of the system (Figure81).

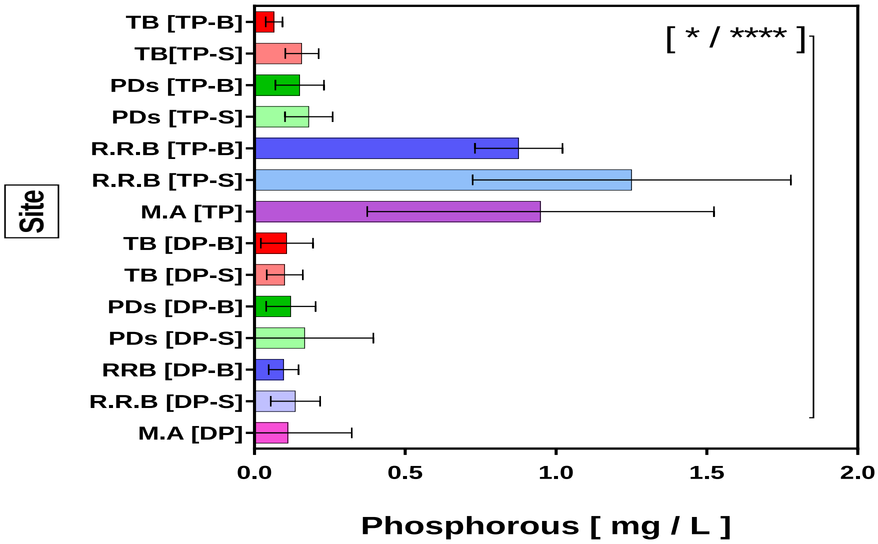

Figure 79- Variation in average total phosphorous levels [TP] between Mark Addy (MA), Regent Road Bridge (RRB), Pomona Docks (PDs) and Turning Basin (TB) sites, March 2013 to August 2014. Statistics (P<0.05) showed significance differences for [TP] between sites. Data shown as mean ± SE. n (for TP) = 7.

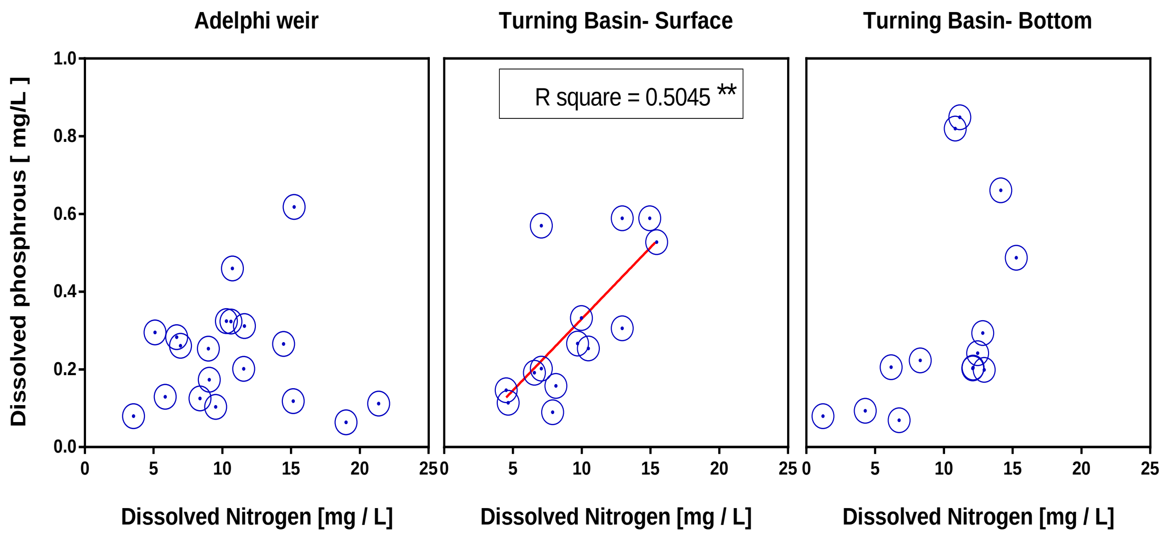

Figure 80- The relationship between dissolved phosphorous and dissolved nitrogen at Mark Addy and Turning Basin between March 2013 and August 2015. n = 19 for Mark Addy and 14 for Turning Basin. There was significant relationship at surface level downstream at Turning Basin, correlation analysis (P<0.05) showed significant relationship between nitrate and phosphorous level at Turning Basin-Surface [R^2=0.5045].

Seasonal changes in ammonia levels and the overall average shows that concentrations fluctuate around a mean of 1.6 mg/l with a maximum of around 3 mg/L. The GQA indicates that the average indicates 'Fairly Good' water quality (UKTAG, 2008) Figure 82. However, values sometimes exceeded 2.5mg/l which decreased the grading to D, 'Fair'. Ammonia levels are higher at the bottom of the water column compared to the surface which may reflect decreased oxidation to nitrate. Statistically, the trend shows significance differences both among different sites and over time, Figure 83A and B. In contrast to many parameters overall concentrations of ammonia do not change down-stream.

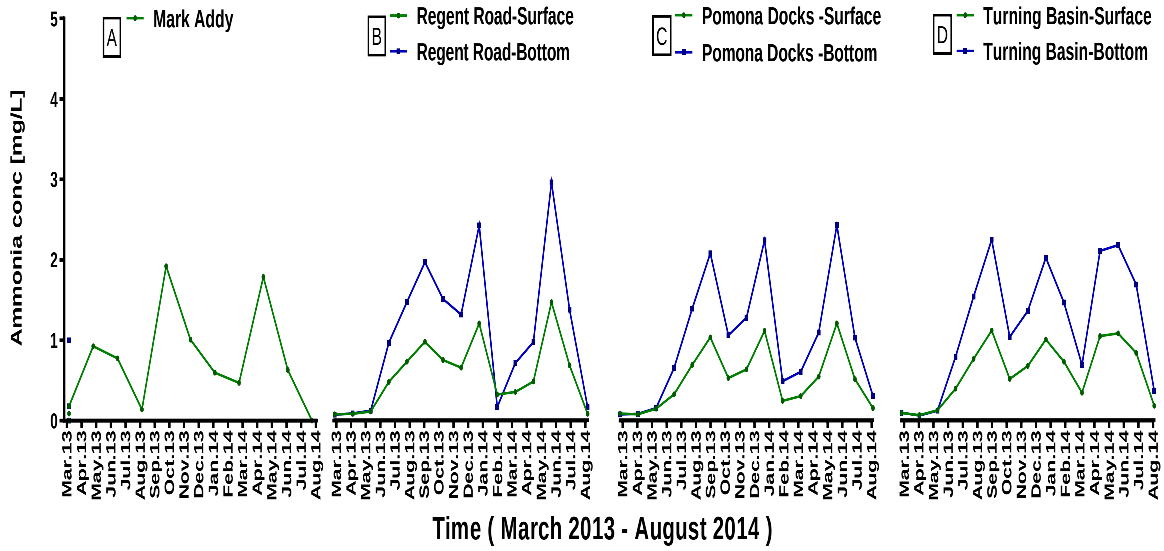

Figure 81- Seasonal changes in ammonia levels at Mark Addy [A], Reagent Road [B], Pomona Docks [C] and Turning Basin [D], between March 2013 and August 2014.S and B refer to surface and bottom of the water column respectively. n=13 for Mark Addy and 15 for other sites.

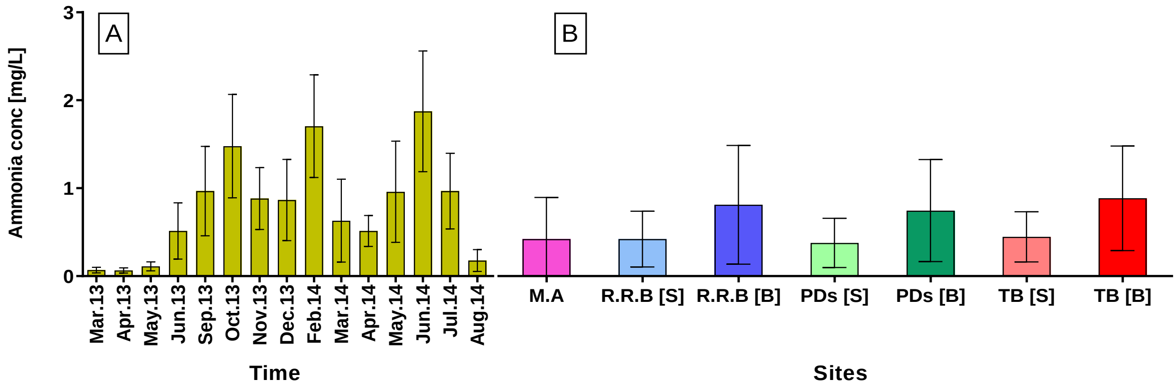

Figure 82- Average of Ammonia levels at Mark Addy, Reagent Road, Pomona Docks and Turning Basin between March 2013 and August 2014. ANOVA ($p<0.05$) showed no significant variance during different seasons and sites. Data shown as mean ± SE, n = 13.

3.2.4 Transparency and water clarity

3.2.4.1 Secchi depth

Secchi depth as an indicator for water clarity showed that the average light penetration was around one meter at all sites. There was however marked periodic decreases along the system and at various periods the water clarity did not exceeded 50cm, Figure 84, in the Turning Basin it fell as low as 20cm. Statistically, there were a slight but significant difference between sites with a decrease in Secchi depth at the Turning Basin but there were no significant seasonal differences, Figure 85.

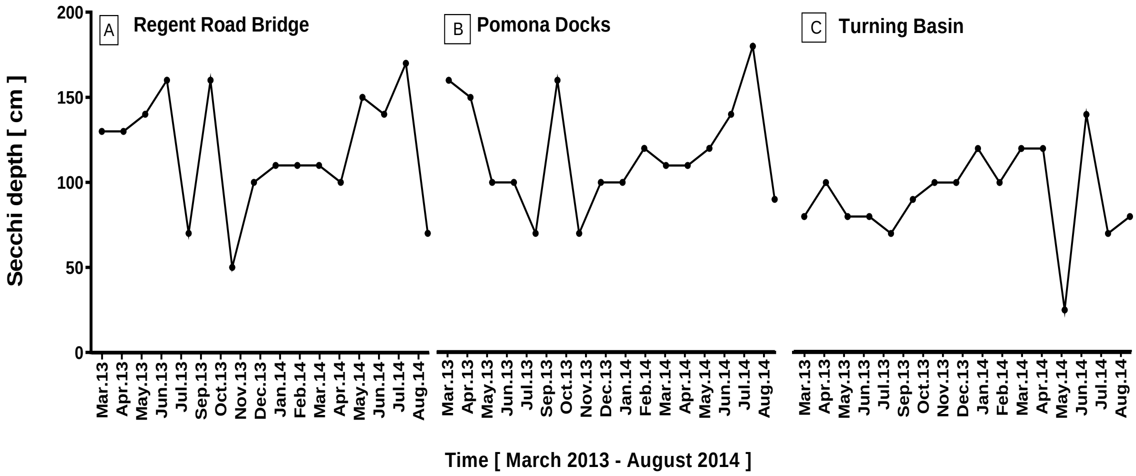

Figure 83- Seasonal changes in Secchi depth at Regent Road Bridge, Pomona Docks and Turning Basin between March 2013 and August 2014. n=17.

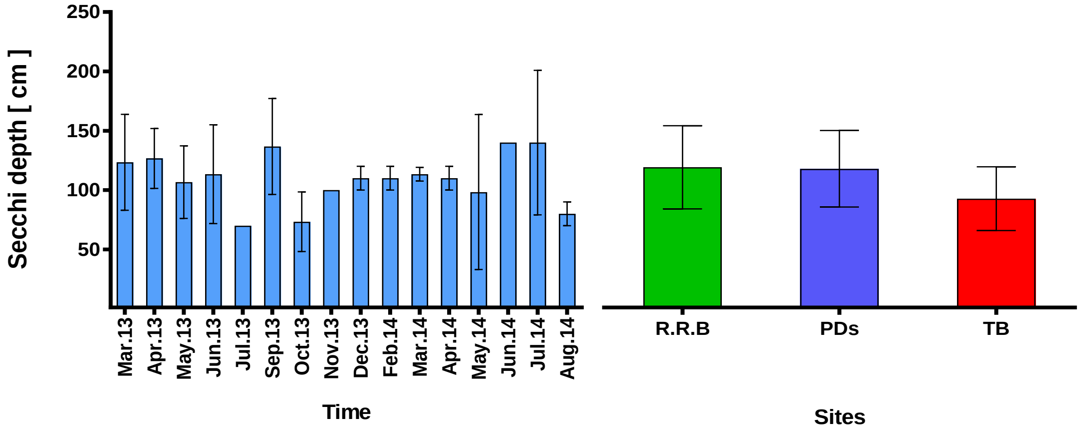

Figure 84- Average Secchi Depth at Mark Addy, Reagent Road, Pomona Docks and Turning Basin between March 2013 and August 2014. ANOVA ($p<0.05$) showed no significant variance during different seasons and sites. Data shown as mean ± SE, n = 16.

3.2.4.2 Suspended solids (TSS) and total organic matter (TOM)

Two of the most significant parameters to evaluate the quality of fresh water systems is total suspended solids [TSS] and total organic matter [TOM] as both are often markedly influenced by pollution and re-engineering of waterbodies. TSS was relatively low at the surface and varied between 1 - 10 mg/l. TSS were therefore within the normal range of up to 20 mg/L (WFD standard) with only occasional increases at Pomona Docks and the Turning Basin that slightly exceeded this threshold, reaching concentrations of 8around 30mg/L (Figure 86). There was little trend most of the time but some peaks occurred later in autumn 2013 and the middle of the summer of 2014 at Adelphi Weir where the TSS exceeded 20 mg/l, Figure 86A. TSS increased downstream and the values fluctuated between 10 - 20 mg/l; for instance, the peak in late spring of 2013 at Turning Basin, TSS level was over 30 mg/l, figure 86D. TOM fluctuated between 5 - 15 mg/l except upstream where the level generally was under 5 mg/l until Spring 2014 where there was a marked increase to 10 mg/l, figure 87A.

The average TOM levels where high with organic matter accounting for much of the TSS, fluctuating from 20 to 80% of the total at all sites and with no significant differences between surface and bottom of the water column, including downstream which again indicates that the system is well mixed. The high percentage of TOM indicates that most suspended solids originated mainly from organic debris, Figure 88.

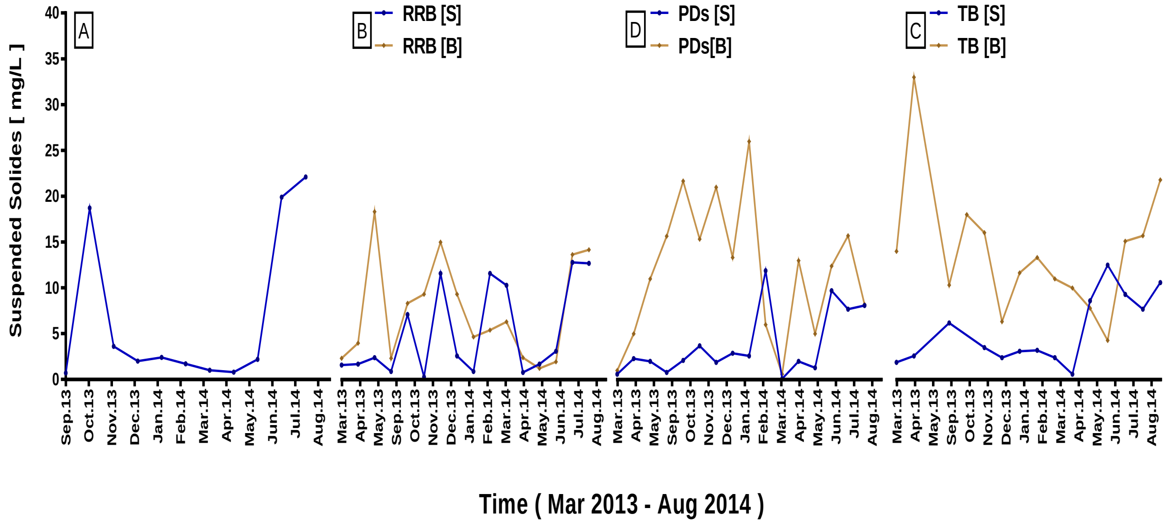

Figure 85- Seasonal changes in the levels of suspended solids (TSS) at Mark Addy [A], Reagent Road [B], Pomona Docks [C] and Turning Basin [D], between March 2013 and August 2014.S and B refer to surface and bottom of the water column respectively. n=11 for Mark Addy and 16 for other sites.

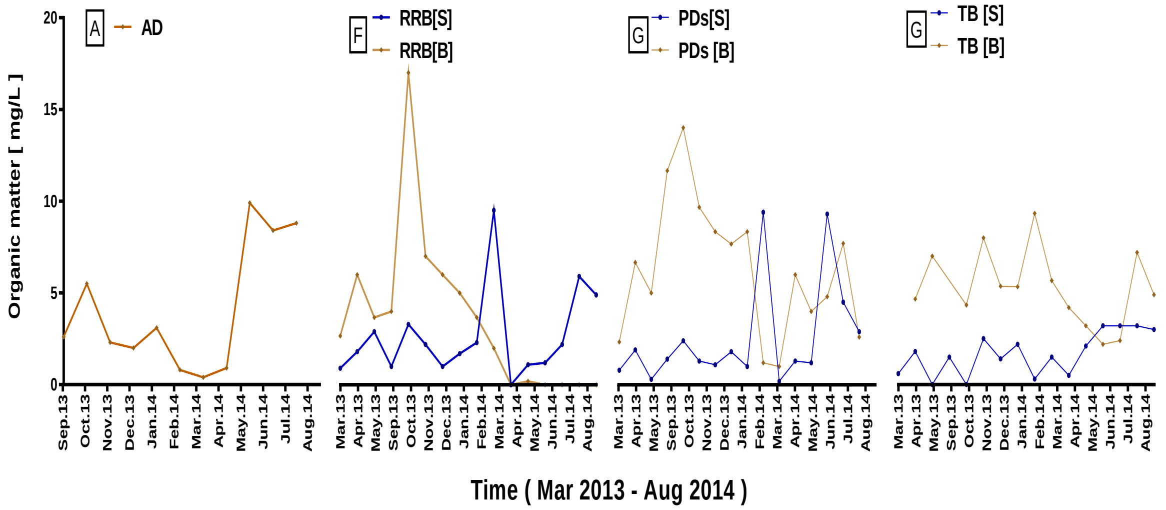

Figure 86- Seasonal changes in the level of organic matter (TOM) at Mark Addy [A], Reagent Road [B], Pomona Docks [C] and Turning Basin [D], between March 2013 and August 2014.S and B refer to surface and bottom of the water column respectively. n=11 for Mark Addy and 16 for other sites.

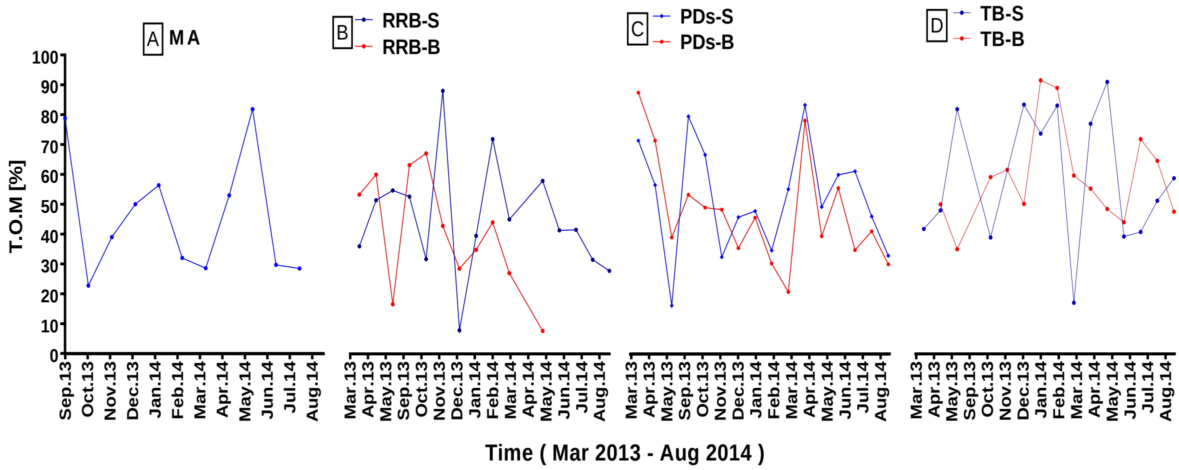

Figure 87- Seasonal changes in the percentage of organic matter (TOM) at Mark Addy [A], Reagent Road [B], Pomona Docks [C] and Turning Basin [D], between March 2013 - August 2014.S and B refer to surface and bottom of the water column respectively. n=11 for Mark Addy and 16 for other sites.

According to the statistical analysis, there is significant difference in TSS and TOM between sites and different seasons, Figure 89.

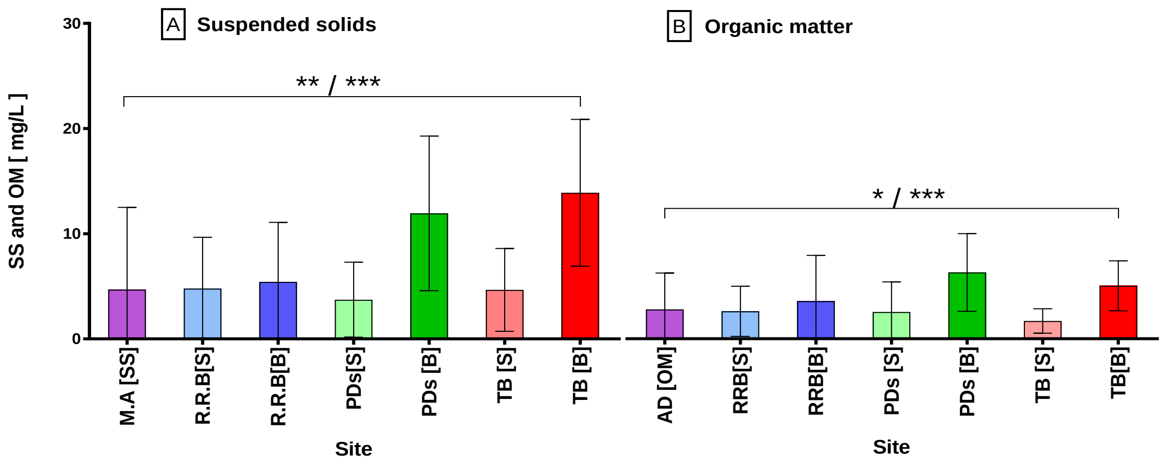

Figure 88- Variation of overall average total suspended Solids (TSS) and organic matter (TOM) between March 2013 and August 2014. ANOVA (P<0.05) showed significance differences between sites for both TSS and TOM. Data shown as mean ±SEM, n = 13.

3.2.4.3 Effect of suspended solids (TSS) and chlorophyll on water clarity and transparency

The relationship between TSS and TOM and Secchi depth in the system is illustrated on Figure 90. According to the statistics, there were no a clear evidence that either TSS or TOM affected Secchi depth and hence water clarity and light penetration along the system.

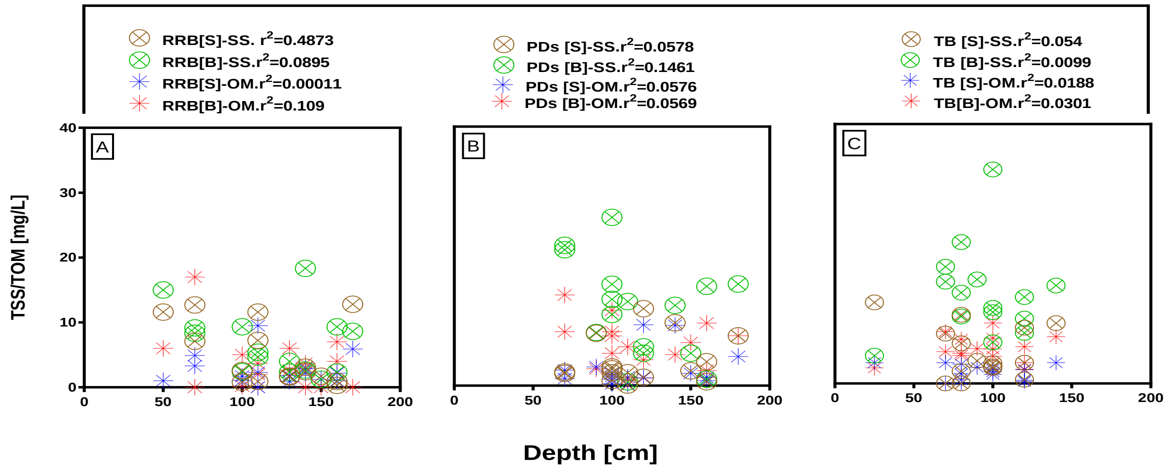

Figure 89- Correlation coefficient between (TSS), (TOM) and Secchi Depth. March 2013 - August 2014. Number of XY values=16. Deviation from zero not significant at all sites.

Statistical analysis also showed that there was no significant effect of chlorophyll-a on water clarity and light penetration along the system at different sites or between the surface and bottom of the water column, Figure 91.

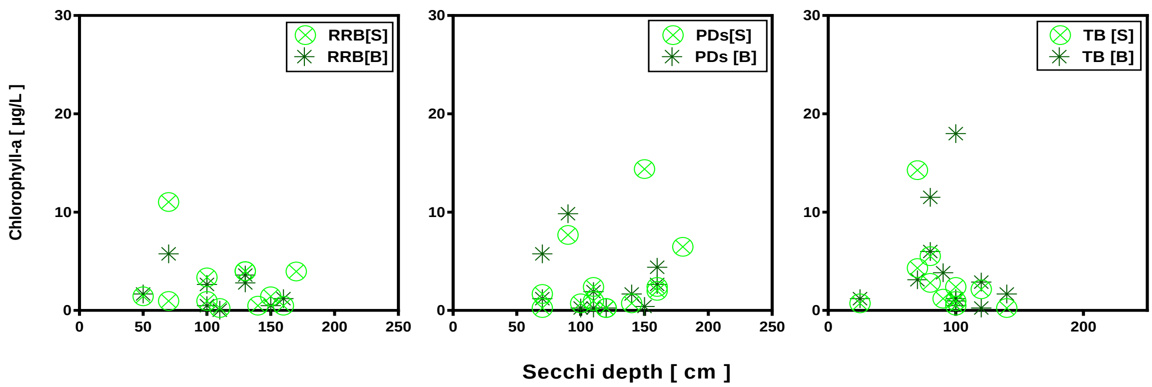

Figure 90- Relationship between chlorophyll-a and water clarity and light penetration at Regent Road Bridge (RRB), Pomona Docks (PMD) and Salford Quays (SQs). XY numbers = 9, 12, 13. Statistical analysis showed no significant relationship (P>0.05) at any site.

3.2.5 Heavy metals

As expected, concentrations of individual heavy metals varied markedly so those at relatively high concentrations of above 100 µg/l, specifically Fe, Zn and Mn, are examined separately to Cu, Pb and As.

3.2.5.1 High concentration heavy metals

Amounts of high concentration heavy metals were generally high in the whole system according to the most recent UK-WFD (2015) standards where the limit is Fe 1 mg/l, Zn 10.9 mg/l, and Mn 123 mg/l. The Fe concentration fluctuated within the range of 50 - 200µg/l, during March 2013, and was relatively high both at the bottom of the water column at Regent Road Bridge (550 µg/l), and further downstream at the Turning Basin surface (750 µg/l). There was a decrease on April 2013 where the concentration fell to 50µg/l at all sites. The trend was for a decrease between February (200 µg/l) and July (50 µg/l) 2014 with a slight increase over the summer up to August to more than 100 µg /l and a decrease thereafter, Figure 92. Although the average Fe at each site indicates concentrations mostly around 200 µg/l at all sites, the seasonal average showed significant change with time.

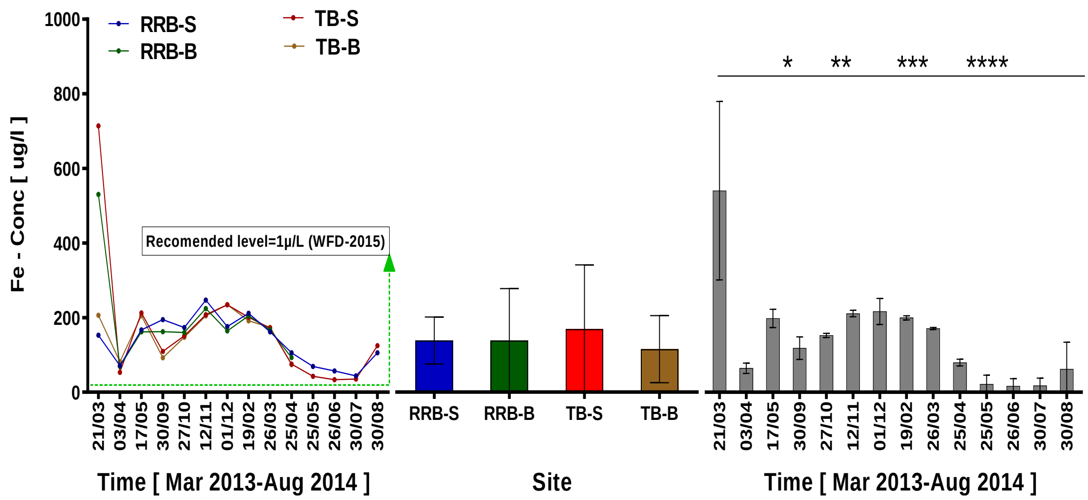

Figure 91- Seasonal changes and statistical analysis of Fe concentration at Regent Road Bridge (RRB) and Turning Basin (TB) between March 2013 and August 2014. Statistics (P < 0.05) showed no significant differences in the average Fe concentration at the two sites or with depth. There were significant differences with time. Data shown as mean ±SEM, n = 14.

The level of Zn was often high when compared to the EQS of <10.9µg/l due to significant differences between seasons, Figure 93, including a continuous increase during March, April and May 2013 with the concentration increasing from 50 µg /l to nearly 400 µg /l. There was little change thereafter but with a slight fluctuation around 50 µg/l; the only significant peak in Zn was in October 2013 at Regent Road Bridge at the bottom of the water column where concentrations exceeded 500 µg/l.

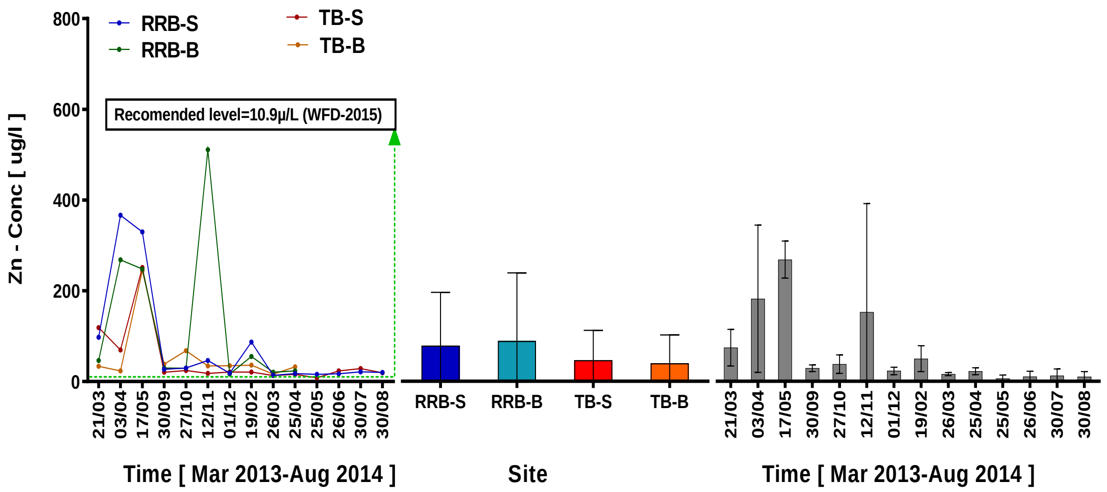

Figure 92- Seasonal changes and statistical analysis of the high concentration heavy metal Zn between Regent Road Bridge (RRB) and Turning Basin (TB) from March 2013 to August 2014. Statistics ($P < 0.05$) showed no significant differences for different sites and during all seasons. Data shown as mean ±SEM, n = 14.

The general trend in Mn concentration was a decrease at all sites from a high in March 2013. There were several peaks that exceeded the recommended EQS of <123ug/l and concentrations fluctuated between 25µg/l and 150µg/l. There was significant difference only with season, Figure 94.

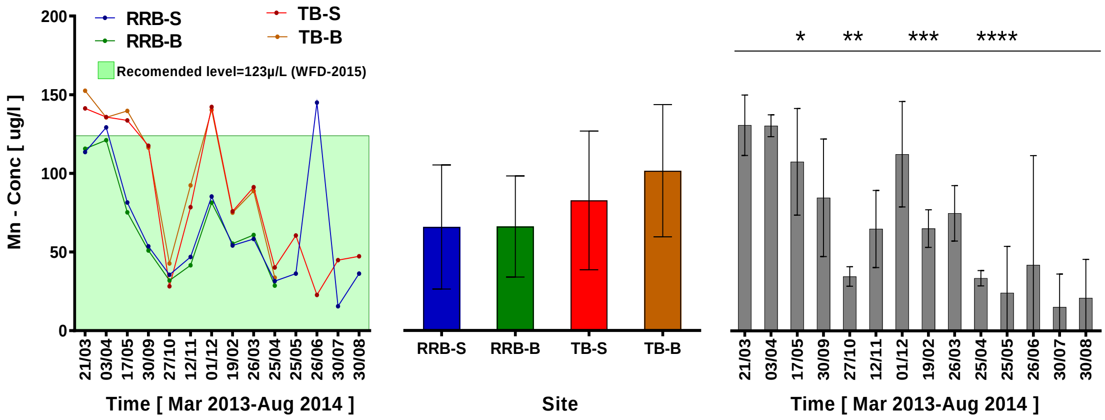

Figure 93- Seasonal changes and statistical analysis of the high concentration heavy metal Mn between Regent Road Bridge (RRB) and Turning Basin (TB) from March 2013 to August 2014. Statistics ($P < 0.05$) showed no significant differences between different sites. There were significant differences between seasons. Data shown as mean ±SEM, n = 14.

3.2.5.2 Low concentration heavy metals

The concentration of Cu mostly exceeded recommended standard (EQSs 1µg/l) and mostly fluctuated around 5 µg/l, with a range of between 2-10µg/l and with some peaks at all sites. There was significant difference between upstream at the surface and downstream in the lower water column. There was also a noticeable decrease between upstream and downstream; this also can be seen where the level of the bottom concentration is by far lower than the surface, Figure 95.

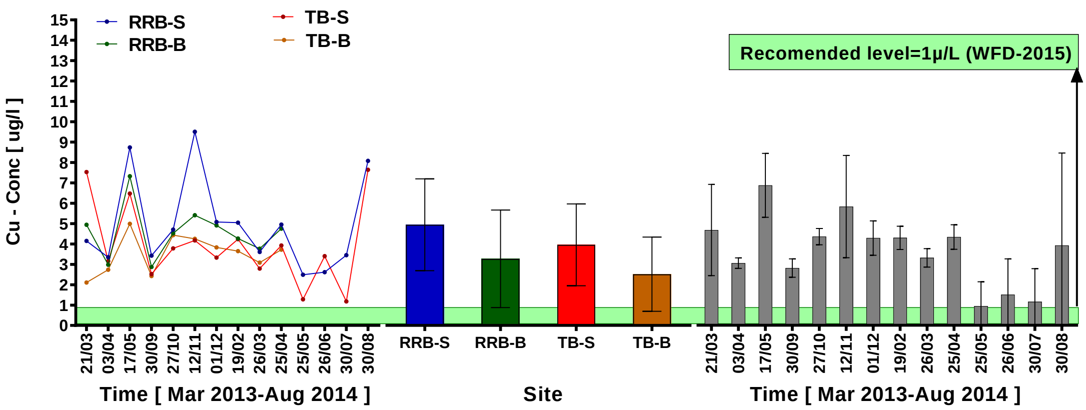

Figure 94- Seasonal changes and statistical analysis of the low concentration heavy metal Cu between Regent Road Bridge (RRB) and the Turning Basin (TB) from March 2013 to August 2014. Statistics ($P < 0.05$) showed significant differences between different sites. Data shown as mean ±SEM, n = 14. The level of As and Pb was low

(Recommended standard EQSs 50µg/l and14 µg/l respectively; UK-WFD/2015) and the trend was a decrease in both metals with time although the concentrations were always below 3µg/l. There were some significantly higher concentrations at specific times but remained below the EQSs. There were no significant difference between sites for both As and Pb, figure 96 and figure 97.

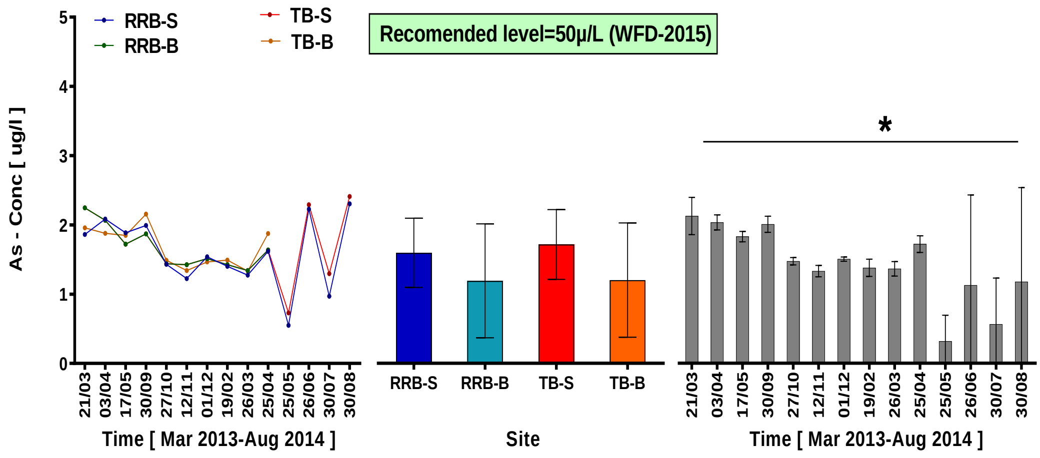

Figure 95- Seasonal changes and statistical analysis of low the concentration heavy metal Pb between Regent Road Bridge (RRB) and the Turning Basin (TB) from March 2013 to August 2014. Statistics (P < 0.05) showed no significant differences for different sites but there were significant differences of time. Data shown as mean ± SEM, n = 14.

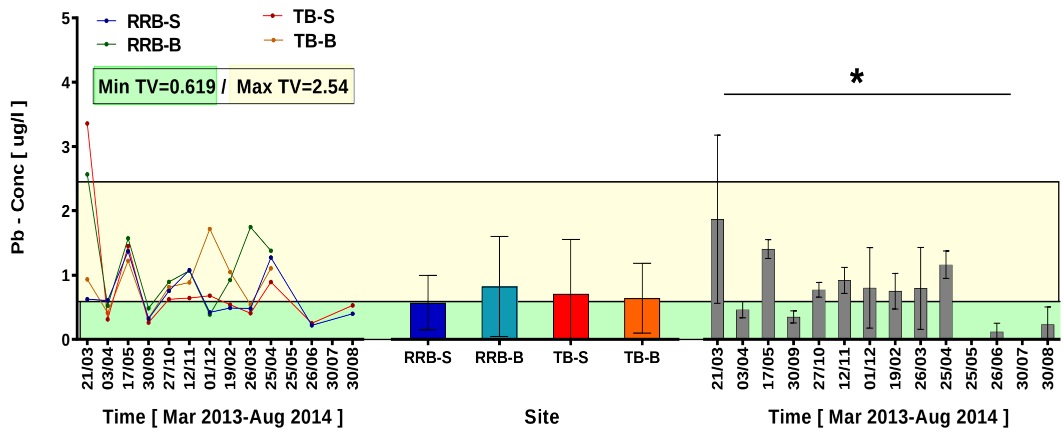

Figure 96- Seasonal changes and statistical analysis of the low concentration heavy metal As between Regent Road Bridge (RRB) and the Turning Basin (TB) from March 2013 to August 2014. Minimum Threshold Value 0.619 and Maximum Threshold Value 2.54 µg/l. Statistics (P < 0.05) showed no significant differences for different sites but there were significant differences with time. Data shown as mean ± SEM, n = 14.

3.2.5.3 The effect of discharge [Q] on heavy metal concentration

Two common metals from each group of the high and low concentration group were selected to examine the relationship between discharge and heavy metal concentration. Analysis of Fe and As showed that there was no relationship between concentration of these heavy metals and discharge at Regent Road (upstream) and downstream at Pomona Docks, Figure 98. The positive R^2values suggest that discharge has possible effect that could lead to an increase of the concentration of some specific trace metals; possibly due to disturbing and re-suspending the sediment.

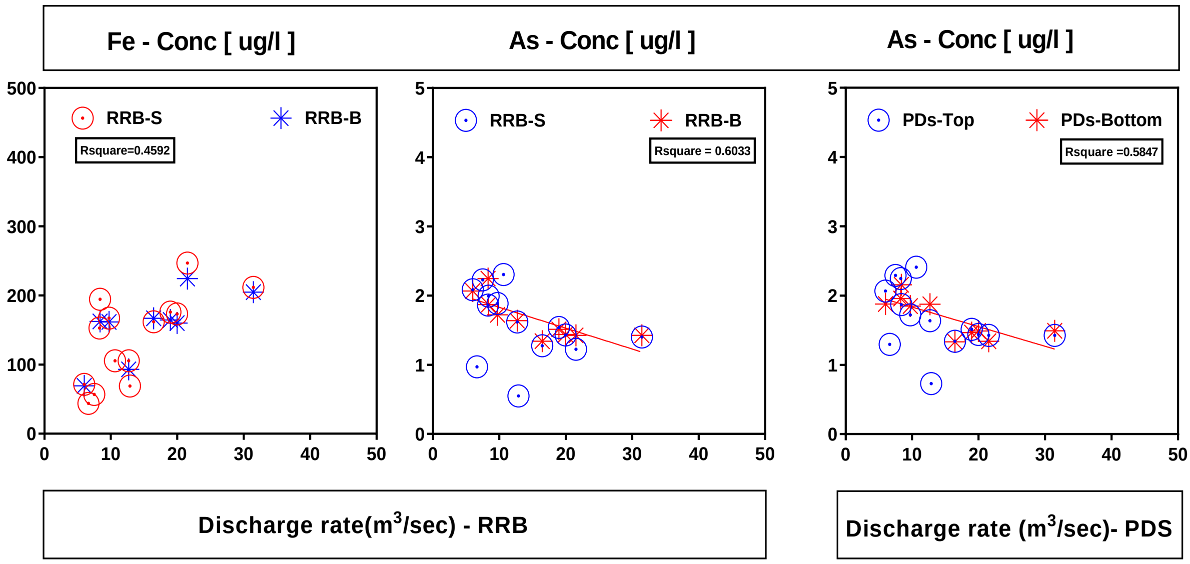

Figure 97- Effect of discharge [Q] on heavy metal concentration at Regent Road Bridge [RRB] and Pomona Docks [PDs] between March 2013 and August 2014. Fe represents a high level heavy metal and As a low level heavy metal. Statistics showed there was no significant effect of discharge on the level of heavy metal upstream and downstream. N[X=10, y=14].

3.2.5.4 Ordination analysis

Ordination analysis was carried out to examine the grouping of sites in relation to key chemical-chemical parameters Figure 99. This revealed PC1 and PC2 contribute more than 50% of the physio-chemical parameters shown in Table14. The first PC accounted for 35% of the overall variance and was most heavily weighted ($p<0.05$) to suspended solids, discharge and Secchi depth whereas that second PC axis accounted for 25% and was most heavily weighted to discharge. The plot of PC1 and PC2 revealed Pomona Docks and the Turning Basin sites clustered whereas the upper site Regent Road Bridge grouped differently (Figure 99).

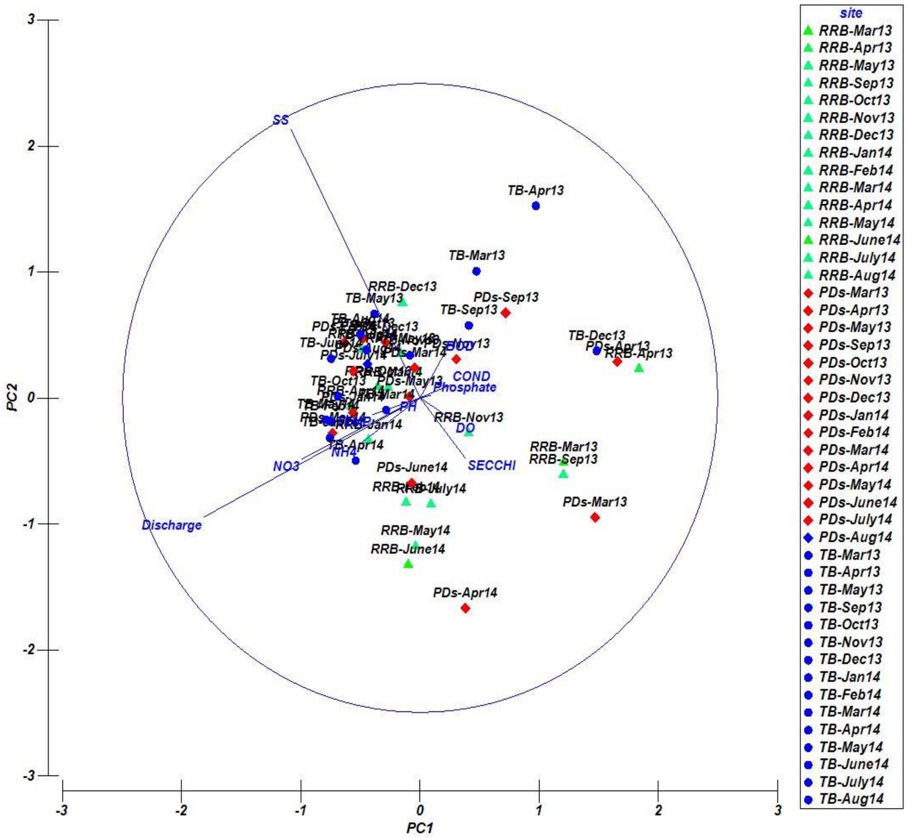

Figure 98 - Ordination diagram of physio-chemical parameters upstream at Reagent Road Bridge [RRB], Pomona Docks [PDs], and downstream at the Turning Basin [TB] between March 2013 and August 2014.

Table 14: **Principal component analysis of physio-chemical variables between March 2013 and August 2014. The major contributory variables are emboldened**

Variable	PC1	PC2	PC3	PC4	PC5
Dissolved oxygen	0.113	-0.073	0.148	0.446	-0.046
PH	-0.003	-0.004	0.003	-0.005	0.016
Temperature	-0.16	-0.053	-0.1	-0.811	0.023
Conductivity	0.102	0.041	-0.076	0.215	0.493
BOD	0.079	0.136	0.243	0.027	-0.426
NH_4	-0.207	-0.149	-0.114	-0.064	0.46
NO_3	-0.4	-0.195	0.853	-0.033	0.219
PO_4-p	0.036	0.008	0.024	-0.07	-0.037
Suspended solids	**-0.434**	**0.855**	0.031	0.042	-0.066
Discharge	**-0.728**	-0.38	-0.354	0.261	-0.3
Secchi depth	**0.152**	-0.192	0.188	-0.129	-0.467

In summary, the seasonal survey reveal similar levels of pollution and spatial changes as was observed following analysis of the historical date from the later period, namely 2010-2012. Similarities included consistently high conductivity, elevated levels of phosphate but low levels of nitrate. Stratification was again not observed and levels of DO were generally high and BOD low. Ammonia was also generally low although occasional elevated levels resulted in a GQA classification of 'Fair'. Suspended solid concentrations did not reach the very high values observed during the historical survey but did sometimes exceed standards, although by a smaller amount. The organic content of the suspended solids was often very high, pointing to a significant organic input. Primary productivity is likely again to have been constrained by suspended solids given that phosphate indicates hypereutrophic conditions whereas chlorophyll concentrations are indicative of mesotrophy. The episodically high levels of suspended solids may therefore continue to constrain phytoplankton density despite the elevated levels of nutrients, in particular phosphorus, which would otherwise result in eutrophication. PCA suggests that suspended solids, Secchi depth and discharge are key variables delineating sites and hence potential influences on the in-stream biota. Heavy metals often exceed EQSs and are influenced by discharge events that are responsible for resuspension of sediment contaminants. The benthic invertebrate community is mostly composed of pollution-tolerant taxa and exhibits a high degree of dominance.

The community is probably degraded by a complex combination of geomorphological physio-chemical and seasonal variation as well as discharge events that is responsible for an increase of pollutants such as, possibly, ammonia plus discharges that drive up the level of conductivity (Graça et al., 2004). Potential impacts on the benthic invertebrate community will be considered further in the discussion.

3.2.6 Biological structure

3.2.6.1 Planktonic community

Phytoplankton

Densities of phytoplankton were low as the highest values reached just under 800 per litre even in summer time. Average values showed significant differences between sites and seasons, 100. The most common species fluctuated between 0 and just over 600 individuals/litre at different sites. The most common individuals among the phytoplankton community were *Synedra* (S), *Nitzschia* (NT), *Asterionella* (A), and *Diatoma* (D). Generally, numbers reached nearly 800/ litre during the summer phytoplankton peak, Figure 101A, B and C. There are a wide variety of low density taxa (*Melosira sp* less than 30/L) within the phytoplanktonic population, for instance: *Diatoma* (D), *Navicula* (NV) and *Melosira* (M) that remained below 150, 80 and 30 individuals/L respectively, figure 101D, E and F.

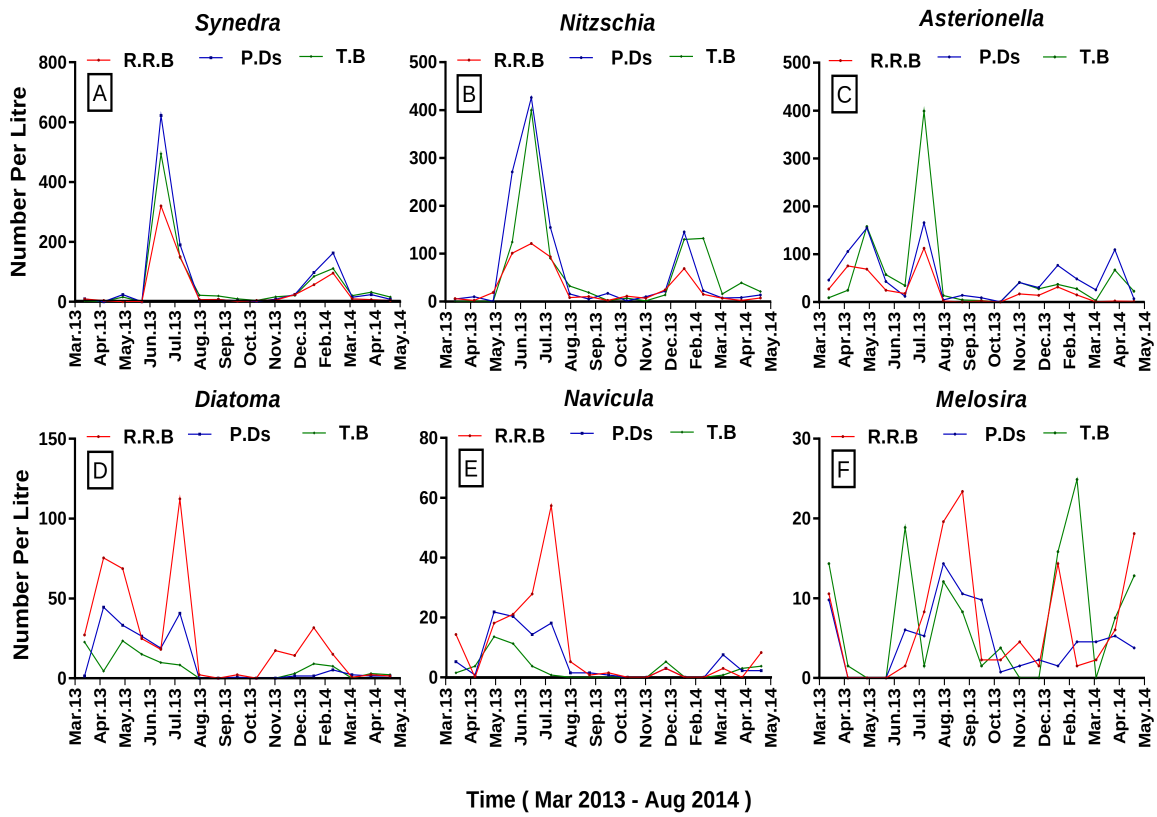

Figure 99- Seasonal changes in the most common (100-600 individuals/L) phytoplankton taxa at Regent Road Bridge (RRB), Pomona Docks (PD) and the Turning Basin (TB) between March 2013 and August 2014.

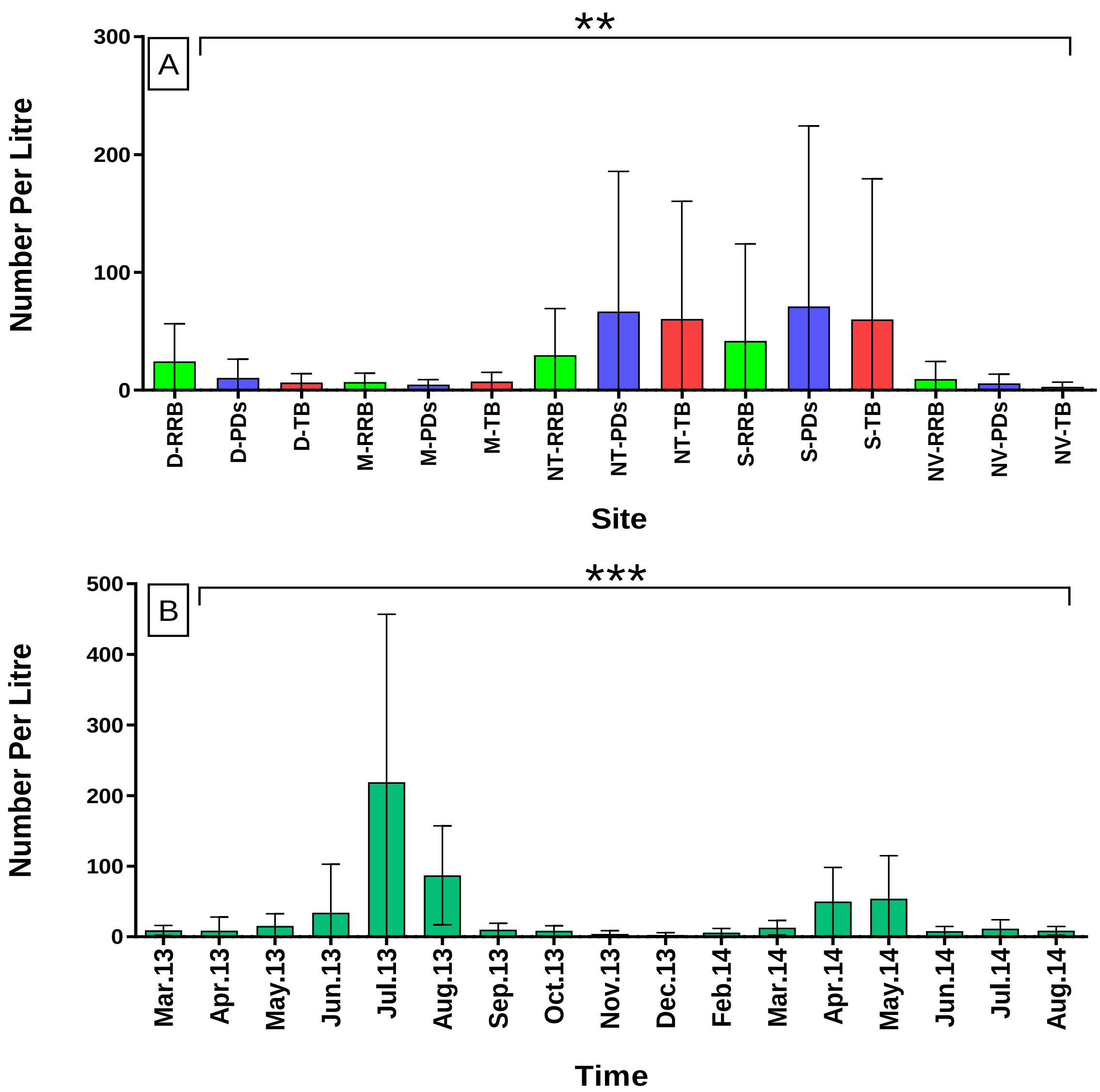

Figure 100- Seasonal average of the most common phytoplankton [D=*Diatoma*, M=*Melosira*, NT=*Nitzschia*, S=*Synedra*, NV=*Navicula*] between March 2013 and August 2014. Statistics ($p<0.05$) showed significant differences between sites (A), n=15, and with time (B), n=17.

There is potential for discharge as a surrogate for flow to disrupt primary productivity by flushing out the phytoplankton. However, statistical analysis showed that there was no effect of discharge on seasonal density of the most commonphytoplankton genera at Regent Road Bridge, Pomona Docks and the Turning Basin between 2000 and 2012. The R^2 were below 0.16 for all genera and $P<0.05$ showed there was no effect of discharge on all phytoplankton biota/ genera.

The TSI was calculated using the mean values between March and August 2013 (the period where algal productivity would be expected to be highest) for total phosphate, chlorophyll a and Secchi depth in the Turning Basin. The range was between 50-60 (Table 15) which, according to Carlson's (1977), classification is mildly eutrophic with decreased transparency, an anoxic hypolimnion and largely coarse fisheries that supports all swimmable/aesthetic uses but remains "threatened". However, the TSI for phosphorus indicates a high level of eutrophication with score between 69 and 79 while that based on chlorophyll only ranges from 41 to 51 which Carlson states indicates moderately clear water although with the potential for hypolimnetic anoxia in the summer. The reason for the disparity is shown to the Secchi TSI which, with the exception of storm events, ranges from 57.07 to 62.00 (Table15), again indicating eutrophic conditions. However, the reason for the low Secchi depth is the high suspended solid load which reduces photic depth and high primary production. Storm events do not markedly change the indices although the Secchi index decreases slightly; presumably due to run-off containing less suspended solids due to dilution.

Table 16: TSI index for seasonal survey 2013 and 2014 for Regent Road Bridge and Turning Basin, and for storm events between July and September 2014

	Regent Road Bridge		Turning Basin		Pomona Docks(Storm events)
Time	**2013**	**2014**	**2013**	**2014**	**1ST July – 15th October 2014**
TSI-P	**77.97**	**73.86**	**79.30**	**68.79**	**70.33**
TSI-C	**40.87**	**43.91**	**46.78**	**47.22**	**50.48**
TSI-S	**57.07**	**56.66**	**61.20**	**62.00**	**56.33**
Average	**58.64**	**58.15**	**62.43**	**59.34**	**59.05**

In summary, in common with the long term (historical) TSI (Table12), Suspended solids are still the main parameter which restrains phytoplankton productivity in the lower Irwell and the upper MSC.

Ordination analysis revealed that PC1 and PC2 ($P<0.05$) contributes 80% of phytoplankton biodiversity and the significant parameters influencing the *Synedra, Nitzschia, Asterionella* and *Diatoma* communities are temperature, BOD, ammonia and phosphate (Figure 102). No obvious grouping of sites is revealed by the analysis.

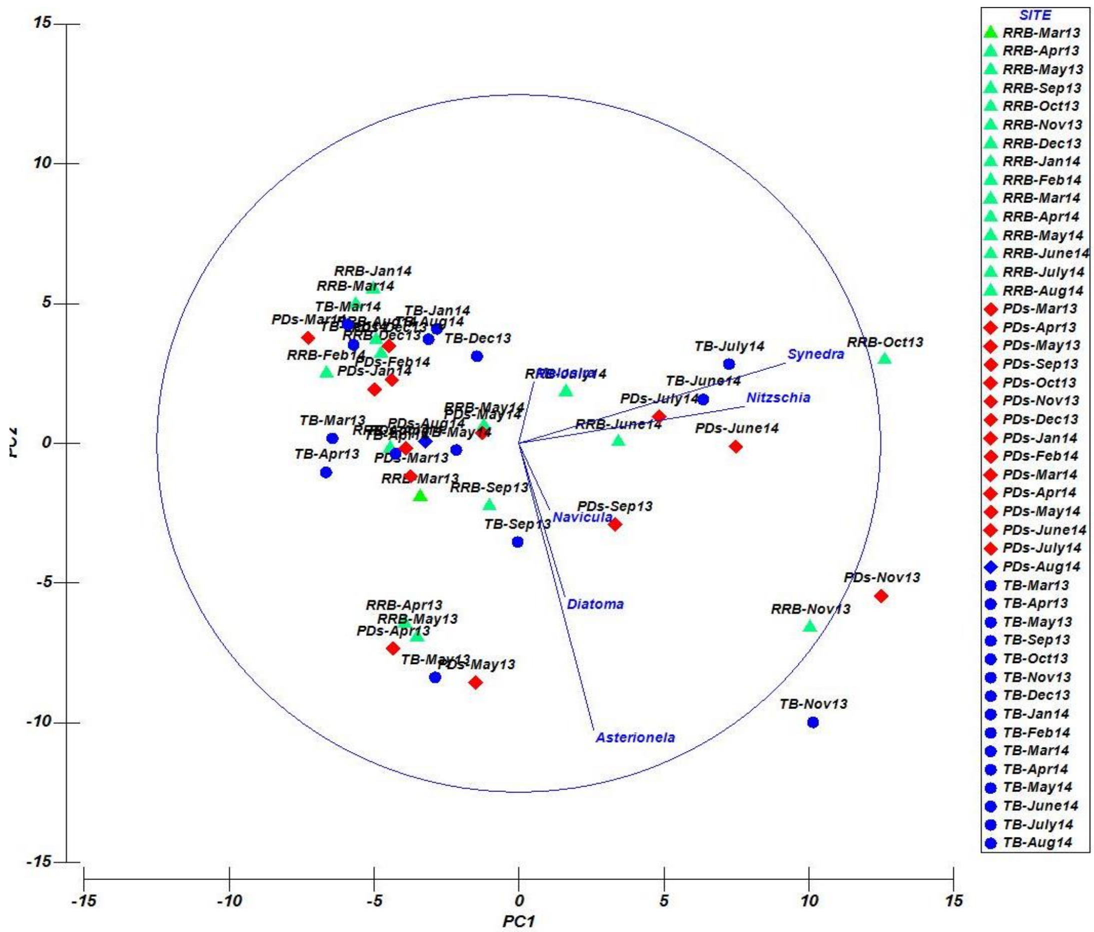

Figure 101: Ordination diagram of phytoplankton community (BIOENV) upstream at Reagent Road Bridge [RRB], Pomona Docks [PDs], and downstream at the Turning Basin [TB] between March 2013 –August 2014. PC1 60.3% and PC220.2%.

PC1 (60.3%) and PC2 (20.2%) ($P<0.05$) represents significant factors which are temperature, BOD, ammonia and phosphate that contributes to 80.5% of phytoplankton biodiversity such as Synedra, Nitzschia, Asterionella and Diatoma.

Zooplankton community

The most common zooplankton taxa were Cyclopoida*,* the Cladocerans *Macrothrix, Eurycercus, Daphnia, Dipterna* and Sidae. These taxa are shown in Figure 103 according to their density, from A to F. There was a clear variation between different sites along the system. For instance, Turning Basin downstream, all species were common and with relatively high density ranging between 800-5000/m^2. Zooplankton population density was low at the middle siteat Regent Road Bridge (RRB) and Pomona Docks (PMD). Statistical analysis showed only significant differences between zooplankton populations at different sites, Figure 104. The downstream sites, in particular the Turning Basin, were the favoured by most zooplankton populations. For instance, Cyclopoida, *Macrothrix*, *Eurycercus* and *Daphnia* reached 900, 700, 600, 500m^2 respectively, Figure 105. In addition, seasonal averages of Cyclopoida and Cladocera were similar during all seasons with peaks in the summer/autumn 2013 and spring 2014 following phytoplankton peaks which were more marked during 2013 than 2014.

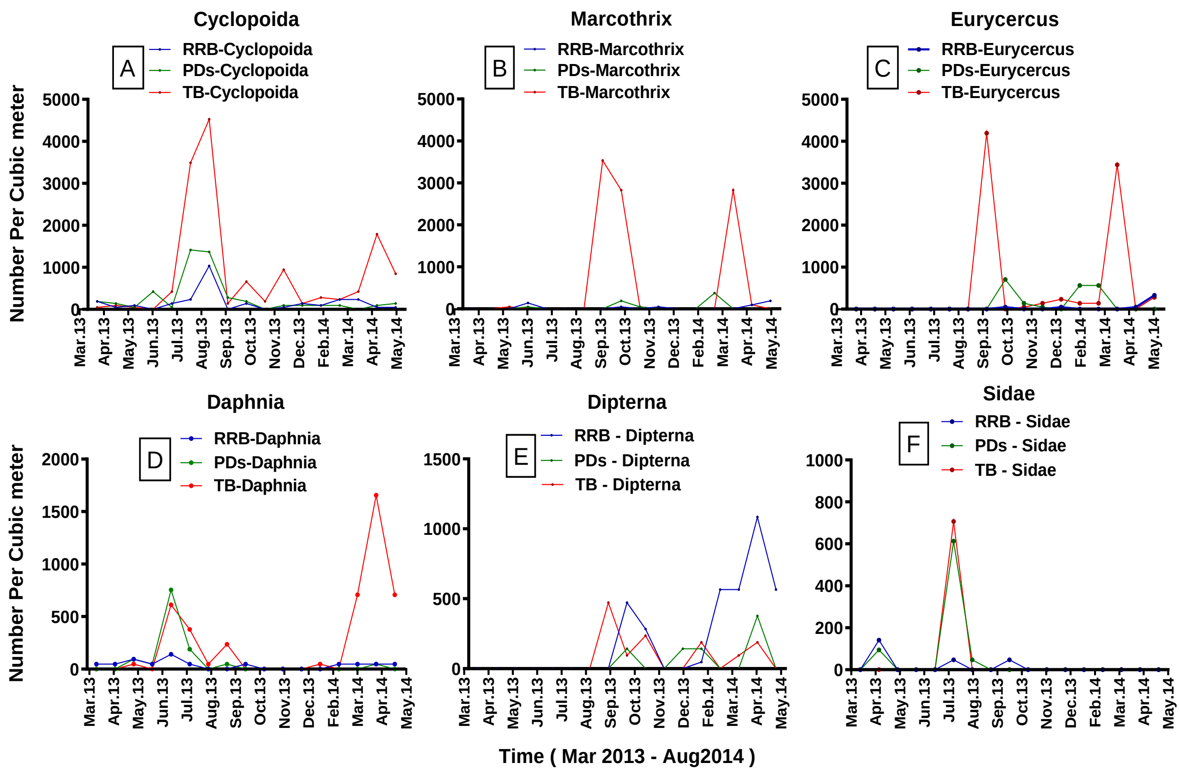

Figure 102- Seasonal changes in the most common Zooplankton populations at Regent Road Bridge (RRB), Pomona Docks (PD) and the Turning Basin (TB) between March 2013 and August 2014.

Many studies suggest that zooplankton density is subject to either vertical migration or horizontal movement due to high flow rate and discharge. Therefore, a 32 hour investigation was carried during two typical days (Figure106). Zooplankton density was low throughout the sampling period but there were some trends that point to possible diurnal effects on zooplankton density as some species were higher during the night and low during the day such as *Cyclopoida* and *Helopedium*. Others were rather low at night and high during the day, such as *Daphnia* and *Macrothrix*. The other trends are either a slight increase during night and day time respectively, as with *Bosmina*, or continuous fluctuations during continuous survey over two days such as *Eurycercus*.

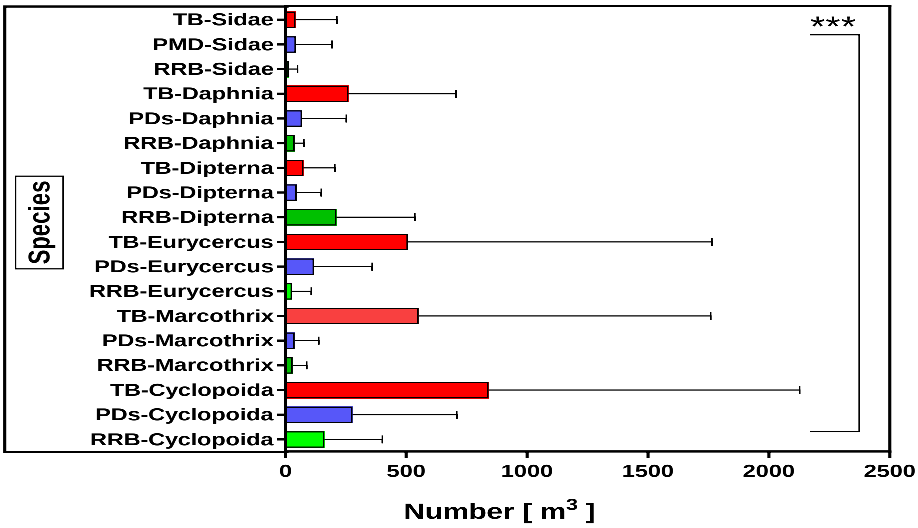

Figure 103- Seasonal average of the most common zooplankton between March 2013 and August 2014. Statistics (p<0.05) showed significant differences between sites. Data shown as mean ±SEM, n = 18.

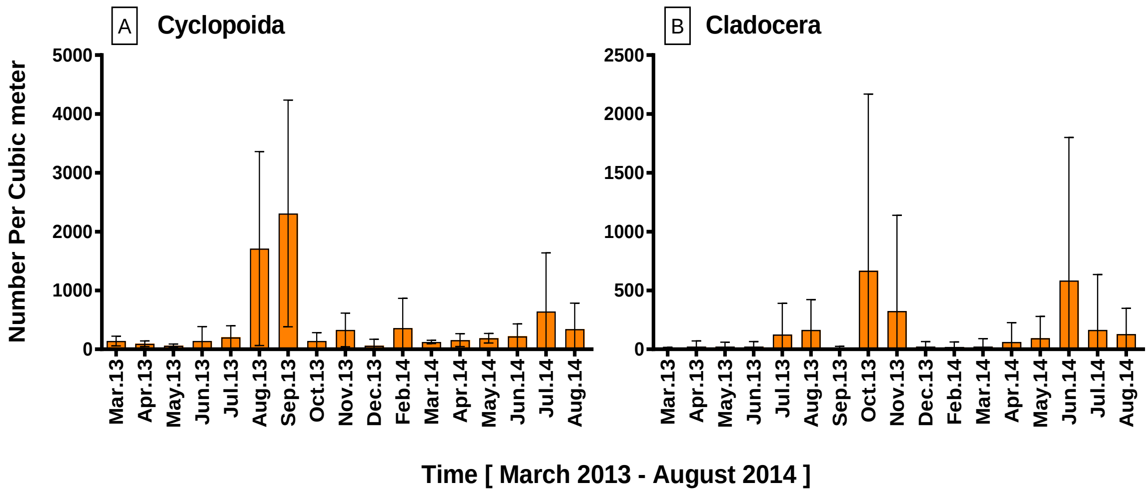

Figure 104- Seasonal average of *Cyclopoida* (A) and *Cladocera* (B) populations between from all three sites from March 2013 to August 2014. Statistics (p<0.05) showed significant differences during different seasons. Data shown as mean ±SEM, n = 13

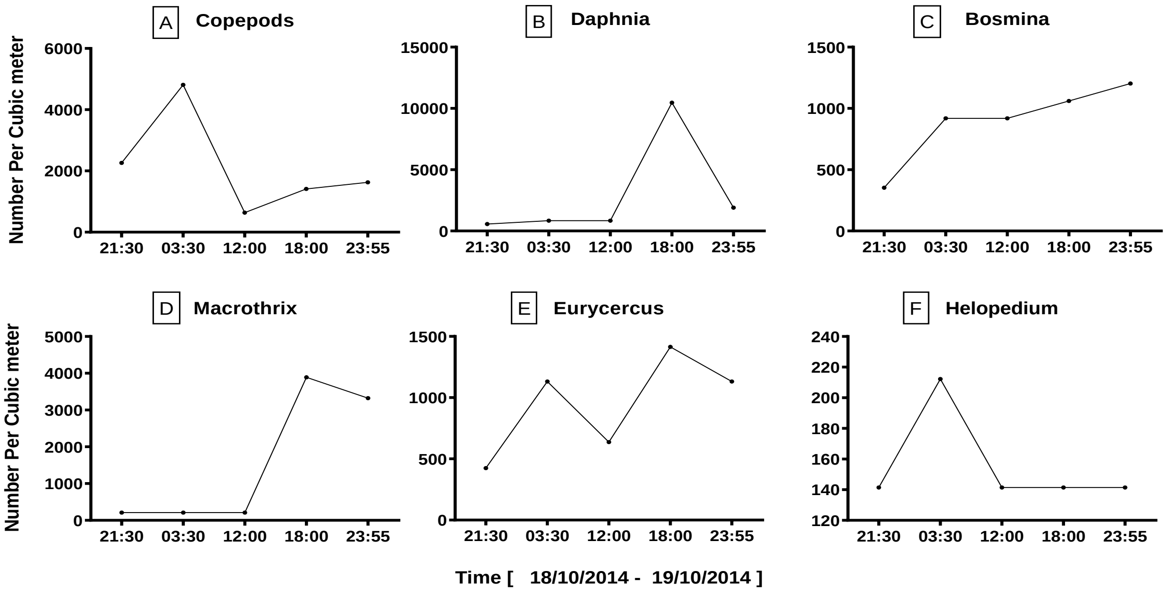

Figure 105- Daily changes in the most common Zooplankton [21:30. 18/10/2014 - 23:55. 19/10/2014].

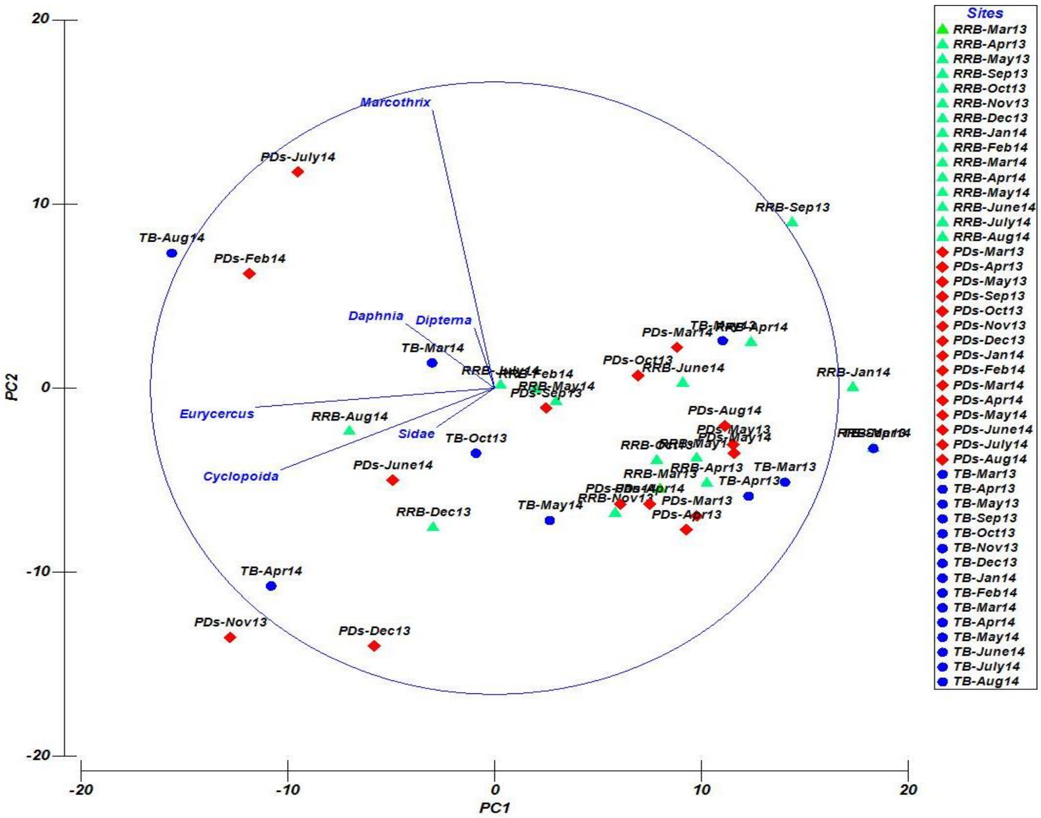

Figure 106: Ordination diagram of phytoplankton community (BIOENV) upstream at Reagent Road Bridge [RRB], Pomona Docks [PDs], and downstream at Turning Basin [TB] between March 2013 –August 2014. The percentage of PC1 and PC2 were 34.5% and 27.2% respectively.

PCA indicate that PC1 (34.5%) and PC2 (27.2%) ($P<0.05$) represents the most significant factors which are contributing to 61.7% of zooplankton community such as *Cycoida, Eurycercus, Macrothrix* and *Daphnia*. The physio-chemical parameters are DO, ammonia, nitrate, phosphate and discharge. Nitrate and phosphate are not likely to directly influence the zooplankton community.

3.2.6.2 Benthic invertebrates

Biodiversity of the benthic invertebrate community was measured using the Shannon-Weiner taxa diversity and Simpson indices (Table16). The overall scores show that biodiversity was low even though there were few sites showed relatively high scores such as Old Trafford Road Bridge (0.9) at September 2014 and during October 2014 at Mark Addy, Regent Road Bridge and Old Trafford Road Bridge as the scores were (0.7, 0.8, 0.6) respectively. The Shannon-Weiner index usually range between 1.5 and 3.5 (Magurran, 1988) and the Simpson 0 (infinitive diversity) -1 (Complete uniformity or no diversity), (Dejong, 2017). The lower values are characteristic of a community dominated by a few pollution-tolerant taxa. Therefore, all sites are indicative of a markedly polluted system.

Table 17: Shannon Weiner (SH-W) and Simpson scores for the benthic invertebrate community at Mark Addy, Regent Road Bridge (RRB), Pomona Docks (PD), Trafford Road Bridge (TRB) and the Turning Basin (TB) between September and December 2014.

Site	15/09/2014		15/10/2014		15/11/2014	15/12/2014		
	SH-W	Simpson	SH-W	Simpson	SH-W	Simpson	SH-W	Simpson
M.A	0.305	0.183	0.72	0.431	0.29	0.123	0.18	0.088
R.R.B	0.361	0.152	0.81	0.4611	0.35	0.146	0.45	0.2218
T.R.B	0.986	0.523	0.60	0.333	0.30	0.138	0.63	0.3018
T.B	0.651	0.357	0.26	0.128	0.17	0.0744	0.41	0.2088

Polluted systems are characterised by a high degree of dominance and in the Irwell/MSC some individual taxa occurred in very large numbers, for instance *Lumbriculidae* which reached 1,200 individuals per colonization sampler. Other relatively common taxa were *Asellidae*, *Gammaridae* and *Erpobdellidae*, figure 108. All these taxa also dominated the community in the Turning Basin as recorded during the analysis of the historical dataset.

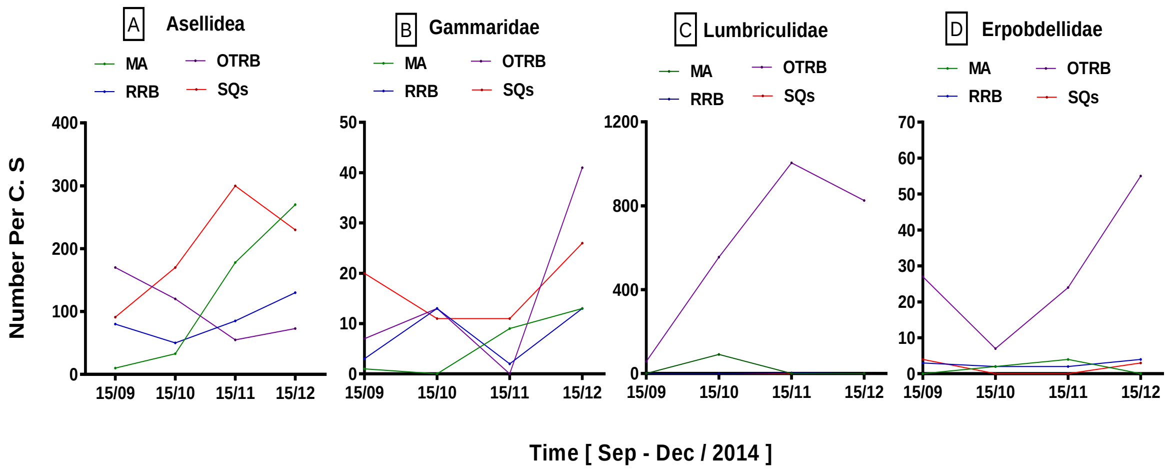

Figure 107- Density of the major groups of benthic invertebrates between August and December 2014.

Statistical analysis showed that there were significant differences between *Asellus aquaticus* and Gammaridae at all sites except Old Trafford Road Bridge where the analysis showed that the difference was only between numbers of Lumbriculidae and Gammaridae. The common trend at most sites were an increase in the number of benthic invertebrates between Septepmber and December, especially Asellidae and Lumbriculidae, except at Regent Road Bridge where total mean benthic invertibrates numbers was stable over most of the sampling period,

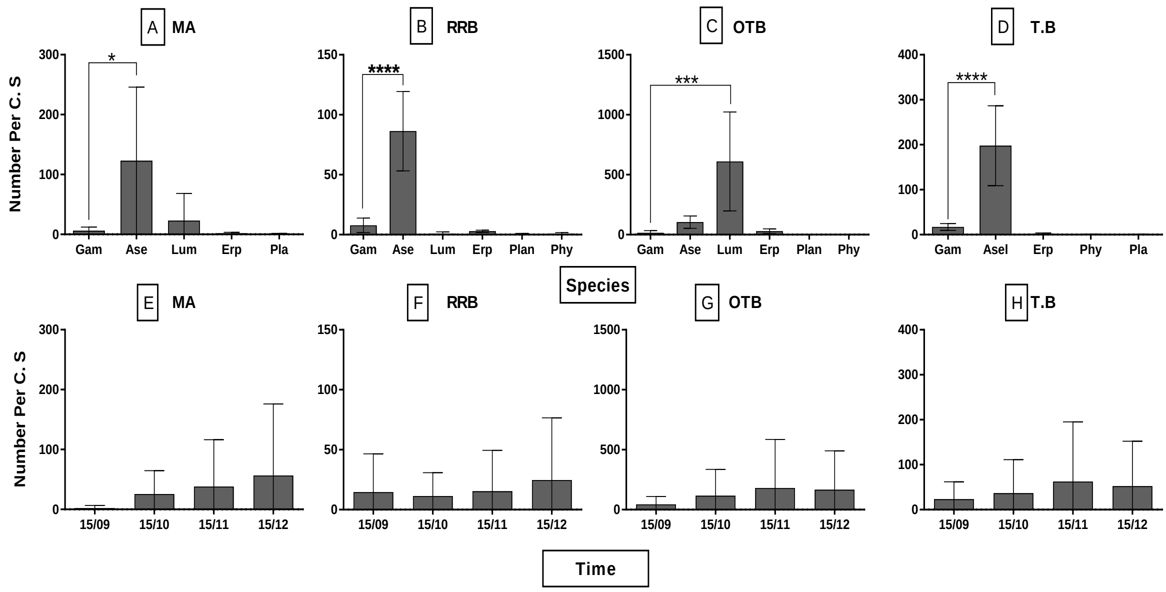

Figure 108- Average (top) and seasonal distribution (bottom) in benthic invertebrates at Mark Addy (M.A), Regent Road Bridge (R.R.B) and Old Trafford Road Bridge (O.T.R.B). Data shown as mean ±SEM, n = 5. Gam = *Gammaridae*, Ase = *Asellidae*, Erp= *Erpobdellidae*. Lum= *Lumbriculidae*. Erp= *Erpobdellidae*. Pla= *Palnirdae*. Phy= *Physidae*.

According to Table 17 that shows both the BMWP and ASPT scores from each site, it is clear that all sites are mostly within the category of either heavily polluted or polluted and impacted system according to Seaby and Henderson (2007).

Table 18: BMWP and ASPT scores for the invertebrate community recorded in the colonisation samplers at four-time intervals between October and December 2014.

Time	15/09/2014		15/10/2014		15/11/2014		15/12/2014	
Sites	*BMWP*	*ASPT*	*BMWP*	*ASPT*	*BMWP*	*ASPT*	*BMWP*	*ASPT*
Mark Addy	9	4.5	12	3	12	4	9	4.5
Regent Rd Bridge	15	3.75	20	4	13	3.25	17	4.25
Old Trafford Rd Bridge	18	3	13	3.25	7	2.33	13	3.25
Turning Basin	22	5.5	12	4	12	4	12	4

From the data in tables 10 and 11above that give flow rate and amount of sediment deposited in the colonisation samplers, the relationship was positive between overall numbers of benthic invertebrates at Trafford Road Bridge and amount of sediment whereas statistical analysis showed no relationship between the benthic invertebrate community and flow velocity.

The benthic invertebrate community at the above sites was compared with the shallow erosional sites of Adelphi Weir. The community was assessed by means of the kick-sampling method as this site was shallow and hence could be sampled directly but was not suitable for the positioning of colonisation samplers. There were two families with a BMWP score 5 of BMWP, Hydropsychidae was found at all sampling times between Sep 2013 and Feb 2014, whereas mayflies (Baetidae) were found only once among other species in Feb 2014. Both the BMWP and ASPT indices indicate that the system is impacted upstream with very low score according to the kick sampling assessment, table19.

Table 20: BMWP and ASPT score of Adelphi Weir according to kick sampling measurements.

Time	BMWP	ASPT
30/09/2013	14	3.5
12/11/2013	12	3
01/12/2013	12	3
19/02/2014	16	3.2

Correlation analysis between the physio-chemical parameters that there are three major groups of correlation (Table 20): as low correlation in red, medium correlation in yellow and high correlation in green. It is very clear that there is high positive correlation between discharge and suspended solids, whereas a negative correlation reflects the effect of discharge on thermal status of the river. In addition, the positive and high correlation between conductivity and temperature supports that temperature is one of the main contributors that affects conductivity in freshwater environments.

Table 21: Correlation values of physio-chemical parameters and benthic invertebrate population

Envi-Factors	BOD	Ammonia	pH	Conductivity	Q	DO	Temp	NO3	PO4.P
BOD									
Ammonia	0.02								
pH	0.3	0.58							
Conductivity	-0.22	-0.36	0.21						
Q	0.44	0.46	0.24	-0.85					
DO	0.25	-0.29	-0.49	-0.64	0.37				
Temp	-0.28	-0.38	0.06	0.97	-0.94	-0.55			
NO3	-0.57	0.19	0.27	0.01	0.1	-0.13	-0.12		
PO4.P	0.16	-0.53	-0.54	-0.05	-0.12	0.51	-0.05	-0.03	
SS	0.61	0.39	0.29	-0.58	0.84	0.01	-0.7	-0.2	-0.17

There was positive relationship with sediment size and conductivity upstream, whereas the community downstream was affected positively by the increase in the amount of sediment per colonisation sampler, ammonia and conductivity, Figure 98.

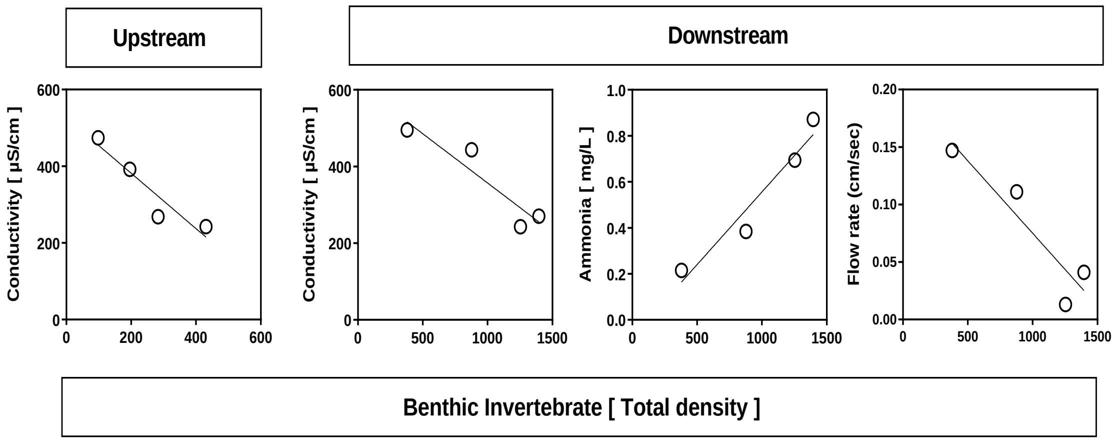

Figure 98 Effect of physio-chemical parameters on the benthic invertebrate community upstream at Mark Addy [MA], and downstream at Old Trafford Road Bridge [OTRB] between Sep-Dec/2014.

Ordination between the benthic invertebrate population and environmental variables was carried out. Principal components for both environmental factors (Table 21A) and biological data (Table 21B) showed that the first PCA was the most significant ($p<0.05$) and contributes 41% variation to the benthic invertebrate communities and population density as Eigen values shown in Table 22. The physical-chemical variables were therefore conductivity, discharge and suspended solids (PC1) and pH, ammonia, DO and PO_4-P (PC2; contributing 27% variation). The families *Lumbricidae* and *Asselidae* were the dominant invertebrates in the colonisation samplers.

Table 22: Environmental variables (A) and benthic invertebrate (B) ordination ($p<0.05$) with the principal components. The major contributory variables are emboldened.

(A)

Variable	***PC1***	***PC2***	***PC3***	***PC4***	***PC5***
BOD	0.256	-0.049	0.617	0.322	-0.206
Ammonia	0.231	**0.430**	-0.145	-0.19	-0.312
pH	0.074	**0.519**	0.144	0.457	-0.345
Conductivity	**-0.446**	0.155	0.216	0.239	0.083
Discharge	**0.484**	0.055	-0.073	0.105	0.105
DO	0.22	**-0.473**	-0.108	0.052	-0.591
Temp	-0.47	0.092	0.211	0.023	-0.08
NO_3	-0.03	0.175	-0.641	0.541	0.101
PO_4-P	-0.029	**-0.485**	0.006	0.535	0.171
SS	**0.415**	0.135	0.242	0.035	0.572

(B)

Variable	***PC1***	***PC2***	***PC3***	***PC4***	***PC5***
Gammaridae	0.012	-0.055	-0.593	0.802	0.037
Asellidae	-0.08	**-0.995**	0.049	-0.031	-0.002
Lumbriculidae	**0.996**	-0.078	0.036	0.006	0.001
Erpobdellidae	0.031	-0.023	-0.803	-0.593	-0.04
Planariidae	0	0	0	0	0
Chironomidae	0	-0.001	-0.007	-0.038	0.706
Hydropsychidae	0	0	0	0	0
Baetidae	0	0	0	0	0
Physidae	0	-0.001	-0.007	-0.038	0.706

Table 23: Eigen values, percentage variation and cumulative percentage variation for BioEnv analysis of combination data of environmental factors and benthic invertebrate population.

PC	*Eigenvalues*	*%Variation*	*Cumulative % variation*
1	***4.08***	***40.8***	***40.8***
2	***2.67***	***26.7***	***67.5***
3	***1.64***	***16.4***	***83.8***
4	***0.783***	***7.8***	***91.7***
5	***0.464***	***4.6***	***96.3***

Two sites were selected to represent the studied river reach, the Mark Addy for the upstream reach and Old Trafford Road Bridge for Upper Manchester Ship Canal and Turning Basin. The first PCA accounted for more than 40% of the variables and presented in Figure110.

Table 24: Correlation between physio-chemical variables and benthic invertebrate populations by using BIOENV analysis.

No	*Correlation*	*N.of.V*	*Variables*
1	0.435	**3**	***pH,temperature, PO_4-P***
2	0.421	2	***Ammonia, PO_4-P***
3	0.413	1	PO4-P
4	0.355	3	Ammonia, discharge, PO_4-P
5	0.346	3	Ammonia, conductivity, PO4-P
6	0.337	4	Ammonia, conductivity, DO, PO4-P
7	0.337	4	Ammonia, conductivity, temperature, PO4-P
8	0.321	4	Ammonia, discharge, temperature, PO4-P
9	0.304	2	Temperature, PO4-P
10	0.295	3	Ammonia, NO3, PO4-P

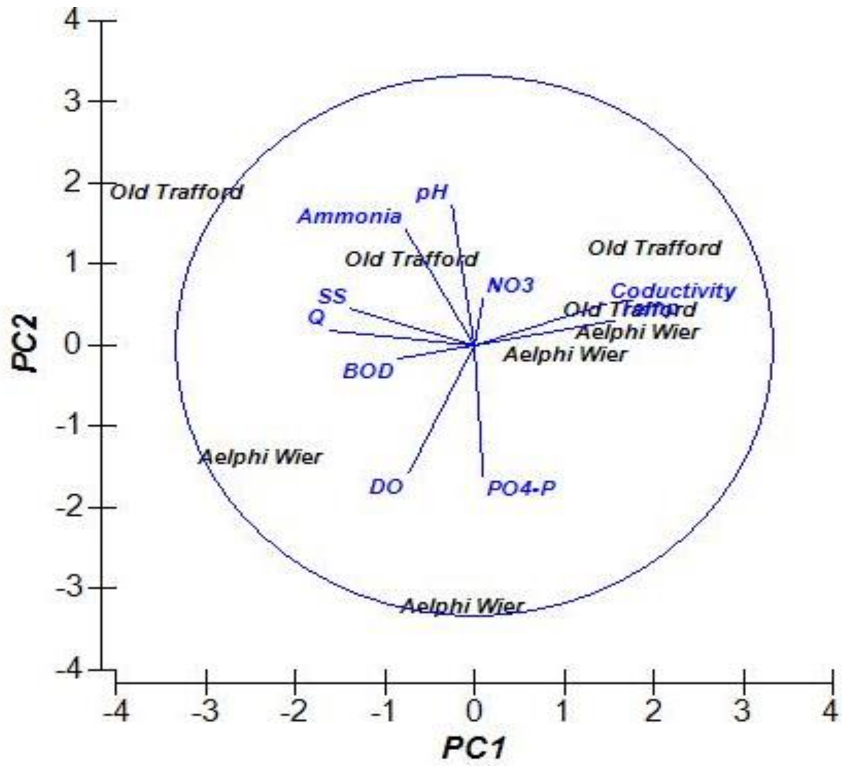

Figure 109: Ordination diagram of physio-chemical variables and benthic invertebrate community (BIOENV) upstream at Mark Addy [MA], and downstream at Old Trafford Road Bridge [OTRB] between Sep-Dec/2014. PC1 ($P<0.05$) represents conductivity, discharge, and suspended solids that contributes 41% for benthic invertebrate variation, and PC2 represents pH, ammonia, DO and PO_4-P which contribute just for 27%. Dominant families were *Lumbricidae* and *Asselidae.*

Environmental variables in the PC matrix (Figure111) were combined with the benthic invertebrate community in a BIOENV analysis to identify the key physical-chemical variables affecting the community. From the correlation matrix, the key variables which best influenced the community were pH, temperature, ammonia and P-PO_4 (Table 23). Some of these parameters are likely to be surrogates for other variables as pH and temperature were both within the tolerance limits for benthic invertebrates (see sections 3.2.2/3.2.3/3.2.4)

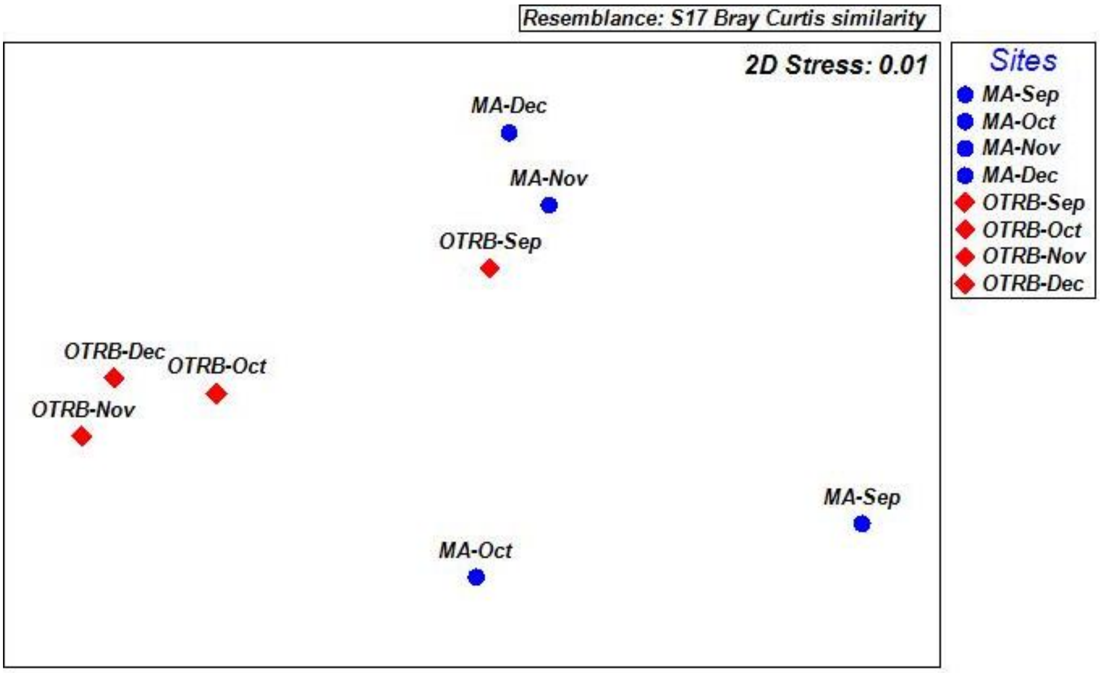

Figure 110: Multi-Dimensional scaling (MDS) ordination plots of benthic invertebrate populations at Mark Addy [MA] and Old Trafford Road Bridge [OTRB] between Sep-Dec/2014.

3.3 Short term change in water quality and ecology

The short-term survey took place between July and October 2014 to investigate the effect of changes in precipitation on discharge and, hence directly and indirectly on water quality. Each measurement was usually taken either the day before or the day after a storm event and sometime during the event where sites were accessible.

3.3.1 Episodic discharge

According to Figure112, it is very clear that there were marked temporal changes in discharge over the period of the study. The average discharge was more than 12m^3/sec upstream at the Mark Addy site, and more than 17m^3/sec at Pomona Docks due to the inputs of the rivers Irk and Medlock. The difference between upstream and downstream is around 5 m^3/sec and the major contribution to discharge at the MSC is from the River Irwell (8 m^3/sec), while the contribution of both the Irk and Medlock is between 1 and 2 m^3/sec, figure 113. As expected, statistical analysis showed a highly significant difference between the two sites, figure 114.

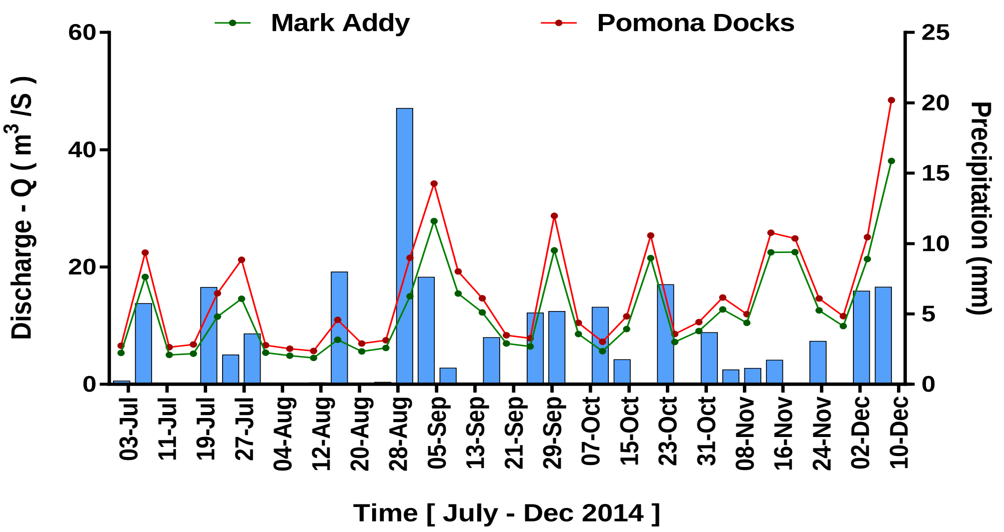

Figure 111- Discharge at Mark Addy and Pomona Docks between July and December 2014 (left Y axis) plus rainfall average over the same period (right Y axis). Data shown as mean ± SE. p<0.05 indicates two sites statistically are not significantly different. n=23.

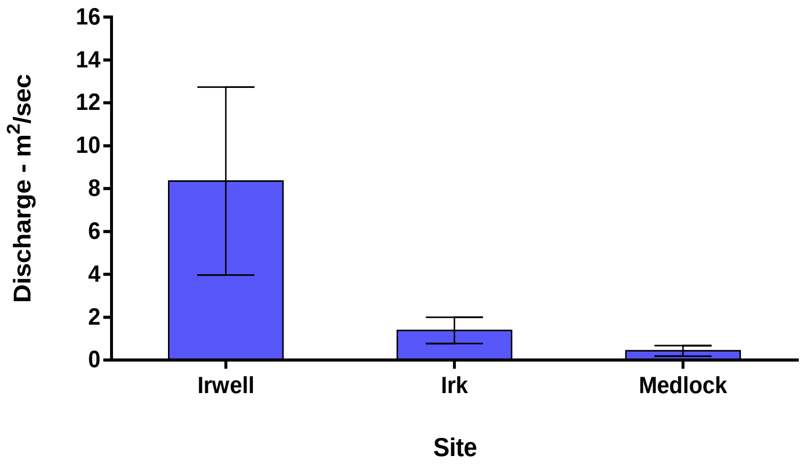

Figure 112- The contribution of discharge [Q] at the Irwell, Irk and Medlock to the overall mean discharge in the system between July and December 2014. Data shown as mean ± SE. $p<0.05$ indicates that discharge from the Irwell is significantly higher than the other two rivers, whereas there was no significant difference in discharge between the Irk and Medlock. n=33.

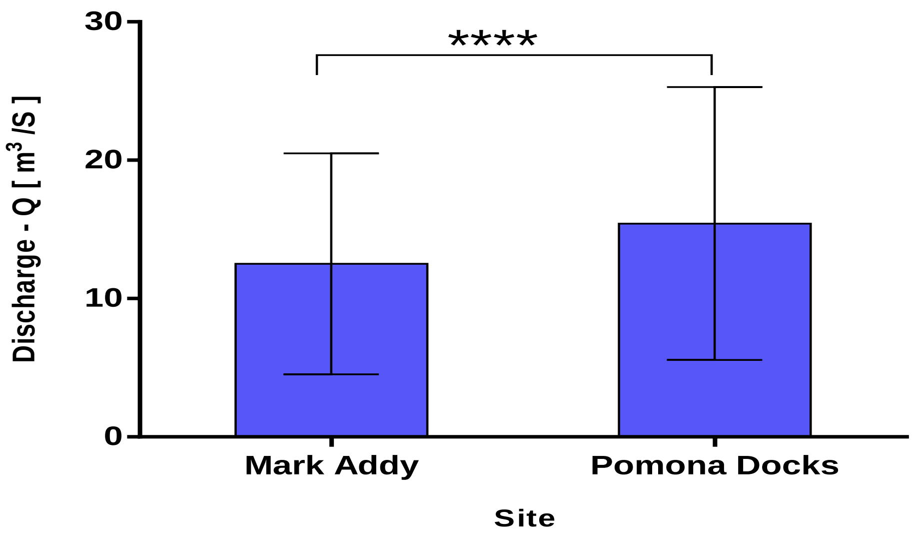

Figure 113- Average of discharge [Q] at Mark Addy and Pomona Docks between July and December 2014. Data shown as mean ± SE. $p<0.05$ indicates two sites statistically are significantly different. n=33.

3.3.2 Flux

The main goal of the short term survey is to investigate the effect of discharge on the main waterquality parameters, specifically to what extent discharge affected the biological oxygen demand, and the transport of suspended solids and nutrients. The influence of short-term changes in discharge on phytoplankton populations was also examined.

3.3.2.1 Biological oxygen demand

The average of flux of biodegradable organic material as indicated by BOD ranged between 0and 12000 kg/day. Generally, the BODwas changeable according discharge and therefore rainfall which has significant effect at both Mark Addy and the bottom of the water column at Pomona Docks, Figure 115.

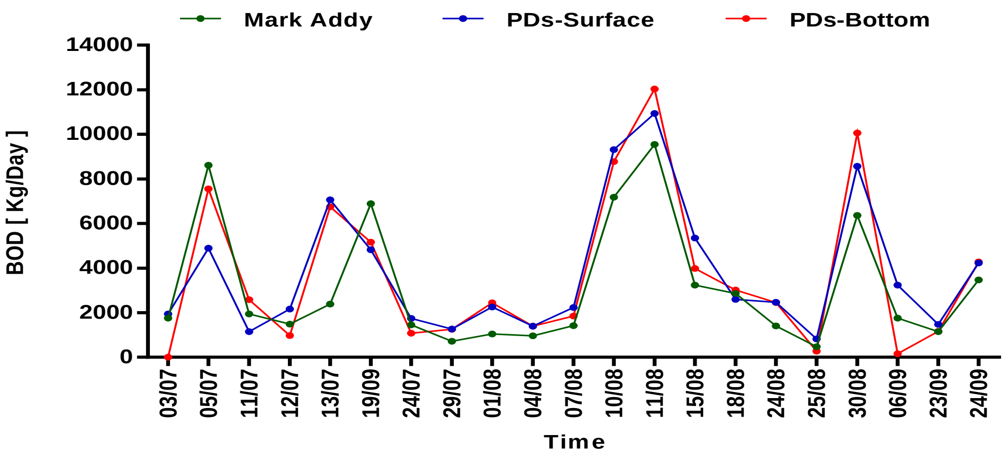

Figure 114- Flux of organic material as indicated by BOD at Mark Addy, Pomona Docks [PDs] surface and bottom between July and September 2014. The [R2= 0.831, 0.0770, 0.861] were for Mark Addy and Pomona Docks surface and bottom respectively. P<0.05- Significant for MA and PDs-B. n= 21.

3.3.2.2 Suspended solids

Average suspended solids ranged between 0 and 40000 kg/ dayand the flux of suspended solids was higher at Pomona. Two peaks in suspended solids flux were recorded; the first was between the 10^{th} and the 11^{th} of August, and the other peak was on the 30^{th} of August. These peaks corresponded to high discharge and rainfall.

Statistical analysis showed significant effect of discharge on suspended solids flux at all sites especially Pomona Docks Bottom, figure 116.

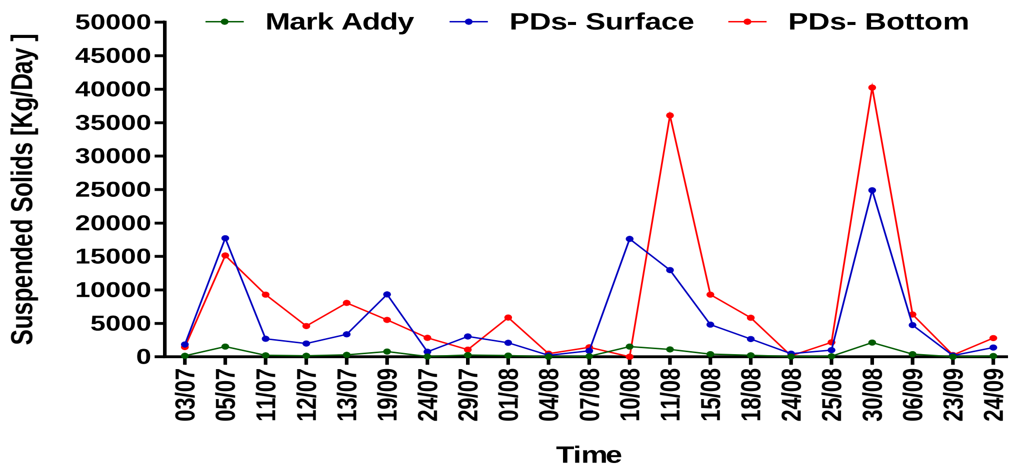

Figure 115- - - Flux of suspended solids at Mark Addy, Pomona Docks [PDs] surface and bottom between July and September 2014. The [R^2= 0.504, 0.661, 0.750] were for Mark Addy and Pomona Docks surface and bottom respectively. P<0.05-Significant for all sites. n= 21

3.3.2.3 Nutrients

The flux of phosphorous and nitrogen is shown in Figure 117. Phosphorous flux was between 980 and 1451 mg/sec. The level of phosphorus was changeable between 3rd of July and 25th of August, ranging between 20 and 250 kg/day; afterward there was a very large peak on 30th of August in which reached over 500 kg phosphorus/day. Generally, statistical analysis showed significant effect of discharge on phosphorous level between Mark Addy and Pomona Docks. The flux of nitrate ranged between 500 and 8000 kg/day. The trend was mostly changeable with no noticeable differences between all sites. Statistically, there was significant effect of discharge on nitrogen flux at all sites especially at Mark Addy. Despite the slight variation between the trends of both phosphorous and nitrate levels, it can be seen that most of peaks are coincidence and in similar patterns.

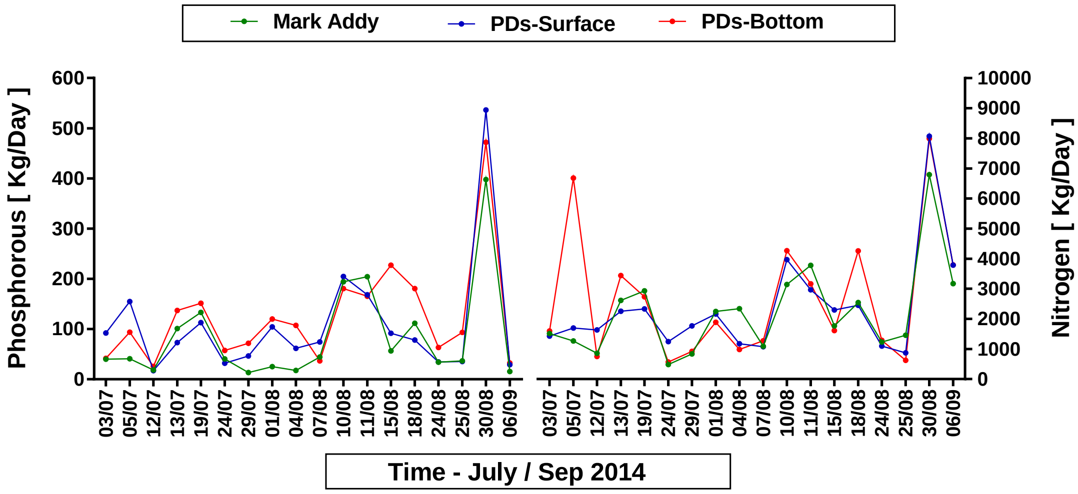

Figure 116- Nutrient flux at Mark Addy and Pomona Docks [PDs] surface and bottom between July and September 2014. R^2 for Nitrogen = 0.494, 0.390, 0.525 for Mark Addy and Pomona Docks surface and bottom respectively; (Phosphorous = 0.614, 0.488, 0.493 were for Mark Addy and Pomona Docks surface and bottom respectively). P<0.05- shows significant effect of discharge on nitrogen and phosphorous levels for all sites. n =18.

3.3.3 Physical and chemical parameters

3.3.3.1 pH

The general trend in pH is a slight overall decrease from July to December 2014, Figure118. There were two small troughs; the largest was on first of August where pH fell to 5.5 at both sites and depths. The other decrease was on the 16th of October were the pH of upstream and downstream surface fell to 6.5 while the trend remain the same at the bottom of the Pomona docks site. With regard to the statistical analysis, Figure 119 shows that there was a slight difference between upstream sites at Mark Addy where pH I was noticeably lower than downstream at Pomona Docks. The general average was similar, mainly 7.5, but wider standard deviation at Mark Addy. Statistics showed also slight differences during different seasons.

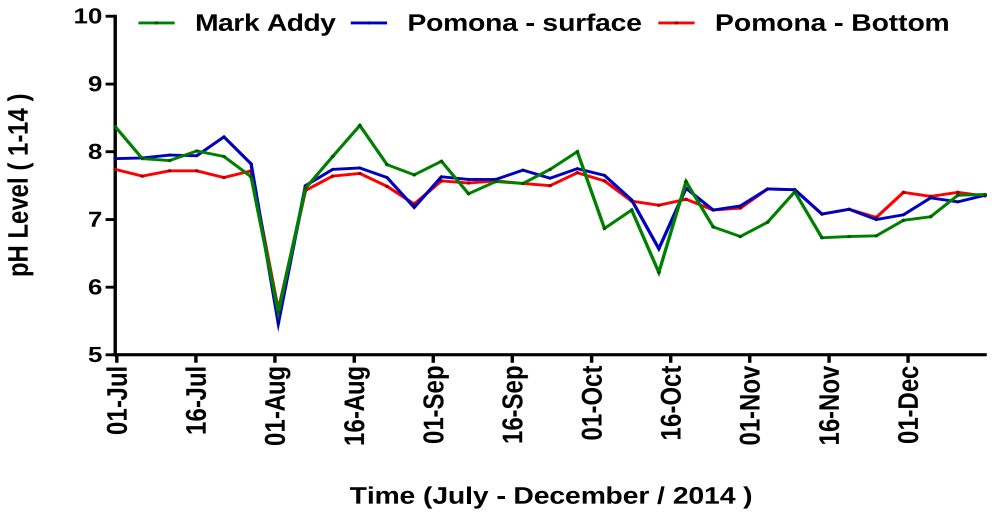

Figure 118- Short term changes in pH at Mark Addy, Pomona Docks Surface and Pomona Docks bottom between July and December 2014. The green line presents the values of Mark Addy. n=33.

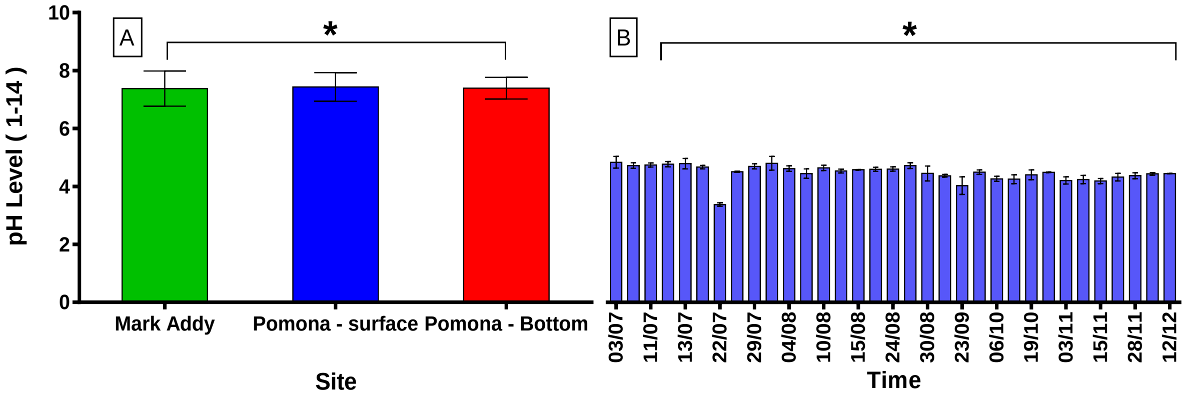

Figure 119: Average of pH level at Mark Addy, Pomona Docks Surface and bottom between July and December 2014. (P < 0.05) showed significant differences between sites n=3, and along different seasons, n = 33.

3.3.3.2 Temperature

The following graph, figure 120, shows typical overall decrease in temperature between a summer high of 20^{0}C and winter low of nearly 5^{0}C. The trend fluctuated slightly with a peak on 1st August where the water temperature reached just over 25 ^{0}C. As expected, there was a marked similarity between upstream and downstream both surface and bottom at the Pomona Docks site showing that the system is well-mixed and there is no indication for thermal stratification, Figure 121.

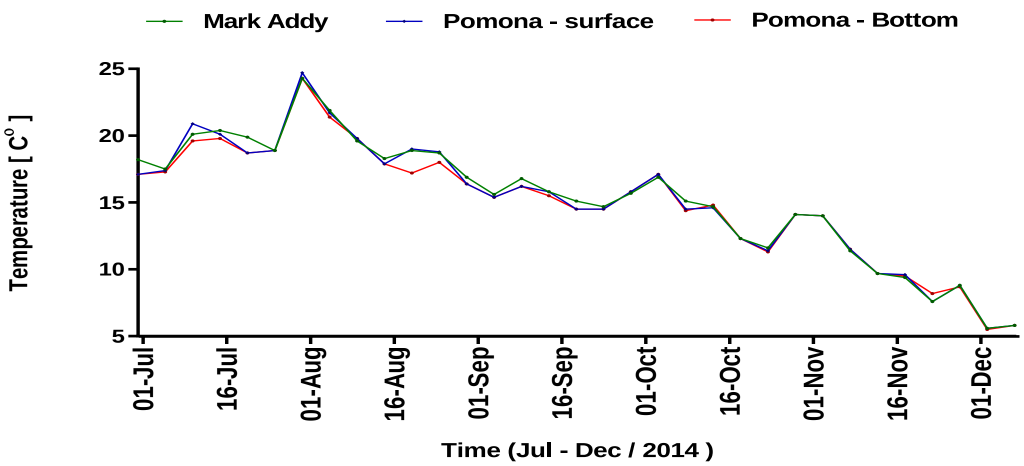

Figure 120- Short term changes in temperature (C^0) at Mark Addy and Pomona Docks Surface and bottom between July and December 2014. n=33

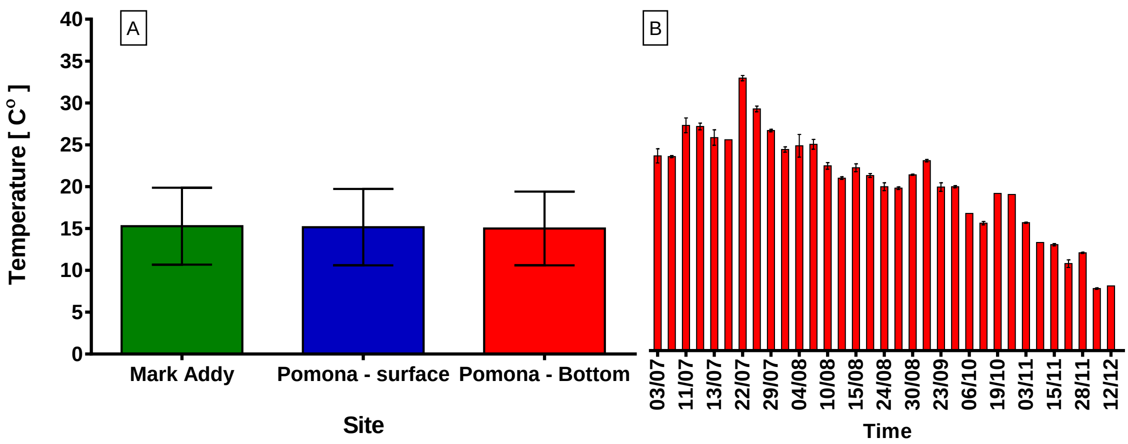

Figure 121- Average temperature (C^0) at Mark Addy, Pomona Docks Surface and bottom between July and December 2014. No significant differences ($P < 0.05$) both between sites and over the sampling period, n = 33.

3.3.3.3 Conductivity

Conductivity fluctuated between 300 µS/cm and 600 µS/cm which considered by far over the normal range of (>70 - 250µS/cm according to UK-WFD/2015), figure 122. In addition, there were a slight variation between upstream and downstream and conductivity at Pomona Docks surface and bottom is similar. There were no significant ($P < 0.05$) differences between sites and between seasons, figure 123.

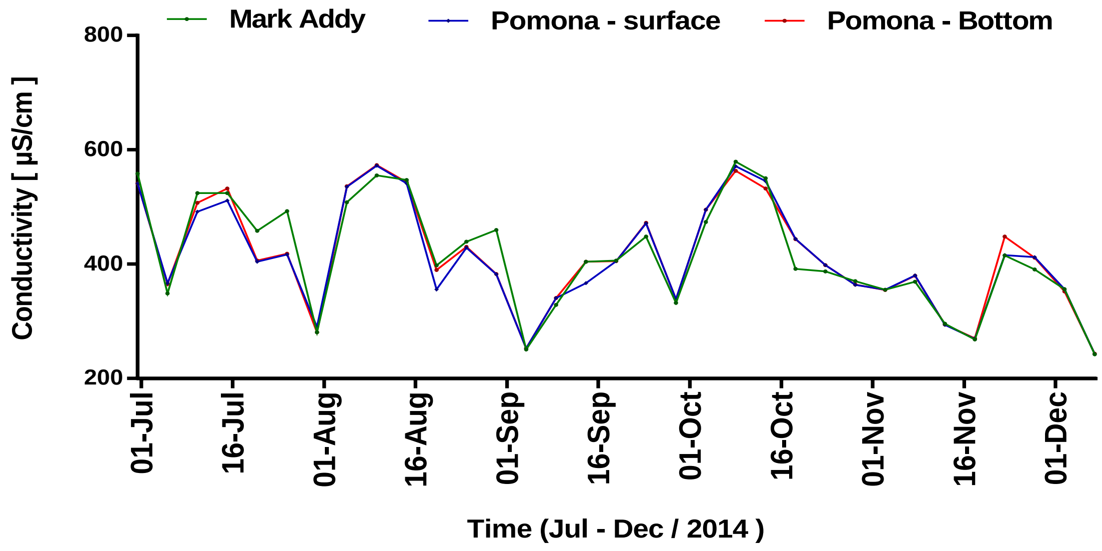

Figure 122- Short term changes in Conductivity at Mark Addy and Pomona Docks Surface and bottom between July and December 2014. The green line symbolizes the values of Mark Addy, blue lines for Pomona Docks surface and red for Pomona Docks bottom. n=33.

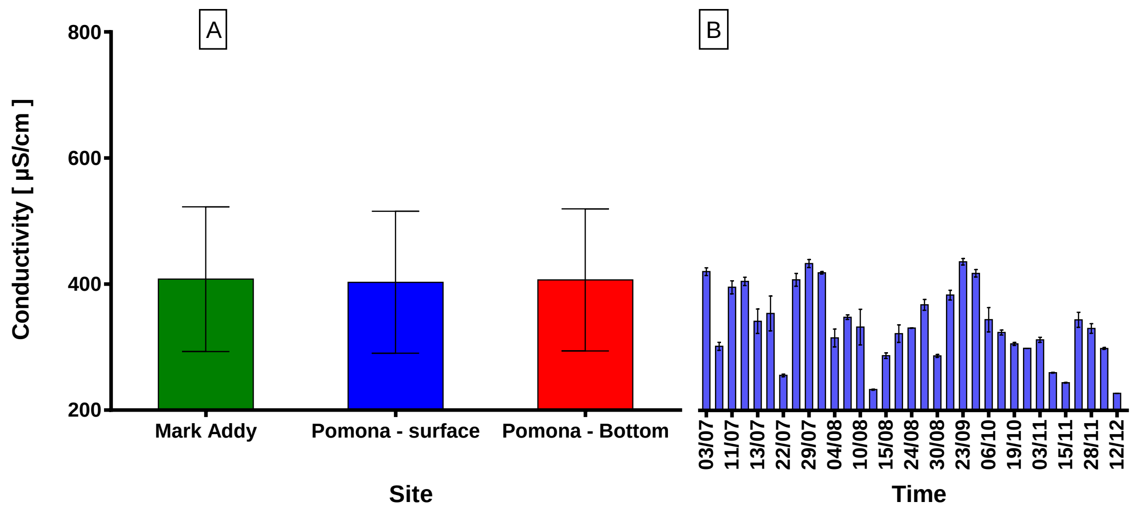

Figure 123- The average conductivity at Mark Addy, Pomona Docks Surface and bottom between July and December 2014. (P < 0.05) showed no significant differences both between sites and during time scale, n = 33.

3.3.3.4 Dissolved oxygen

Dissolved oxygen (DO) ranged between 5 mg/l and 10 mg/l from July to the end of September. There was a marked increase in early October at Mark Addy and both at the surface and bottom of the water column at Pomona Docks as values reached 20 mg/l at Mark Addy, 15 mg/l at the bottom of Pomona Docks and just over 10 mg/l at surface. The trend for all sites was mostly stable with a slightly increase during autumn and early winter of 2014, figure 124. Although the range was between 5 mg/l to 20 mg/l, the average of DO was mainly 10 mg/l upstream and just under this value at lower part of the system. Do was higher upstream than downstream, and the DO at the bottom of Pomona Docks was lower than the surface. Statistically, there was a significant difference between upstream at Mark Addy and downstream at Pomona Docks surface and with seasons, figure 125.

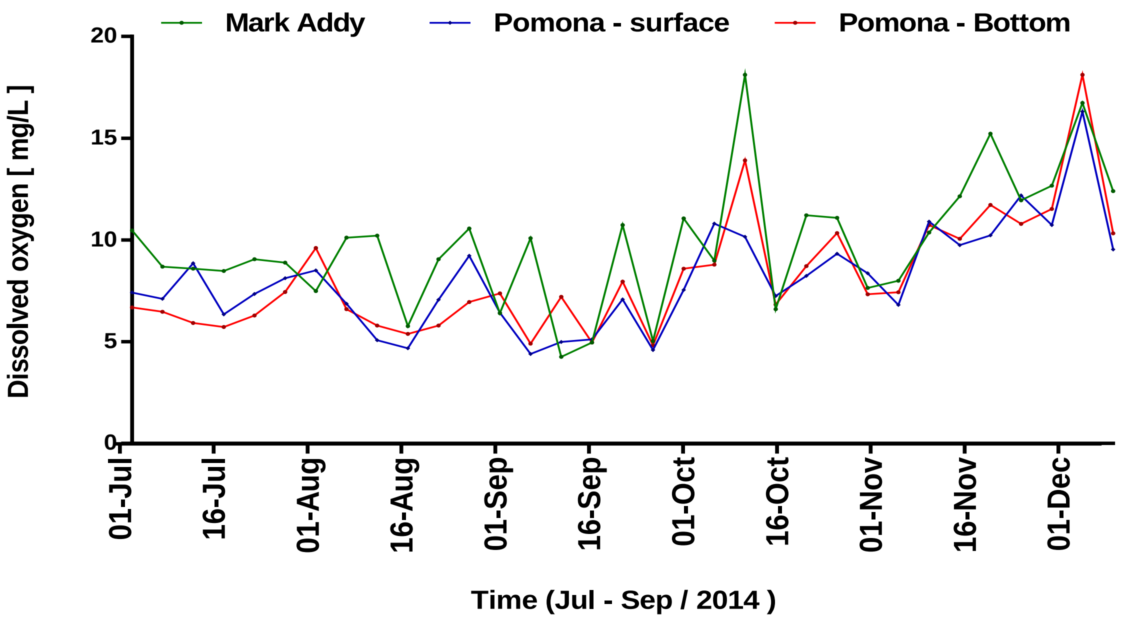

Figure 124- Sort term changes in Dissolved oxygen at Mark Addy, Pomona Docks Surface and Pomona Docks bottom between July and December 2014. The values performed in green line for Mark Addy, blue lines for Pomona Docks surface and red for Pomona Docks bottom. n=33.

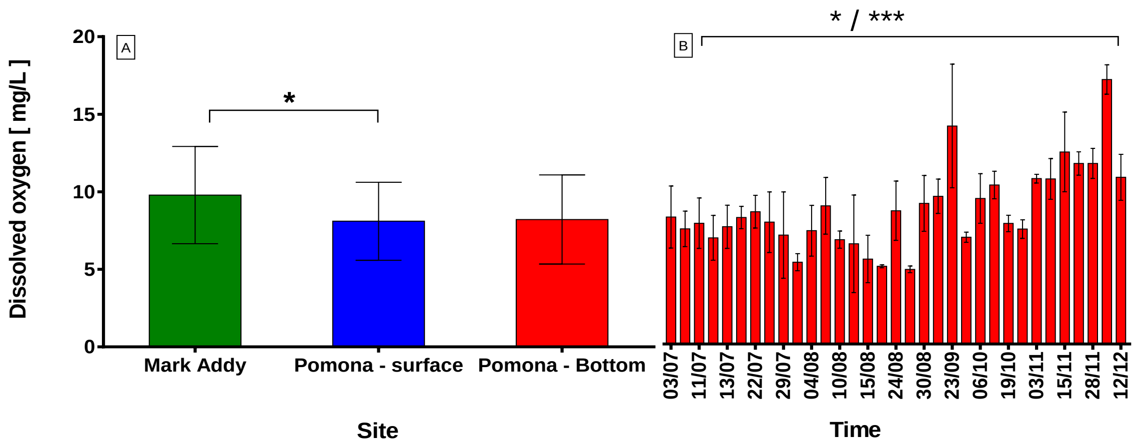

Figure 125- Average Dissolved oxygen at Mark Addy, Pomona Docks Surface and bottom between July and December 2014. (P < 0.05) showed significant differences between different sites and for time. n = 33.

3.3.3.5 Biological oxygen demand (BOD)

Biological oxygen demand varied between 0.2 mg/l and 6 mg/l with an average of 3 mg/l, figure 126. Therefore, BOD varied from 'Very Good' to Fairly Good' of the period the short-term study according to the Chemical GQA. There trend was observed at all sites and BOD peaked at just under 6 mg/l on the 16th August and dropped markedly on 12th September to just over 0.5 mg/l. However, according to the statistical analysis, there was no overall difference between either sites and with time, Figure127.

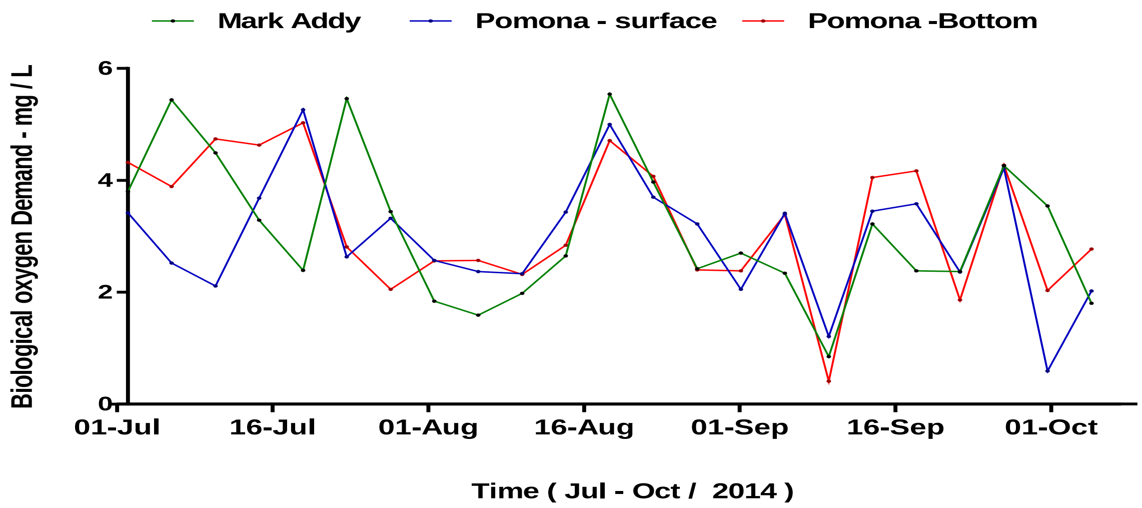

Figure 126- Short term changes in biological oxygen demand (BOD) at Mark Addy, Pomona Docks Surface and bottom between July and December 2014. The green line represents the values of Mark Addy, blue lines for Pomona Docks surface and red for Pomona Docks bottom. n=21.

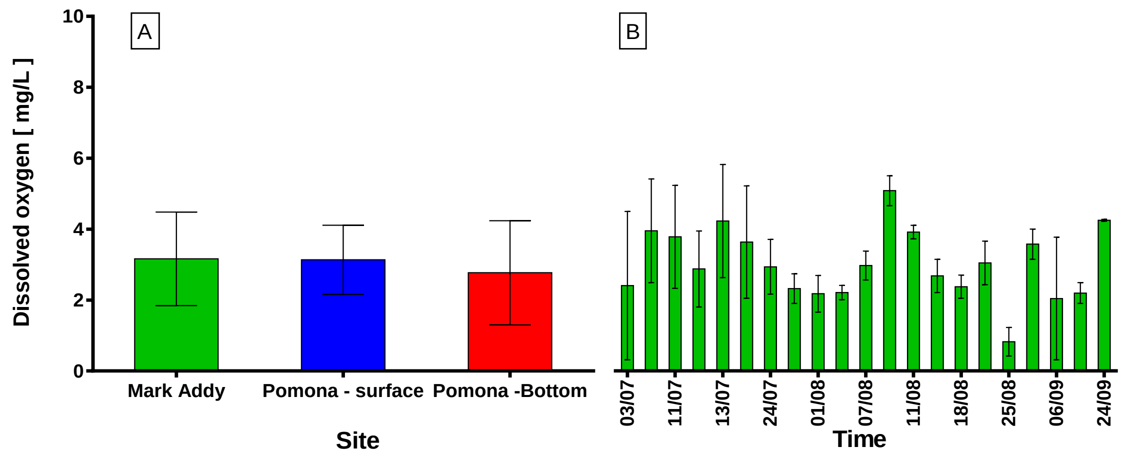

Figure 127- Average Biological Oxygen Demand (BOD) at Mark Addy, Pomona Docks Surface and Pomona Docks bottom between July and December 2014. ($P < 0.05$) showed no significant differences both between sites and during time scale, n = 21.

3.3.3.6 Chlorophyll-a

The range of chlorophyll-a was normally between 5 µg/l and 20 µg/l, Figure 128. There was a significant peak at the Mark Addy site on 8th of August, where the concentration reached nearly 60µg/l. The other peak occurred at all sites on the 10th of October where the concentration was once again around 60µg/l. Chlorophyll-a concentrations are therefore once again generally within the mesotrophic range of

2.6-20µg/l according to Carlson (1996) although occasional values are indicative of eutrophic conditions (20-56 µg/l) and occasional hypereutrophy were observed at the Mark Addy (56 µg/l-plus). The reason for this exceptionally high value is unclear as discharge was not high and therefore is unlikely to be due to suspension of epibenthic algae.

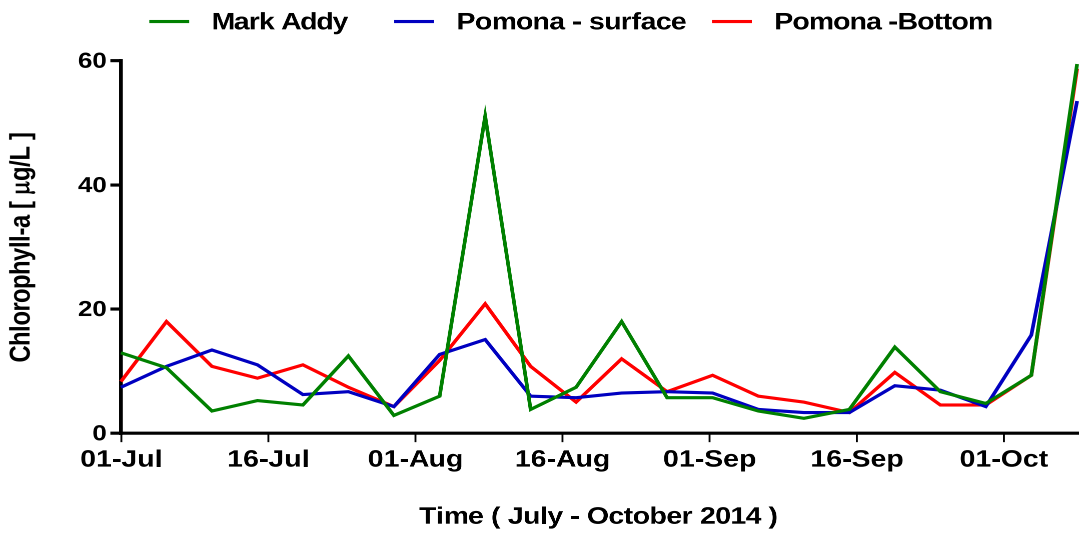

Figure 128- Short term changes in Chlorophyll-a at Mark Addy, Pomona Docks Surface and Pomona Docks bottom between July and December 2014. The green line symbolizes the values of Mark Addy, blue lines for Pomona Docks surface and red for Pomona Docks bottom. n=22.

Statistical analysis showed that the mean chlorophyll-a concentration was the same between sites. However, there were slightly ($P < 0.05$) significant differences between a few periods of time, the first was during 24th of July and 11th of August, the second time was between 25th of August and 23th of September 2014,figure 129.

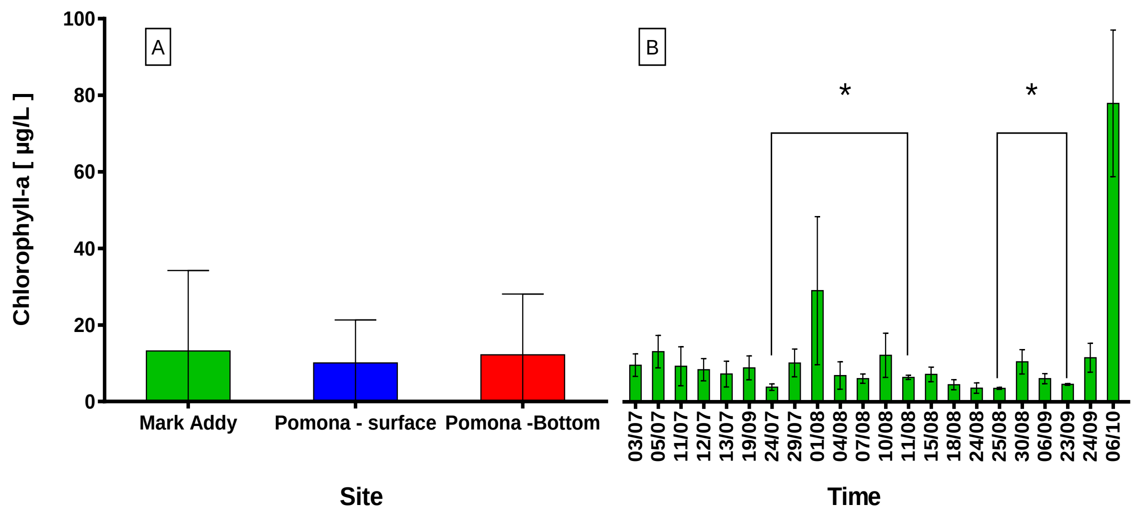

Figure 129- Average values plus and statistical analysis chlorophyll-a at Mark Addy, Pomona Docks Surface and bottom between July and December 2014. (P < 0.05) showed no significant differences both between sites but there were a significant difference a long time, n = 22.

The relationship between phosphorous and chlorophyll-a is shown in figure 130 where statistical analysis showed no significant relationship between nutrients and chlorophyll-a.

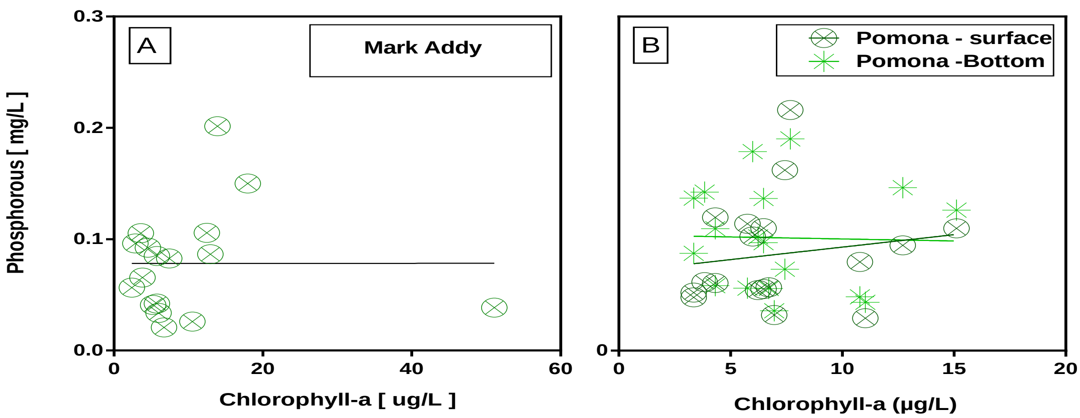

Figure 130- Relationship between phosphorous concentration and chlorophyll-a concentration at Mark Addy, Pomona Docks surface and Pomona Docks bottom. The green circles symbolize the values for of surface upstream [A] at Mark Addy and downstream [B] Pomona Docks, the green stars for the values of bottom level of Pomona Docks. Statistically there are no significant effects at all sites [Square (Mark Addy=0.028, Pomona Docks =0.023 and 0.00062 for surface and bottom respectively)]. N = 18.

3.3.4 Nitrogen [NO_3-N], phosphorous [PO_4- P] and ammonia

Dissolved nitrogen (DN) as NO_3-N was between1 mg/l and 3 mg/l with an average of around 2 mg/l at all sites. On the other hand, total nitrogen (TN) ranged between 2 mg/l with an average of 3.5 mg/l for the Mark Addy site and 4 mg/l for Pomona Dock both at the surface and bottom, figure 131. Upstream and downstream DN values were similar with only two significant peaks; one at the Mark Addy site on the 2nd September where the concentration reached almost 6 mg/l, and another peak at Pomona Docks surface where the level reached almost 4gm/l. There was a close complete coincidence between Pomona Docks surface and bottom with a significant peak on 25th September where the concentration reached nearly 7 mg/l, while the amount at Mark Addy upstream was lower at just over 2 mg/l. As can be seen from figure 132 there was a significant difference between mean dissolved nitrogen (DN) and total nitrogen (TN) at both sites indicating that DN only made up to an average of 2 mg/l and 4 mg/l respectively of the TN. However, there was no difference in either DN or TN between sites. Throughout the short-term study NO_3-N remained 'Low' (>5-10 mg/l) or 'Very Low' (<5 mg/l) according to the Nitrate Grade GQA.

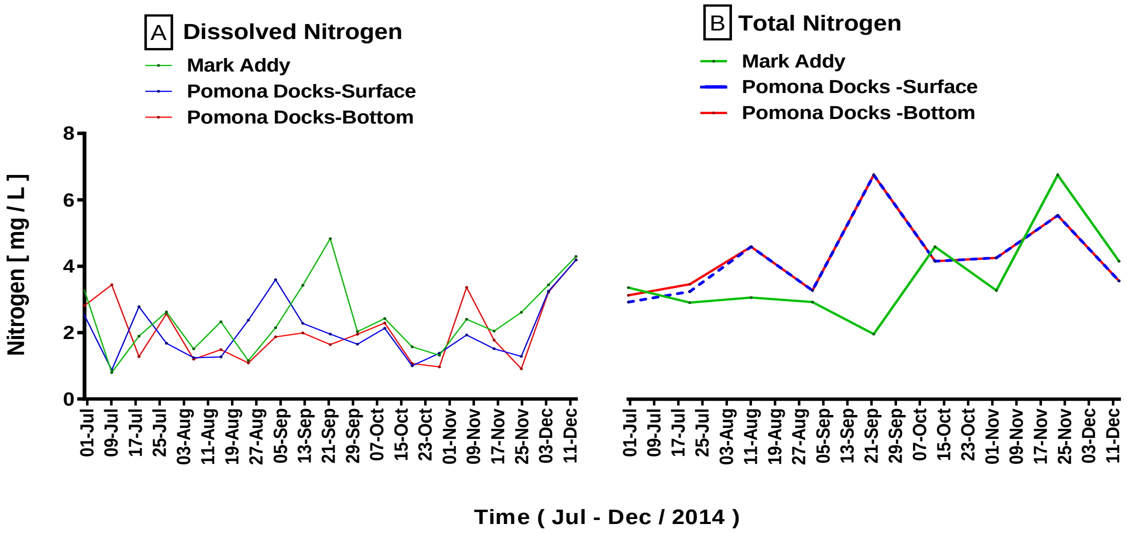

Figure 131- Short term changes Dissolved Nitrogen (DN) as NO_3-N and Total Nitrogen (TN) at Mark Addy, Pomona Docks Surface and Pomona Docks bottom between July and December 2014. The green line shows the values for the Mark Addy, blue lines for Pomona Docks surface and red for Pomona Docks bottom. n=33.

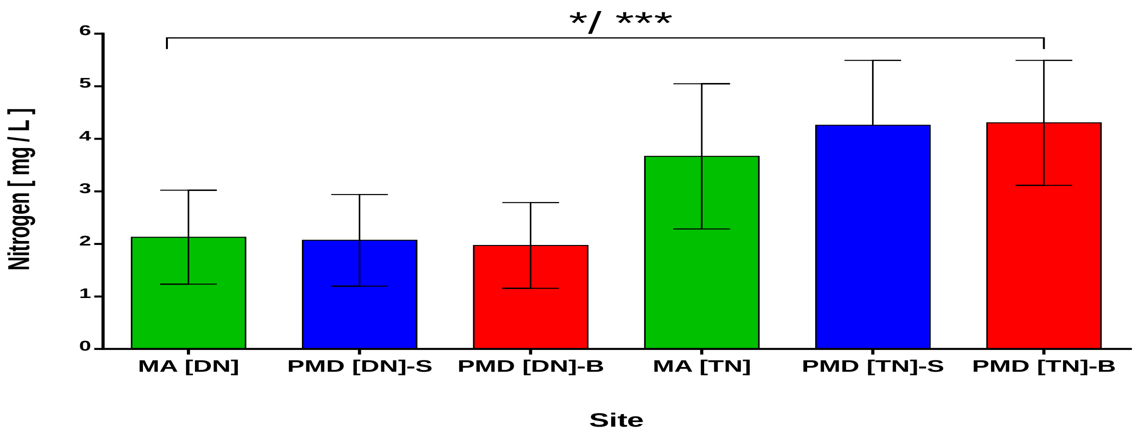

Figure 132- The Average Dissolved (DN) and Total Nitrogen (TN) at Mark Addy, Pomona Docks Surface and Pomona Docks bottom between July and December 2014. P < 0.0001 showed a significant difference between sites for dissolved (DN) and total Nitrogen (TN). There were no differences between site and time for (DN, TN). n = 6.

The range of dissolved phosphorous (DN) was between not detectable and 0.25 mg/l with an average of 0.025 mg/l at all sites. Such large changes indicate a shift of four grades from 'Very Low' (<0.02 mg/l) to 'Very High' (>0.2-1.0 mg/l) according to the Phosphate Grade GQA. Total phosphorous (TN) ranged between 0.25 mg/l and 2.5 mg/l with an average of 1.5 mg/l at the Mark Addy and 0.25 mg/l at Pomona Docks both at the surface and bottom, Figure 133. There were changeable patterns for upstream and downstream concentrations of DN with large fluctuations although the bottom of the water column at the Pomona Docks site was higher than other sites., Total phosphorous in the system upstream was higher than downstream with two significant peaks, one on 15th of October and the other on 3rd December. The amount of total phosphorous downstream was higher at the bottom compared to the surface. There were significant differences between total phosphorous at Mark Addy and Pomona Docks; also, there were significant differences between total phosphorous level at Mark Addy and all sites with regard to dissolved phosphorous, figure 134.

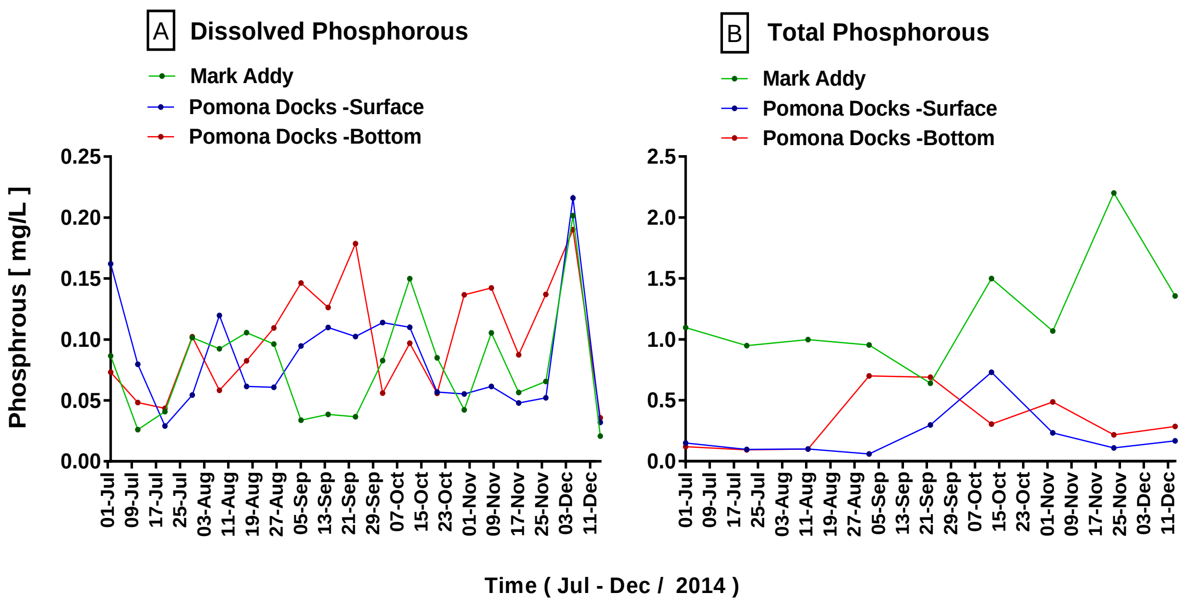

Figure 133- Short term changes in Dissolved phosphorous (DP as PO_4-P) and Total phosphorous (TP) at Mark Addy, Pomona Docks Surface and Pomona Docks bottom between July and December 2014. n=33.

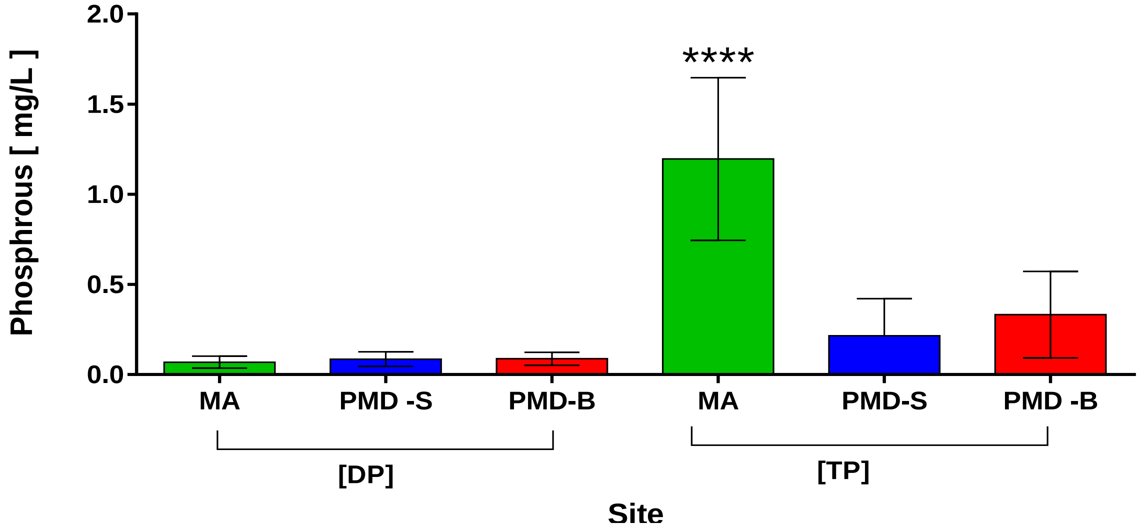

Figure 134- Average (DP as PO_4-P) and total phosphorous (TP) at Mark Addy, Pomona Docks Surface and Pomona Docks bottom between July and December 2014. Mark Addy [TP], $p < 0.0001$, level showed significant difference between (DP-TP) comparing with all other sites. n = 6.

Ammonia ranged between 0.2 mg/l and 2 mg/l with an average of 0.5 mg/l for all sites, figure 135. The trend was mostly a decrease over time but with a significant peak between 23rd and 30th of July that reached 2 mg/l for the bottom of the water column and 1.75 mg/l for the surface of the Pomona Dock site and 1.5 mg/l at Mark Addy. There were no differences between sites, but there were significant seasonal changes of the mean ammonia level in the system at all sites, figure 136.

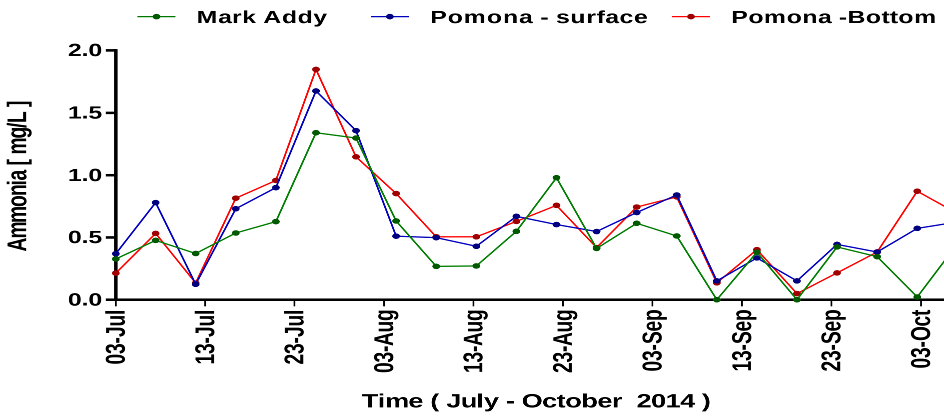

Figure 135- Short term changes in ammonia at Mark Addy, Pomona Docks surface and Pomona Docks bottom between July and October 2014. The green line shows the values for Mark Addy, blue lines for Pomona Docks surface and red for Pomona Docks bottom. n=22.

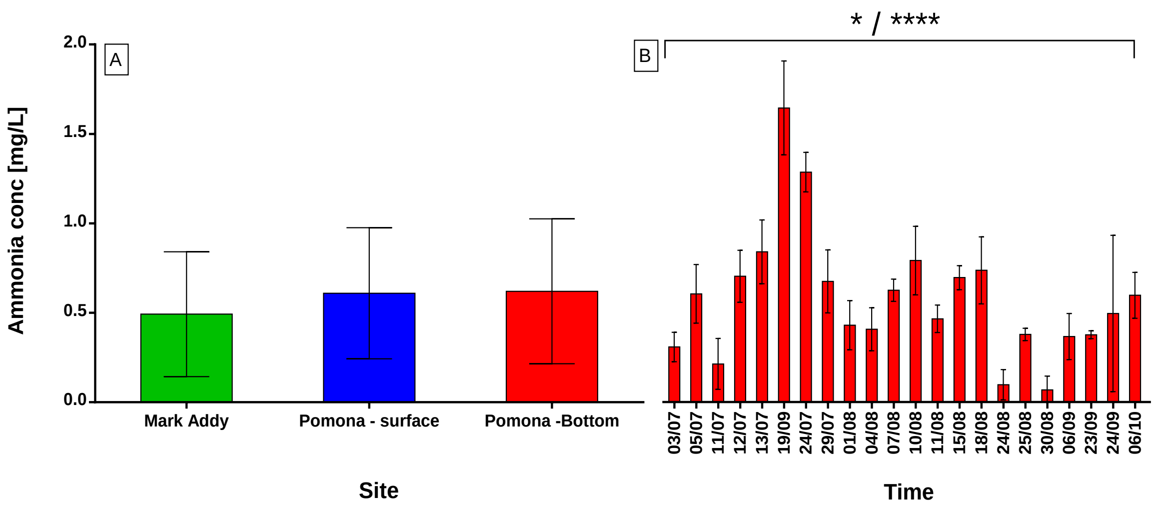

Figure 136- Average of ammonia at Mark Addy, Pomona Docks Surface and Pomona Docks bottom between July and December 2014. (P < 0.05) showed no significant differences between sites but there were a significant difference along time, n = 22.

3.3.5 Transparency and water clarity

The measurements of water clarity by Secchi disc recorded values between 50cm and 150cm. It is very clear that water clarity is very low especially during storm events, Figure 137 when the depth is nearly 0.5m due to the high level of precipitation which in turn affects the level of discharge and increases amounts suspended solids in the system.

lee

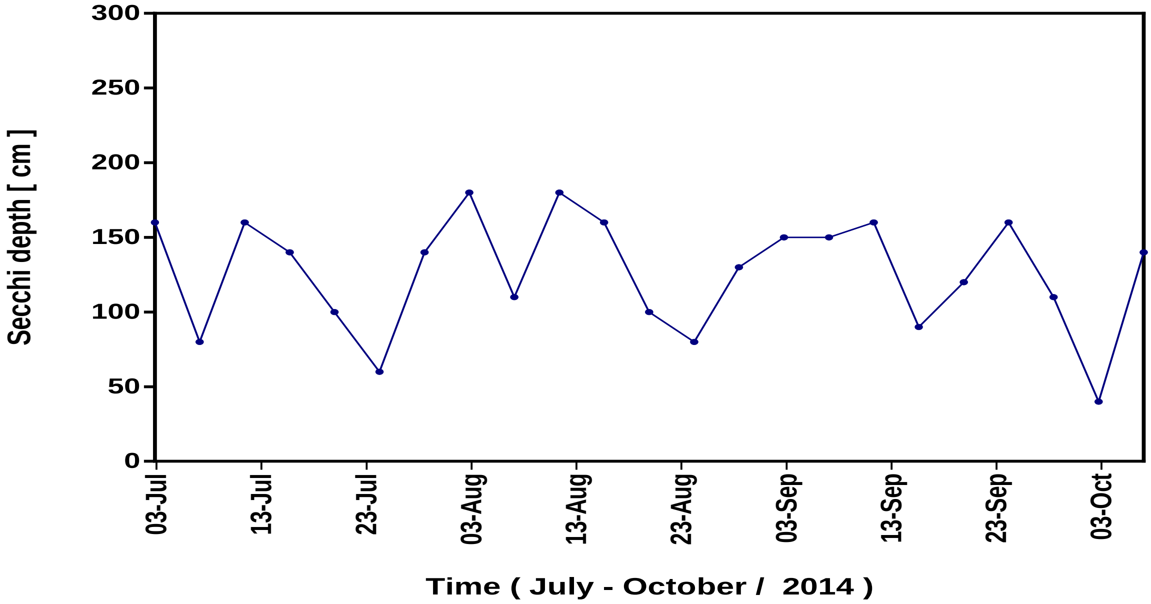

Figure 137- Short term changes in Secchi depth [cm] at Pomona Docks between July and October 2014. Data is the actual measurements taken on site by Secchi disc, n=23

3.3.5.1 Total suspended solids [TSS]

The majority of total suspended solids concentrations ranged between below the detection limit and 20 mg/l. There were three major peaks on 23rd of July 23rd of August and 3rd of October which reached 30mgl, 40 mg/l and 60 mg/l respectively. The general average was more or less than 10 mg/l, Figure 138. In common with the seasonal survey TSS during the short-term study were therefore within the normal range of up to 20 mg/L (WFD standard) with only occasional increases at Pomona Docks and the Turning Basin that slightly exceeded this threshold, reaching concentrations of around 30mg/L (Figure 86). Statistical analysis showed significant differences between Pomona Docks surface and bottom; also, there were significant differences with time, figure139.

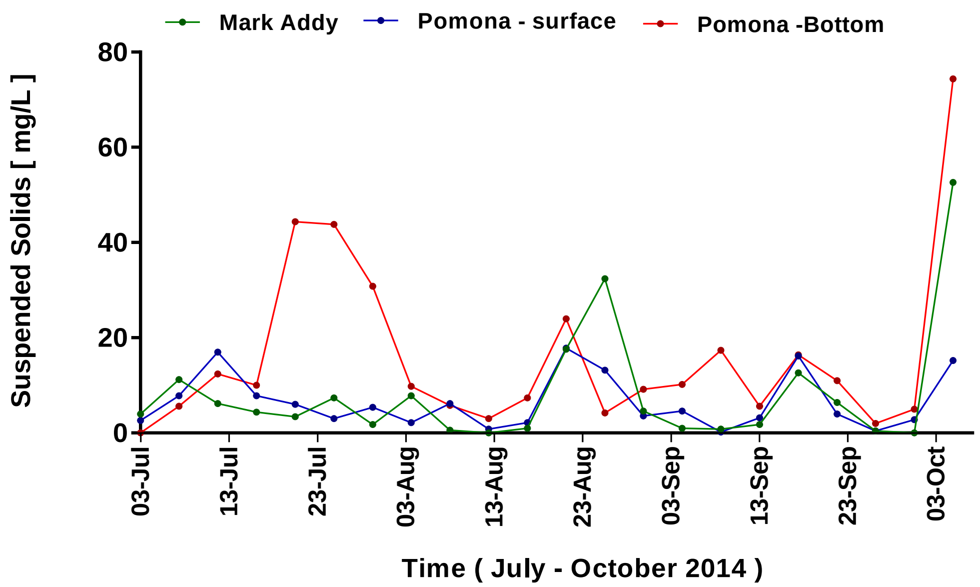

Figure 138- Short term changes in total suspended solids [TSS] at Mark Addy, Pomona Docks Surface and bottom between July and October 2014. The green line shows the values at the Mark Addy, blue lines at Pomona Docks surface and red at Pomona Docks bottom. n=22.

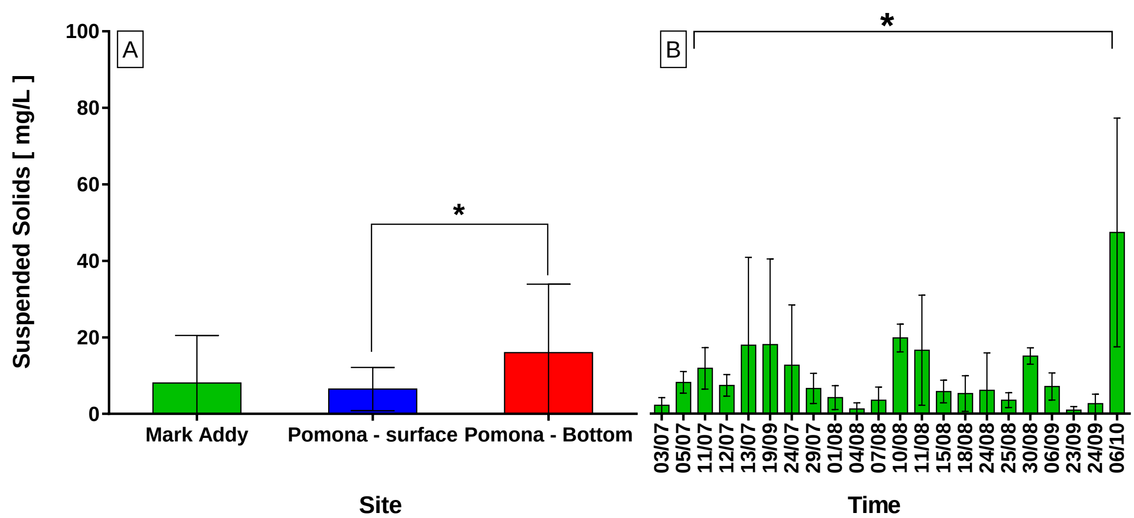

Figure 139- Average suspended solids at Mark Addy, Pomona Docks surface and bottom between July and October 2014. (P < 0.05) showed significant differences between surface and bottom of Pomona Docks, and there was a significant difference a long time, n = 22.

3.3.5.2 Total organic matter [TOM]

The vast majority of values show the level of organic matter ranged between not detectable and 20 mg/l; similarly, to the distribution of the range of total suspended solids, figure 140. There was only one significant peak on the 3rd of October which reached just over 30mgl at the Pomona Docks site, and 50 mg/l at Mark Addy. The overall average was largely around 10 mg/l for all sites, figure 141. Statistical analysis showed that there was no difference between sites, but there were significant differences over different seasons. Once again it is apparent that much of the suspended matter entering the River Irwell is of organic origin.

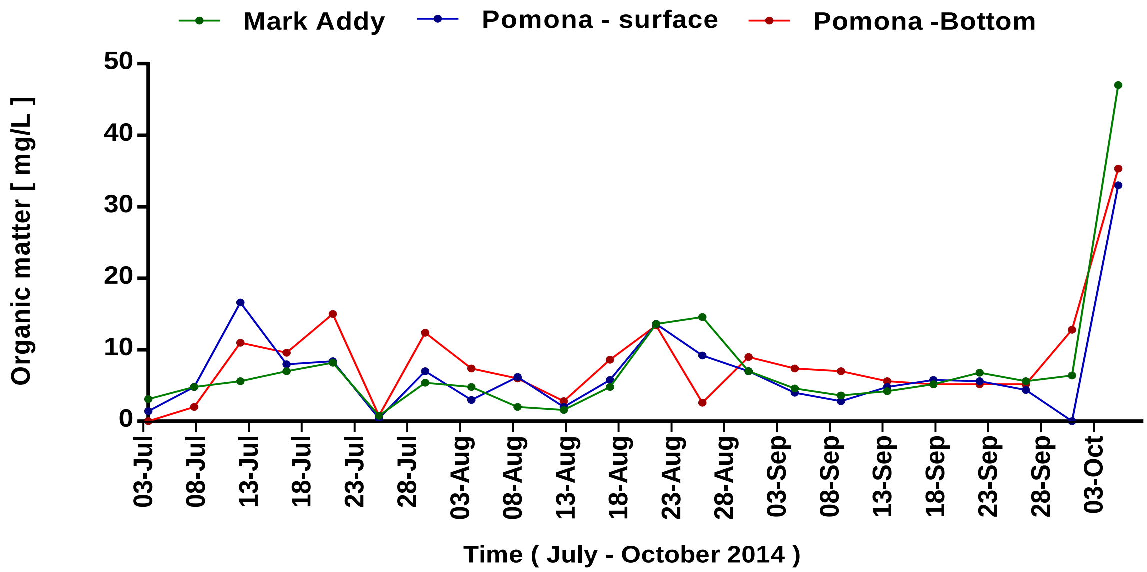

Figure 140- Short term changes in total organic matter (TOM) at Mark Addy, Pomona Docks Surface and bottom between July and October 2014. The green line represents the values of Mark Addy, blue lines for Pomona Docks surface and red for Pomona Docks bottom. n=22.

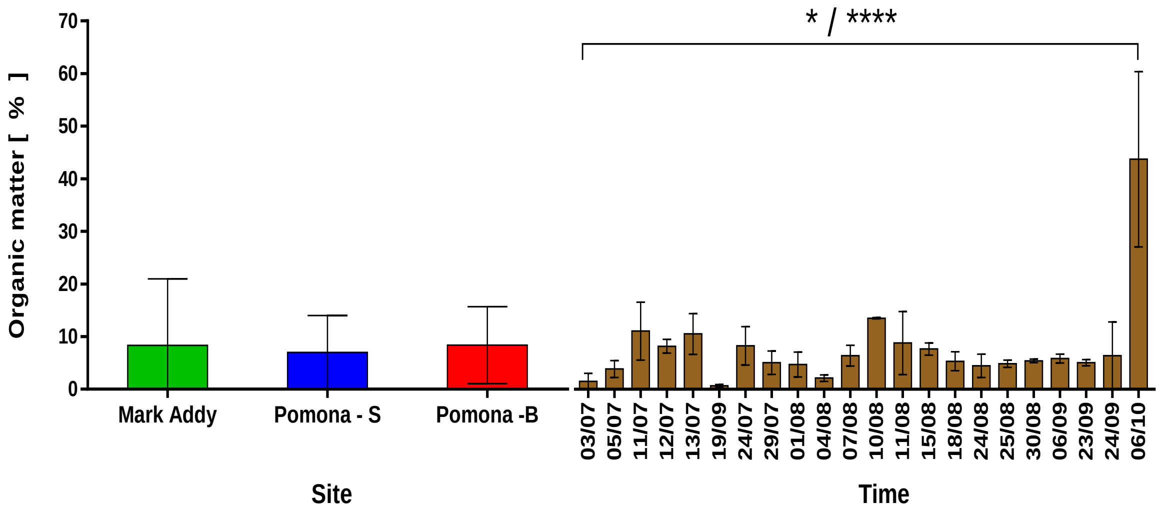

Figure 141- Average organic matter and statistical analysis at Mark Addy, Pomona Docks Surface and Pomona Docks bottom between July and October 2014. (P < 0.05) showed no significant differences between sites but there were a significant difference a long time, n = 23.

3.3.5.3 The effect of suspended solids (TSS), Organic matter (OM) and Chlorophyll-a on water clarity and transparency

Distribution of suspended solids with regard to Secchi depth showed a positive relationship which again indicates that the lack of water clarity is due to TSS (figure 142) rather than chlorophyll which showed no relationship (see figure130 above).

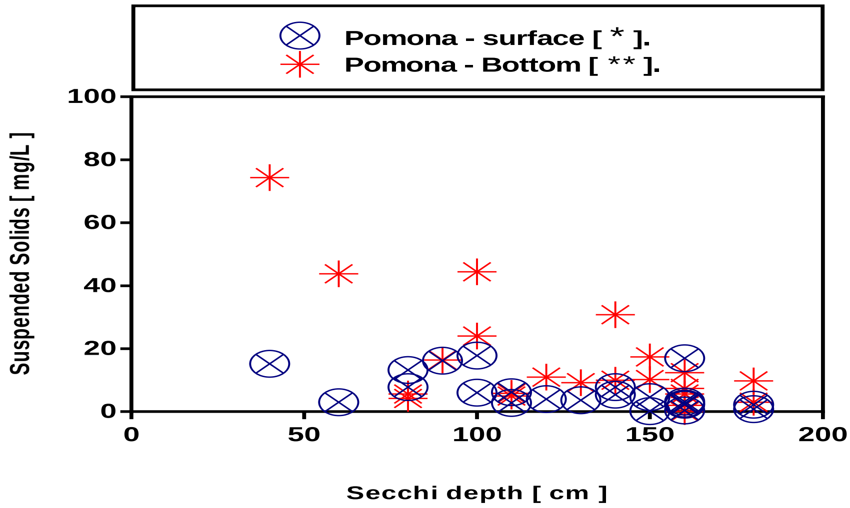

Figure 142- The effect of total suspended solids [TSS] on water clarity. Crossed blue circles symbolize the values of Pomona Docks surface, and red stars for the values of Pomona Docks bottom. (P < 0.05) showed a significant effect of total suspended solids on Secchi depth level especially at bottom level. N= 22.

However, analysis of total organic matter with regard to the Secchi depth showed that the positive effect was only evident at the bottom of the water column at Pomona Dock, figure 143.

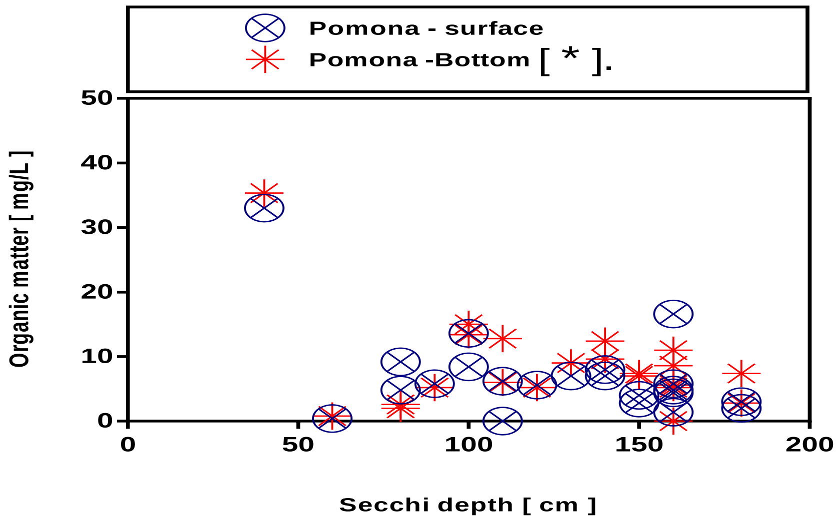

Figure 143- The effect of total organic matter [TOM] on water clarity. Crossed blue circles symbolize the values of Pomona Docks surface, and red stars for the values of Pomona Docks bottom. (P < 0.05) showed a significant effect of total organic matter on bottom level. N= 22.

There was also statistical positive effect of chlorophyll-a on water clarity and Secchi depth at both levels of Pomona Docks but the effect was much more marked at the bottom level, figure 144.

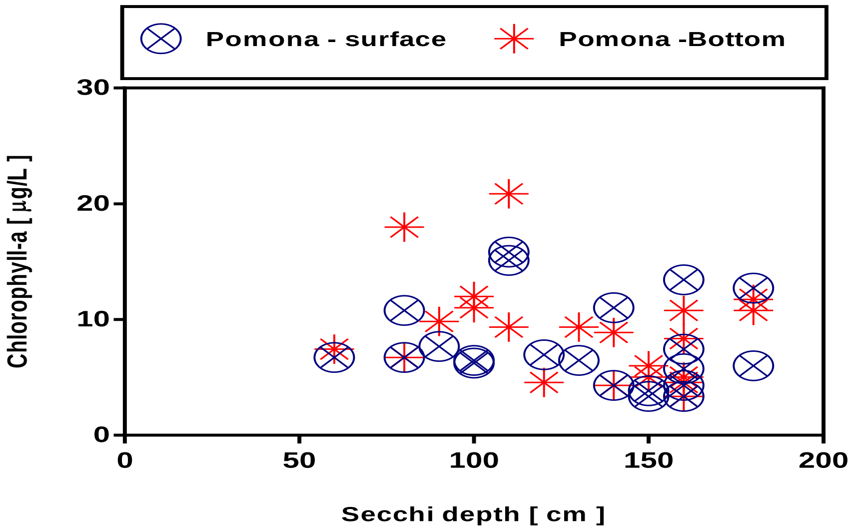

Figure 144- The effect of Chlorophyll-a on water clarity. Crossed blue circles symbolize the values of Pomona Docks surface, and red stars for the values of Pomona Docks bottom. ($P < 0.05$) showed no effect of Chlorophyll-a on water clarity. N= 21.

In summary, suspended solids level was subject to an increase of discharge rate which result in an increase of biological oxygen demand during storm events. Although the average of pH, temperature and conductivity were nearly the same in the system, it was very noticeable that all these parameters were subject to seasonal variations due to the change in discharge level. The same effect was noticed for most chemical parameters but mostly fluctuated slightly unless during the very much high level of discharge during storm event. Reduction of water clarity was directly a subject to high level of suspended solids during storm event particularly where the Secchi depth reached just over 50cm.

3.3.6 Heavy metals

There are again two groups of heavy metals, the high concentration metals Fe, Mn and Zn and the low centration metals Cu, As and Pb.

3.3.6.1 High concentration level heavy metals

Figure 145 represents the change in heavy metal levels between July and October 2014. The level of Fe was excessively high (Recommended level (UK/WFD-2015) =1ug/l). The range of Fe fluctuated between 50µg/l and 300µg/l; the trend was for an increase between 1st September and the end of November, reaching 300µg/l, and then decreasing to 200µg/l by the beginning of December. Manganese levels were mostly by far below the normal level (123ug/L. UK-WFD/2015) and just exceeded the normal point between 1st of October and middle of November. The range was between 5 µg/l to just over 200 µg/l. From the 1st of December the concentration of Mn decreased to less than 50 µg/l. The level of Zn level was relatively constant and only slightly changeable at all sites from July to the end of October; then the trend was for a marked increase in Zn, reaching around 100 µg/l between 15th of November and 1st of December. Statistical analysis showed significant differences between sites only for Fe, and no differences for all high concentration heavy metals at all sites with time.

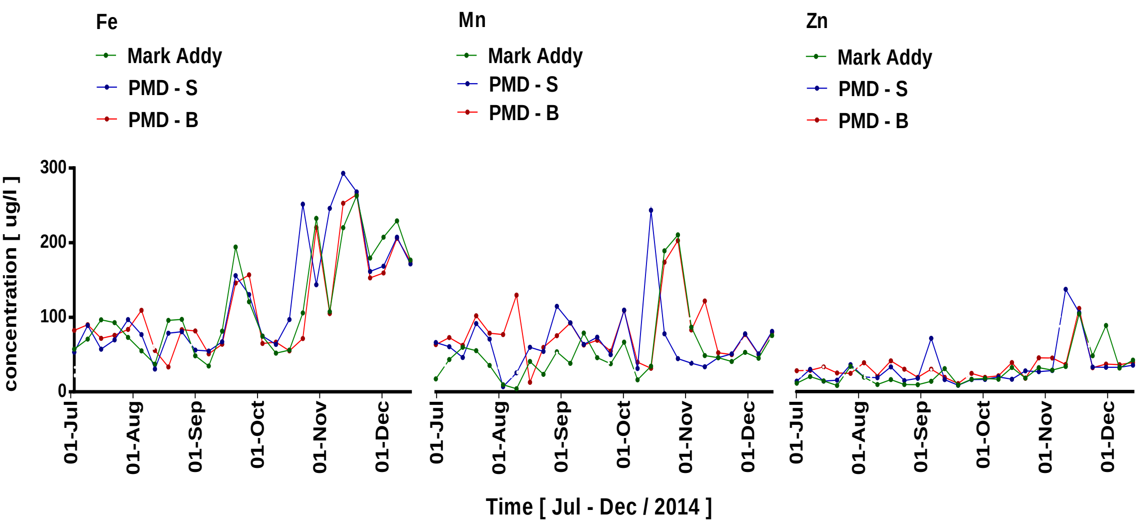

Figure 145- Short term changes of the highest concentration heavy metals at Mark Addy, Pomona Docks surface [PBD-S] and bottom [PMD-B] between July and October 2014. There were significant differences (p<0.05) between the mean of high concentration heavy metals with time, n=26.

3.3.6.2 Low concentration heavy metals

The average concentration for this group of trace metals was 50ug/l for both Cu and As, and 10ug/l for Pb. The range of Cu was between 2 µg/l and 12 µg/l where As and Pb mostly below 2 µg/l. The level of Cu fluctuated widely, and the overall trend was an increase, Figure 146. The level of both As and Pb remain steadily with slight decrease in As and slight increase of Pb between November 1st and the beginning of December, there was a peak of Pb at the Mark Addy on the 1st December where the level reached just over 4µg /l. Statistical analysis showed no significant differences between sites for Cu and As but there were significant differences with time for the same metals. In contrast there was no difference between sites or with time for Pb.

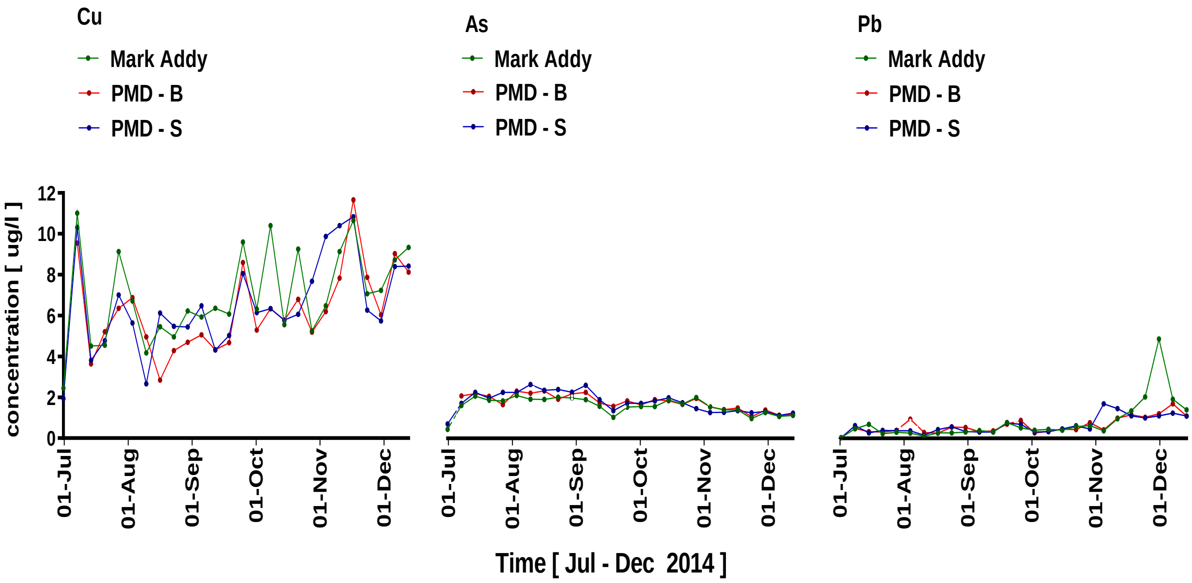

Figure 146- Short term changes of the low concentration heavy metals at Mark Addy, Pomona Docks surface [PBD-S] and bottom [PMD-B] between July and October 2014. There were significant differences (p<0.05) between the mean of high concentration heavy metals with time, n=2

3.3.6.3 Discharge effect on heavy metals concentrations

According to figure 147, it can be seen that there was slightly significant effect of discharge on the level of both low and high concentration heavy metals downstream at Pomona Docks, the effect was much greater for Fe rather than Pb. On the other hand, analysis showed no effect of discharge on heavy metals upstream at Mark Addy which may indicate resuspension of trace metals at high discharge.

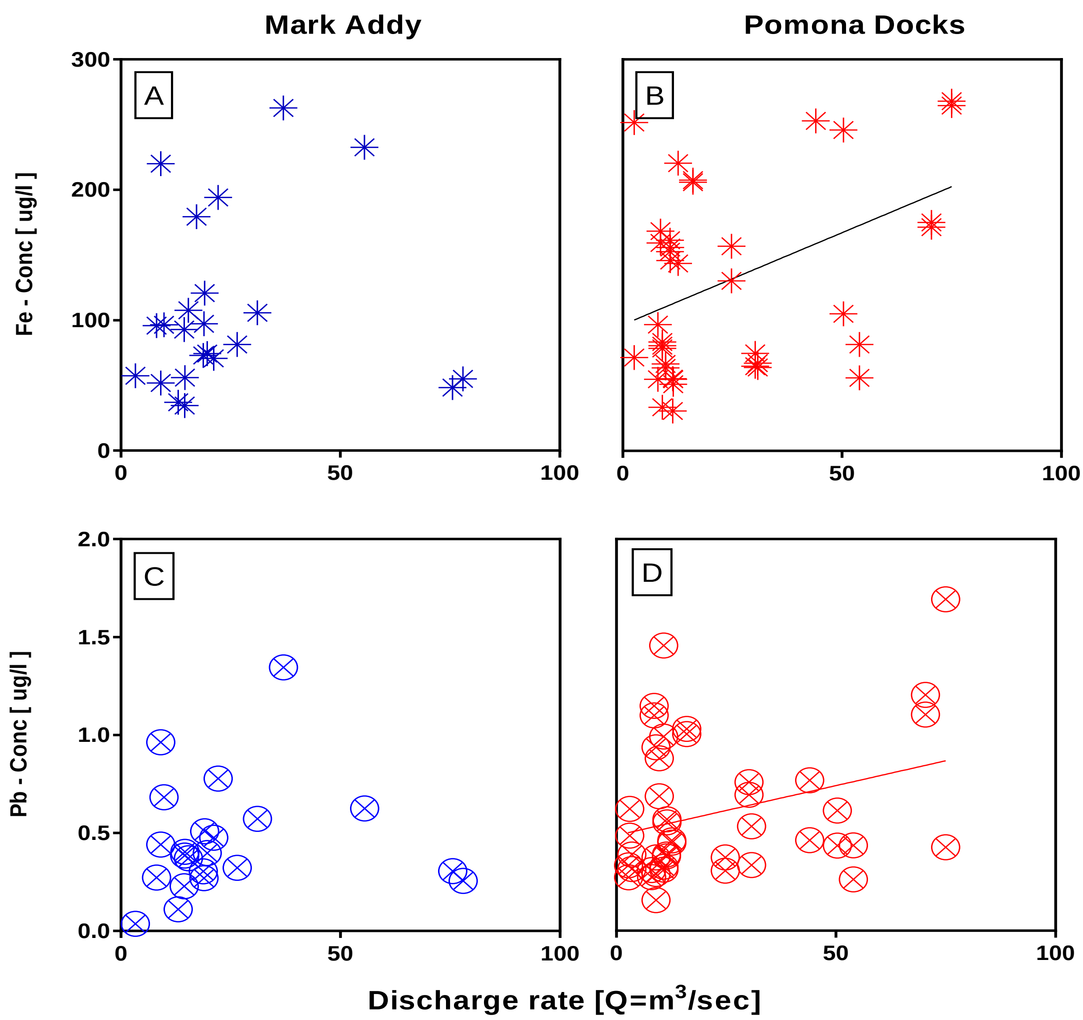

Figure 147- Relationship between discharge at heavy metals concentration at Mark Addy and Pomona Docks between July and December 2014. No positive effect of discharge [Q] ($p<0.05$) on the high concentration heavy metals Fe, Pb at Mark Addy. The relationship was positive and there was slightly significant deviation from zero at Pomana Docks as r^2 values were (Fe=0.1633 - Pb=0.0971). n=23.

3.3.6 Biological structure

3.3.6.1 Phytoplankton

High density phytoplankton populations

The most common genera of phytoplankton detected in high numbers during the short-term intensive study are *Asterionella, Nitzschia*, and *Synedra*. These taxa ranged between 100 and 3000, 100 and 800, 100 and 500 individuals per litre respectively, figure 148. The mean numbers over the whole study period was 650 individuals per litre for *Asterionella* and 350 for *Nitzschia* and *Synedra*. The overall average for high density phytoplankton varied with time and is within the range of 50 to 1000 individual per litre. The highest numbers were recorded on 23rd July, while the lowest level was recorded towards the end of the study on 23rd September, figure149.

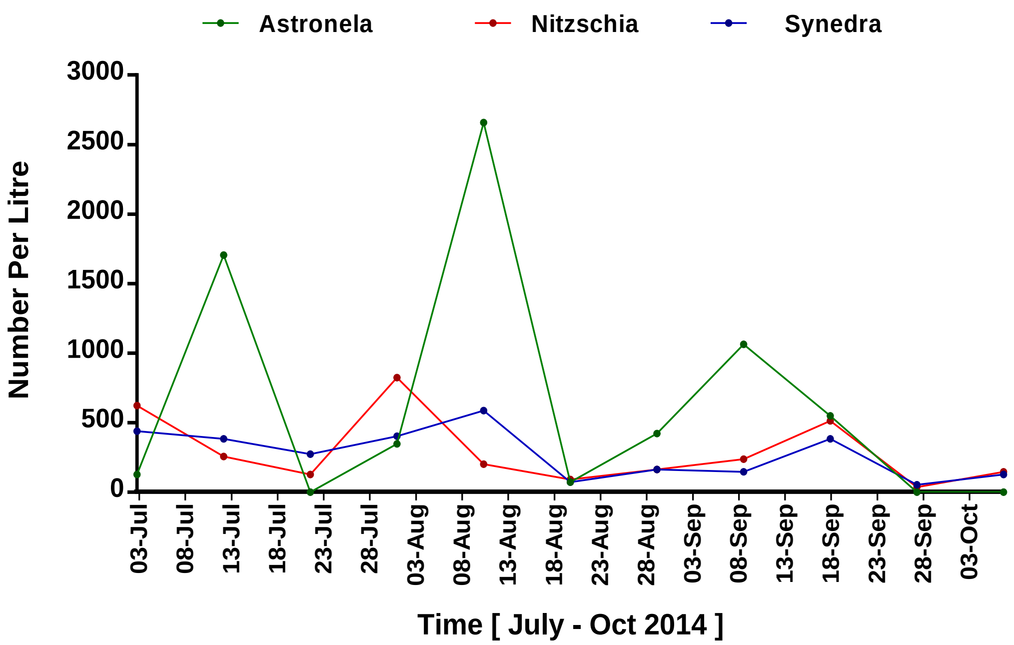

Figure 148- Short term changes in the high-density phytoplankton taxa between July and October 2014. n=11.

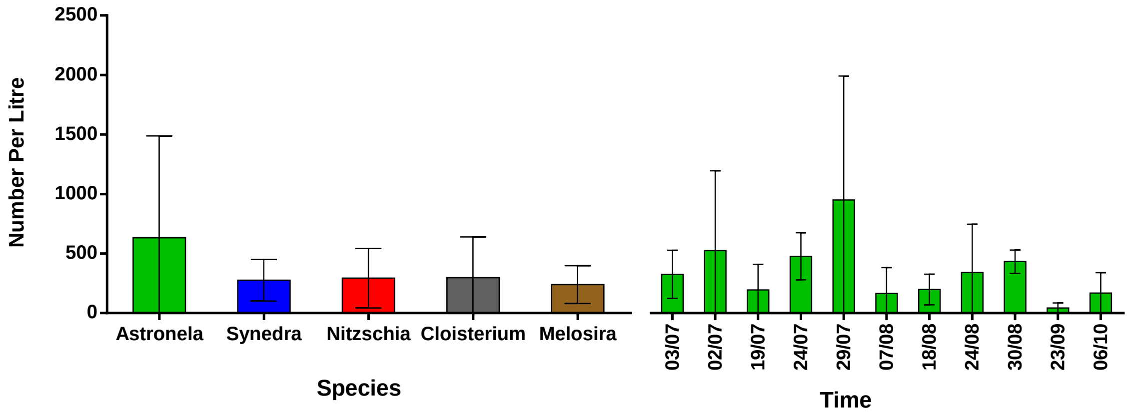

Figure 149- Average of the high-density phytoplankton populations between July and October 2014. There were no significant differences (p<0.05) between the means of species and with time. Brown-Forsythe test (p<0.05) and Bartlett's (p< 0.0001) showed a significant difference between species, n =3 for species and n=11 for time.

Low density phytoplankton populations

From Figure 150, it can be seen that the most common of the low-density population of low density phytoplankton are *Bediastrum*, *Fragilaria*, *Navicula* and *Diatoma*. They ranged between 5 and 600, 0 and 320, 50 and 300, 50 and 200 individuals per litre respectively. The average was 100 individuals per litre for *Bediastrum* and *Navicula*, and 50 individuals per litre for *Fragilaria* and *Diatoma*. The mean for low density phytoplankton varied with time within the range of 25 to 175 individual per litre. The highest densities were on 29th July, the smallest numbers were recorded on 23rd September, figure 151.

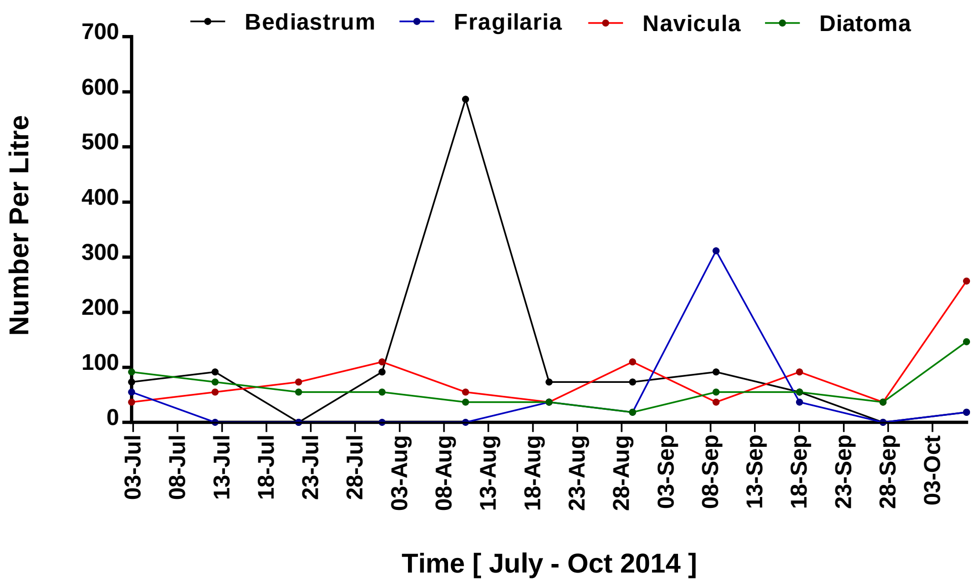

Figure 150- Short term changes in the low-density phytoplankton taxa between July and October 2014. n=11.

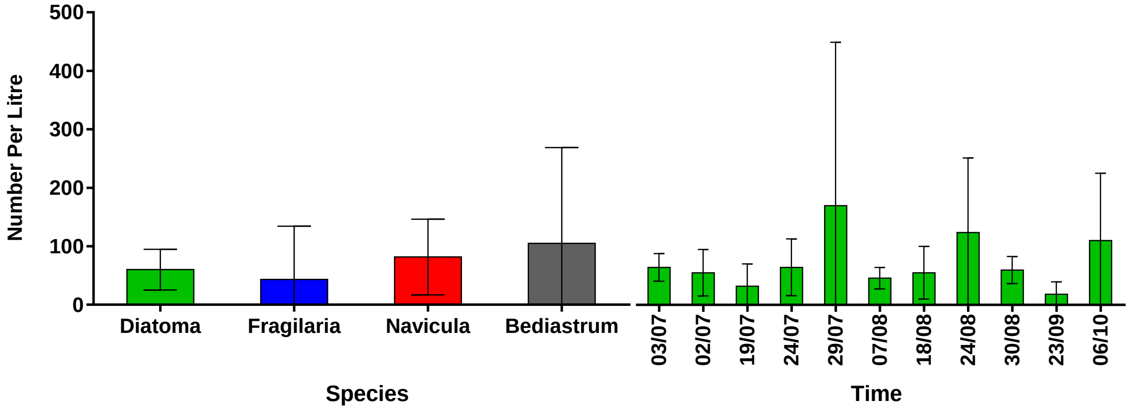

Figure 151- Average of the low-density phytoplankton populations between July and October 2014. Average values of phytoplankton species presented in coloured lines. There were no significant differences ($p<0.05$) between the means of species and with time. Brown-Forsythe test ($p<0.05$) and Bartlett's ($p< 0.0001$) showed a significant difference between species, n =3 for species and n=11 for time.

There was no marked effect of discharge on abundance/distribution of the most common phytoplankton genera at Regent Road Bridge, Pomona Docks and Turning Basin between July and September 2014. R^2 values were below 0.16 for all genera except *Tabellaria* (R^2=0.430) and P<0.05 showed there were no effect of discharge on all phytoplankton biota/ genera except *Tabellaria*. However, this genera was only present at low densities at all sites.

3.3.6.2 Zooplankton

High density zooplankton populations

According to Figure 152, the most common population of high density zooplankton were *Cyclopoids*, *Eurycercus* and *Daphnia*. They ranged between 200 and 15000, 50 and 6000, 50 and 1800 individual/m^3 respectively. The average was 4000 individuals/m^3 for Cyclopoids, 2500 individual/m^3 for *Eurycercus* and 1000 individual/m^3 for *Diatoma*. The average for high density zooplankton varied over the period of study and was within the range of 200 to more than 6000 individuals/m^3. The highest numbers were recorded on 18th August, where the lowest numbers were observed on 23rd of September, figure 153.

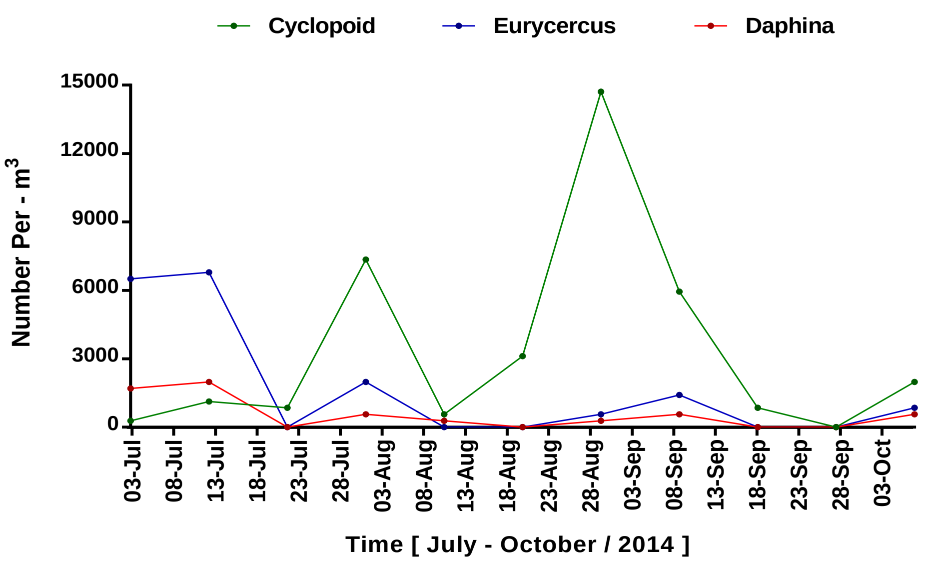

Figure 152- Short term changes in the high density zooplankton taxa between July and October 2014. n=11

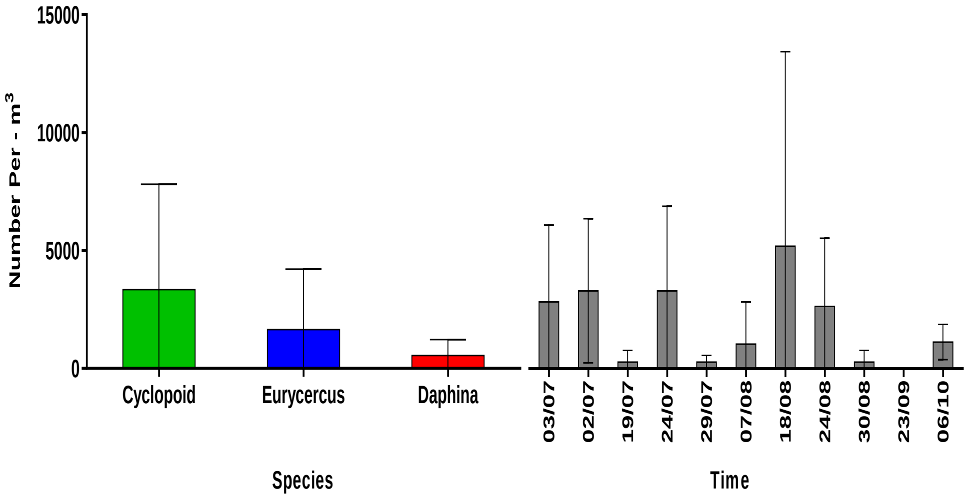

Figure 153- Short term average of the high density of zooplankton taxa between July and October 2014 There were no significant differences ($p<0.05$) between the means of species and with time. n= 3 for species and n=11 for time.

Low density zooplankton populations

The most common low-density zooplanktons are *Macrothrix, Dipterna* and *Bosmina* which ranged between 0 and 2000 individuals/m^3. The general average was 400 for *Macrothrix* and *Dipterna* individuals/m^3, where the average of Bosmina was just over 100 individuals /m^3, Figure 154. The average was 450 individual/m^3 for *Macrothrix* and *Dipterna*, 250 individuals/m^3 for Bosmina. The general average for high density zooplankton varies with time and was within the range of zero to more than 1000 individuals/m3. The highest numbers were on 20th, 29th of July and 8th of August, while no low density zooplankton taxa were present on the 18th and 28th September, figure 155.

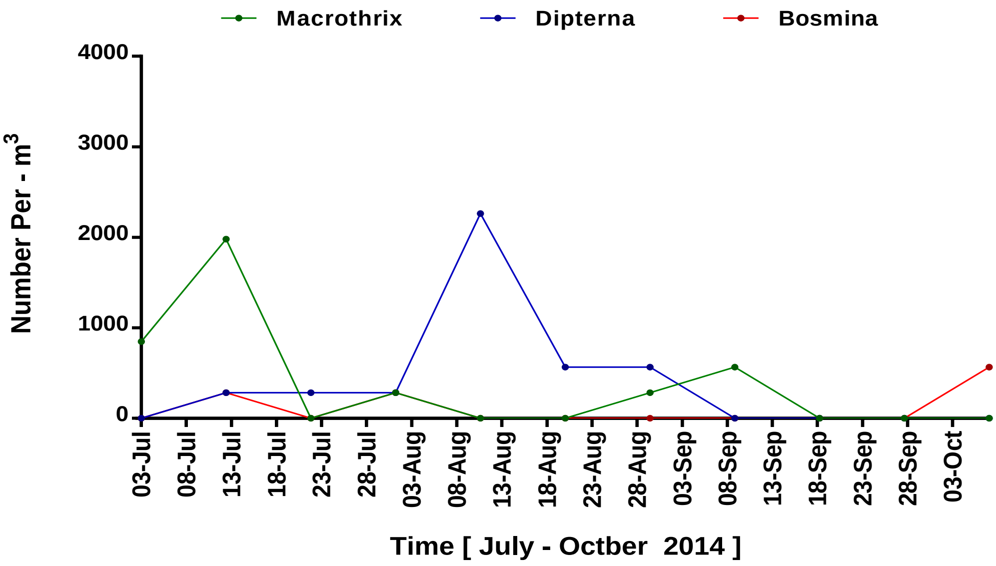

Figure 154- Short term changes in the most important and lo w density zooplankton populations between July and October 2014. n=11

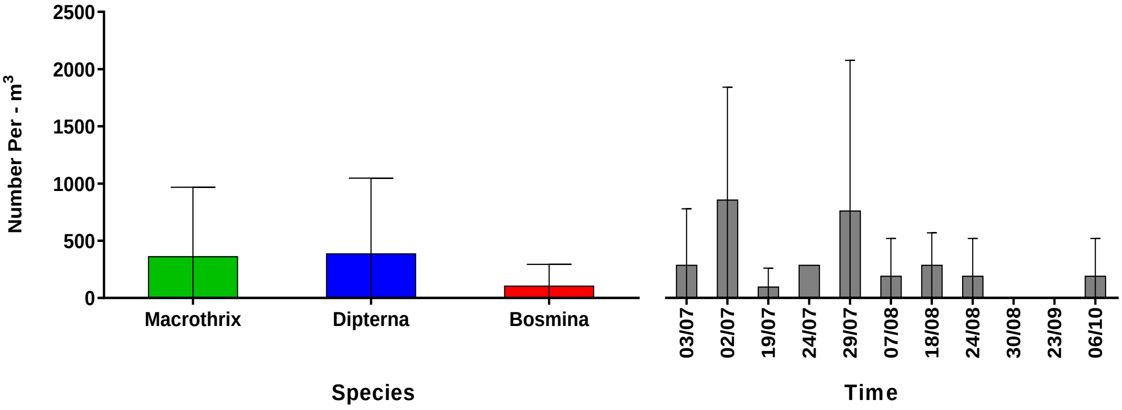

Figure 155- Short term average of low density zooplankton taxa between July and October 2014. There were no significant differences (p<0.05) between the means of species and with time. Brown-Forsythe test (p<0.05) and Bartlett's (p< 0.0001) showed significant differences between species. n= 3 for species and n=11 for time.

Phytoplankton and Zooplankton relationship

As can be seen from Figure 156, there was clear relationship between zooplankton population and the density and distribution of phytoplankton population. There were three major peaks of phytoplankton populations between 3rd of July and 3rd of

October; the first was on 10th of July as they increased from 10,000 reaching more than 12,000 individuals per litre, where the zooplankton population was 2000 and increased up to 3500 individual per cubic meter. The second peak was 23rd and 28th July and phytoplankton increased from 1,000/m^3reaching more than 10,000 individual per litre, and zooplankton population was under 1000 individual/m^3, by the time they reached up to 5,000 individual/m^3. The last peak was particularly high as the phytoplankton population increased from 5,000 on the 18th August to 15,000 individual/m^3on the 28th August by the time, zooplankton population increased steadily between the 18th August as they were just under 1,000 individual/m^3 reaching nearly 5000 individual/m^3 on the 15th September.

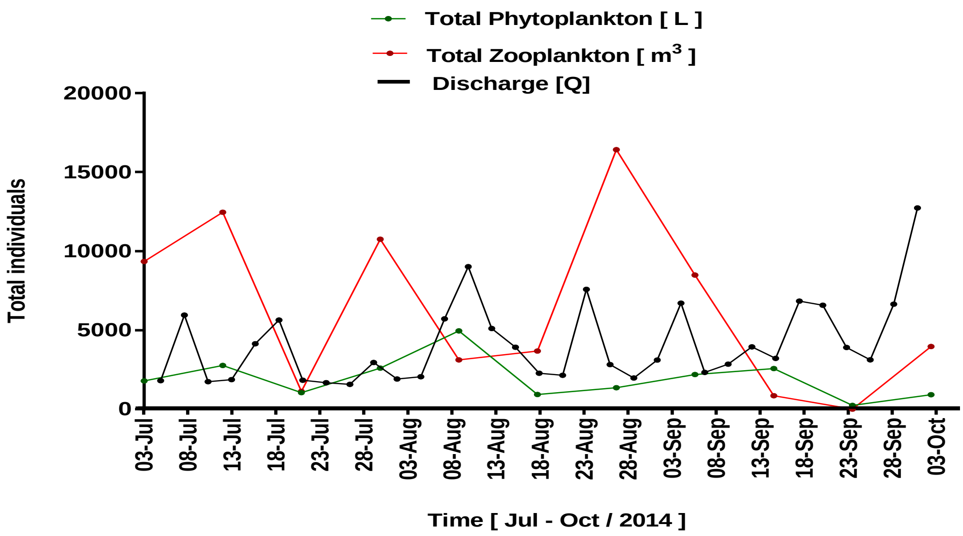

Figure 156- Total phytoplankton and total zooplankton relationship and general distribution of planktonic populations at Pomona docks between July and October 2014. n=11.

It is very clear that planktonic populations were subject to the effect of discharge during almost all storm events. There are many parameters that influence this relationship. Nutrients increased during storm events which may contribute to the rise in phytoplankton population. An increase in phytoplankton increased the number of grazing zooplankton. Increases in organic matter will also positively influence zooplankton populations as is an important allochthonous food source to this group of organisms

4 Discussion

4.1 Introduction

As outlined in Section 1.5.1 the study has two broad aims. The first is to assess past and current chemical and biological water quality of the lower River Irwell and upper reaches of the MSC including the impact of temporal changes in water quality due to high and low discharge events. The second is to examine the impact of canal modification and pollutant depositional rates on the plankton ecology of the river. In addressing these aims it will be possible to assess the efficacy of current water quality management strategies and to inform recommendations regarding potential further improvements.

The first objective therefore was to interrogate past water quality at the two sites that bracket the study reach – Adelphi Weir and the MSC Turning Basin - and use this data to establish the overall water quality and impact on the ecology. A further objective of this part of the study was to ascertain if water quality is improving or deteriorating, and if any changes had impacted on the biota. I also estimated the mass flux and hence the sources of contaminants moving in and out of the system. Lastly, I examined the influence of discharge on water quality and ecology over both the long- and short-term, including assessing the influence of change in the system from a fast-flowing erosional system to a 'linear lake' on water quality and ecology as a result of river channel re-engineering.

Section 4.2 and *Section 4.3* of the discussion will consider water quality and hydrology, including the influence of re-engineering, in particular canalisation. It will first examine how the system compares with current water quality standards and guidelines. Section 4.2 examines discharge and pollutant flux to quantify BOD (as a surrogate for organic pollution) and amounts of nutrients transport by the system. Episodic changes will allow the effect of rainfall events on pollutant flux to be assessed as an indication of the importance of episodic sources such as CSOs. Section 4.3 examines temporal changes in pollutant concentrations, including trace metals, to ascertain if reductions in point source discharges such as WwTWs upstream (Salford City Council, 2012) and water quality management in the lower reaches, in particular the Turning Basin, (Williams et al., 2010) have resulted in an improvement in water quality, including with reference to standards and guidelines. WwTWs upstream are important sources of downstream pollution. There are for example five such works that discharge directly or via tributaries in the Salford part of the Irwell catchment alone (Salford City Council, 2012). The main sources of pollution will be identified, and how these are influenced by the decrease in water flow along the system will be considered. These sections also examine the influence of rainfall events on water quality and ecology from the short-term sampling investigations. The degree to which episodic point-sources, in particular CSOs, affect water quality will be discussed and related the duration of the rainfall event.

Section 4.4 assess the impact of water quality and flow on the biota, specifically the invertebrate benthos at all sites, plus the phytoplankton and zooplankton community that develops down-stream, particularly at the final three of the five sites sampled during this study. This section includes an assessment as to whether point source discharges and water quality management in the lower reaches discussed above have resulted in an improvement in water quality and ecology.

In the following Chapter 5 that follows this discussion the conclusions (*section 5.1*) are followed by recommendations (*section 5.2*) regarding water quality improvements, including point source and diffuse pollution, possible re-engineering solutions and hydrological manipulation such as water column mixing.

4.2 Discharge and pollutant flux

In common with other urban rivers (e.g. Gurnell et al., 2007), discharge is a major influence on water quality and the biota throughout the study area. Data from the National Flow Archive at the gauging station upstream at Adelphi weir, plus those on the rivers Irk and Medlock provided data that was used to estimate discharge in the whole system between 2000 and 2012 down to the MSC Turning Basin.

Discharge and flow are of course markedly affected by precipitation. The annual average of precipitation was between 2 and 4mm over the period 2000 and 2012 while seasonal precipitation was between 1- 4 mm. The seasonal discharge of course was noticeably higher than the historical average as the values fluctuated between 5 to 10 m^3/sec during dry periods when rainfall varied from 0.75 to 3.0 mm/day, and between 10 to 20 m^3/sec during wet periods when rainfall varied from 1.2 to 4.14 mm/day.

Although the system downstream receives various quantities of water from the river Irk and Medlock, the Irwell is still that the main source of discharge to the system and accounts for between 80 and 85 % of the total, whereas the contribution of the Irk and Medlock were responsible for only 11% and 4% respectively. As a result, the average discharge ranged from 12 m^3/sec upstream at Adelphi Weir to 17m^3/sec downstream, an increase of just 30%. As expected, the episodic discharge was related to precipitation which fluctuated between 1mm/day and 20mm/day, and with an average of nearly 5mm/day.

The effect of precipitations on discharge was investigated by matching both data to create assumptions for the relationship between precipitation and discharge within the seasonal and storm hydrographs. As there was a lack of precipitation data, only one gauging station around Manchester city centre was used to collect data, therefore a further investigation is needed for better understanding of this relationship by installing more precipitation measuring facilities along the system to obtain more sufficient data for better understanding of this relationship within the system (Fekete and Charles, 2003). There was coincidence between the peak rainfall and the peak discharge with short lag time confirming that in common with other urban rivers the lower Irwell is markedly affected by urban runoff (Leopold, 1968; Mayer, 2005; Rothwell, 2010; Rothwell et al., 2010). Furthermore, the hydrographs suggest that the system is subject to an immediate increase in discharge during storm events due to engineering modification and canalisation (Richey et al.,1989) as well as the 'flashy' nature of the catchment along the system downstream despite the natural topography and vegetation along the upstream reach that might to expected to increase the lag-time due to soil infiltration (Costa, et al., 2003). PCA analysis of the seasonal data also revealed that discharge (along with Secchi depth and suspended solids discussed below) was a key parameter accounting for the overall variance in the system. It is therefore very apparent that the Irwell is subject to the urban stream syndrome (Walsh et al., 2005), one of the characteristics of which is a 'flashy' hydrograph in parallel with (and linked to) elevated concentrations of nutrients and contaminants and altered channel morphology.

Although statistical analysis showed an overall slight decrease in BOD with increasing discharge the effect of discharge on BOD upstream was relatively small with a slight decrease – for example from 7,500 mg/sec to 5,000 mg/sec with a large increases in discharge from 2.3 to 16.3 m^3/sec.. BOD flux at the lower sites was markedly greater than at the upper sites; the most probably reason for this is resuspension of sediments and inputs from the Irk and Medlock (Williams et al. 2010). Improvements in WwTWs effluent (Salford City Council, 2012) probably account for the decrease with time at the upper site on the Irwell.

There were differences between the surface and bottom BOD of the water column in the Turning Basin; the larger flux recorded in the bottom samples is likely to be due to resuspension of sediment (Williams et al., 2010). The BOD flux fluctuated from 864 to 6480 kg/day in the whole study reach between 2000 and 2010. The high resolution (monthly) data revealed marked seasonal changes in biological BOD at the lower sites where the peak downstream reached 15120 kg/day at Pomona Docks in January 2014. Karagul et al. (2005) and Prakash et al. (2009) emphasise that seasonality has significant effects on most factors at freshwater environments in both temperate and tropical rivers. The seasonal variation of BOD in the Irwell reflects the contribution of combination factors such as flow rate, aeration and both aerobic and non-aerobic decomposition of either suspended solids or sedimentation (Sharma et al., 2013). BOD was more variable in the Turning Basin but with similar levels and timing with regard to the downstream peak. BOD was higher at the bottom of the water column, perhaps due to demand from the sediment.

The flux of BOD during storm events ranged between 1,728 and 10,368 kg/day, the level fluctuated with a similar trend between the whole high-resolution study periods 1st July to 24th September 2014. The BOD during storm events exceeded average historical and seasonal figures; this increase during rainfall events therefore reflects the effect of effluents from sources that resulting from the rise in discharge and which is likely to include CSOs plus surface run-off and resuspension of organic-rich sediment upstream (APEM, 2008). Similar large fluctuations have been observed in the heavily urbanised River Tame in the West Midlands (Lawler et al., 2006).

It was very clear that discharge was major influence on suspended solids (SS) in the lower Irwell. The average flux of SS was similar at the upstream and downstream sites but increased with discharge. At the Turning Basin and Basin-6, the level fluctuated around 50,000 mg/sec between 2000 and 2012. The effect of discharge was statically significant with an increase in flux with increased discharge. Suspended solids flux was considerably higher at the bottom of the water column at high discharge at the deeper downstream sites, suggesting resuspension. Overall, SS flux was very high below Adelphi Weir and fluctuated between 10,000 and 4,320 kg/day and this high level resulted in very low water clarity. It is suggested that the major source of SS is mechanical erosion and resuspension of deposited particulates at high discharge in both the river Irwell, and that its tributaries the Irk and Medlock that contribute to the SS load. In addition, sewer overflows within the urban area of the system is also likely to influence the level of suspended solids in the lower river Irwell and MSC, as elsewhere (Carter et al. 2003; Lawler et al., 2006).

The flux of nitrate upstream at Adelphi Weir was around 173 kg/day and did not vary greatly from 2001 to 2012. However, the nitrate flux varied by far greater degree at the Turning Basin compared to the Adelphi Weir. Nitrate average flux was between 3,000 mg/sec and 1296 kg/day between 2000 and 2012 with an exceptional dramatic increase during 2001 and 2012 as the level was 1,728 and 2,160 kg/day respectively. In contrast the historical level of phosphorous flux increased at both the upstream site at Adelphi Weir and the downstream site at the Turning Basin. The upstream flux was much higher than the lower site; the former ranged between >43.2 kg/day in 2001 and >259.2 kg/day in 2012, whereas the downstream level was noticeably lower and ranged between 30.24 kg/day in 2001 and 172.8 kg/day in 2012. As expected, the seasonal nutrient flux between 2013 and 2014 was higher by far than the average historical levels as reflected sort-tern changes in discharge.

The flux of nitrogen at the upstream sites ranged from 100 to 250 kg/day, whereas the flux at the downstream sites was more changeable with several high and low values between 250 and the exceptionally high 2,500 kg/day. The seasonal change in flux of phosphorous also was more changeable at most downstream sites with sharp peaks and gradual decrease as the range varies between 25 and 300 kg/day.

In summary, the flux of both nitrate and phosphate showed different trends in the upper reaches of the system and downstream between 2000 and 2012 which results in noticeable variance in the amount of nitrate and phosphate transported by the river over the period 2000-2012. The rise in phosphorus is despite the improvements in the WwTWs and points to the lack of investment in tertiary treatment despite the fact that between 2005 and 2010 United Utilities invested over £8m on improvements to the WwTws at Eccles, Worsley, Irlam and Salford (Salford City Council, 2012). That sewage-derived phosphorus rather than diffuse (agricultural) sources of provide the most significant risk for eutrophication is not confined to the River Irwell as shown by studies of 54 other river sites across the UK (Mainstone et al., 2003). The trend in the river Irwell was for a slight decrease in both phosphorous and nitrogen during the seasonal survey between 2013 and 2014 despite the several peaks between March 2013 and August 2014 although of course this may or may not indicate improvements given the relatively short time period. There was no clear trend during the short-term event study for either for nitrate and phosphate flux with continuous fluctuation that reflects clearly the effect of discharge during storm events. Low discharge flux of phosphate is likely to be due to base flow sources, in particular WwTWs and release from groundwater upstream. Episodic increases were likely to result from CSOs and other episodic sources plus agricultural run-off (Williams et al., 2010).

4.3 Past and current water quality

As stated earlier the long-term water quality covers the period from 2000 to 2012, whereas the seasonal water quality survey took place from March 2013 to August 2014. The final part of this study examined the effect of storm events between 1st of July and 30th of August 2014. There are differences in most parameters measured in this study at every level of resolution from daily to long-term trends over a decade.

The pH of fresh water streams is very important as it can play a key role in determining stream community structure (Fiance, 1978). However historical average pH indicates that the lower River Irwell and upper reaches of the MSC are mostly near-neutral and pH generally fluctuated between 6.5 and 8.0; that is within the preferred range for most freshwater biota (Behar, 1997). The pH did not change with time and nor showed rapid changes indicative of episodic pollution by acidic or alkaline effluent. Average seasonal pH was also near-neutral as well with an average of 7.2; there were few periods where pH decreased but these did not fall below 6.5 and hence the water was never acidic. This slight fall in pH was correlated with high levels of discharge arising from increased precipitation. Rainwater is a weak acid with an average of 5.6 due to the solubilisation of atmospheric CO_2 (Mesner and Geiger, 2010). The pH during the episodic discharge study was slightly higher with an average of 7.7.

The other contributor to the pH of rivers is catchment geology and topography plus the degree of urbanisation (Smedley and Allen, 2004). The increase in pH during episodic rainfall events may reflect the 'first-flush' of pollutants from CSO and storm drains that tend to be slightly alkaline (Defra, 2014) followed by subsequent dilution by clean water from upstream run-off that will have a lower pH. Primary productivity can also increase the pH in freshwaters by shifting the carbonate-bicarbonate equilibrium (e.g. Sigee, 2005) and his phenomenon may have accounted for the slight increase in pH in the Turning Basin sometimes observed during the summer.

There were slight differences in temperature between the shallow well-mixed water at the upstream site and the deeper water downstream at the Turning Basin and Basin 6 between 2000 and 2012. The average temperature of the system was $12C^0$ with a range mostly between 5^oC and 20^oC over the last decade while seasonal temperature in 2013-2014 ranged between 2^oC and 20^oC; both reflect natural seasonal changes in air temperature. The relatively low temperature at all sites in the colder months of winter remain within the thermal limits that UK freshwater biota can tolerate (Elliott, 1994) and are within the normal range suggesting that there is no pollution from heated effluent.

There was no evidence of thermal stratification in the deeper sections of the Irwell/MSC during the summer. Thermal gradients were however recorded in the MSC from Pomona Docks prior to installation of oxygenation and water column mixing devices in the early 1990s resulting in serious water quality problems such as bottom water anoxia (Williams et al., 2010) and this study had confirmed that these remediation strategies are effective at presenting stratification, including during the warmer summer period.

The level of long term conductivity within the system ranged between 300 µS/cm and 600 µS/cm over the study period of 2000 and 2012. The UK Link organisation (2006) stated that although any freshwater system with conductivity of between 50 and 200 µS/cm is considered soft uncontaminated water while values up to 600 µS/cm may also indicate the absence of pollution depending on the nature of the catchment, in particular the geology as limestone for example will increase the conductivity. However, the catchment of the Irwell is largely low weathering bedrock (Rothwell et al., 2010); therefore, and in common with other urban rivers (Paul and Meyer, 2001), the elevated conductivity in the Irwell is likely to be a result of polluted run-off.

Seasonal conductivity ranged between 250 and 1,250 µS/cm at all sites again confirming that in common with other rivers (e.g. Lawler et al., 2006) the system is markedly affected by discharge. The fluctuation along the system reflects the effect of seasonality that controls water temperature (high and increases during summer) and precipitation (low and decreases in summer) which in turns affect the level of discharge and water viscosity; factors that are more likely to cause an increase of conductivity (Talling, 2009). During storms conductivity fluctuated between 200 µcS/cm and 600 µcS/cm with an average of just over 400 µcS/cm. Conductivity responded rapidly to increases in discharge arising from rainfall events; for example, an increase in discharge from 5 to 20 m^3/sec over a period between 48 to 72 hours resulted in a doubling of conductivity from 250 to 500 µS/cm. Conductivity reflects the nature of erosional processes and mechanisms of the river through the influence of the river's catchment, topography and geology. In the case of the highly urbanised River Irwell, catchment run-off and hence conductivity is profoundly influenced by anthropogenic factors such as impermeable surfaces such as roads and, importantly, point sources such waste-water discharges from WwTWs and combined sewer systems (CSS) (Mallin, 2000; Tallin, 2009). Salt used during winter for road de-icing is thought to be the main contributor to conductivity increases during the winter months and which in turn can further damage the biota (Kefford, 1998).

Dissolved oxygen (DO) is a vital parameter for aquatic life and that is therefore a key measure of fresh water quality; this parameter is greatly affected by a combination of physical, chemical and biological characteristics, in particular algal biomass (respiration, cell death), allochthonous and autochthonous dissolved and suspended particulate organic matter, ammonia and sediment oxygen demand (Sanchez et al., 2007). However, parameters that influence DO will vary with the water body (Haider, 2013), including the natural of the catchment such as the degree of urbanisation (Paul and Meyer, 2001). The average DO in the Irwell over the period 2013-2014 was 12mg/L (close to 100% saturation) upstream at the Mark Addy site; therefore, the upper site reveals healthy levels of DO required for normal physiochemical and biological processes as the level is within the normal range of fresh water systems of between 7.56 mg/L and 14.62 mg/L (Minnesota Pollution Control Agency, 2009). DO in the lower Irwell/MSC was >10mg/L during the winter but however fell to around 4mg/L during June 2014.

The relationship between dissolved oxygen and temperature was, as expected, very clear through the system. DO was sometimes well below fully saturated levels which is highly likely to be due to pollution. The degree of DO saturation in the upstream region of the system was mostly over 80% from 2000 to 2012 which according to the Environment Agency's Chemical GQA classifies the site as 'Very Good'). In contrast, DO downstream fluctuate between 60% and 80% which according to the GQA classifies the system as between 'Fair' and 'Very Good'. Although there is no evidence of hypoxia (acute oxygen deficiency) and the concentration of DO at all sites was almost always above 4mg/L, which is considered to be suitable for many biota, including fish, and hence a stable aquatic biological community (e.g. Williams et al., 2010). However, during hot periods over the summer DO in the system, especially the Turning Basin, falls to 4 mg/L which is not far from the status of chronic and severe oxygen deficit of a DO of <2mg/L (O'Boyle et al., 2009) but is however sufficient for survival of pollution-tolerant course fish (Williams et al., 2010). DO is however below that required by many benthic invertebrates (Davis, 1975).

Short-term changes in DO revealed direct and rapid response to rainfall events with both increases due to reaeration and effluent dilution and decreases due to pollution. Upstream DO showed short-term (daily) fluctuations between 14 and 8mg/L whereas the downstream sites, which it is suggested were affected by effluents from the Irk and Medlock showed larger fluctuations. The Turning Basin showed far less variation both historically from 2001 and in 2013-2014 but is an atypical site due to injection of oxygen from 2001 to 2012 followed by water column mixing up until the present (Williams et al., 210; APEM Ltd, pers. comm.).

BOD is a key indicator of the quality of waste effluent and hence an important tool to investigate water quality in urban water bodies, including the efficacy of WwTWs to control pollutant discharges (Penn et al., 2009). High and variable BOD is also a characteristic of urban rivers (Paul and Meyer, 2001) and a symptom of the urban stream syndrome (Walsh et al., 2005). BOD can have a marked effect on DO if the oxygen demand arising from the metabolism of organic matter is not satisfied by re-aeration – whether natural or artificial. It is clear from the historical survey of past data from the River Irwell between 2000 and 2012 that the BOD of the system was within the normal range typical of slightly or non-polluted systems resulting in a GQA of 'good' as values were mostly under 4mg/L. During the late1960s and early 1970s BOD fluctuated between 3.5 mg/L and 2.5 mg/L above Adelphi Weir reflecting a reduction in discharges and improvements in effluent treatment especially of discharges into the MSC (Wood and Lee, 1974) as values of up to 20 mg/L were found in the Irwell and tributaries in the 1950s (Harding et al.,1981; Holland, D.G., 1984). The BOD was similar at both downstream site (Turning Basin and Basin 6) as the figures were 3.3 mg/L and 3.8mg/L respectively. As a result of the decline in BOD there was an improvement in DO, including a reduction in DO sags in rivers of the Mersey catchment between 1975 and 1995, including the Irwell (Langston et al., 2006). BOD in the Turning basin varied between 2 and 7 mg/L between 2000 and 2010; thus the general average was 2.5 upstream and 3.5 downstream which reflects the grade 'Good' according to the GQA. The more recent (2013-2014) seasonal study revealed BOD of 1-4 mg/L at Mark Addy and 0.5-7.0mg/L in the Turning Basin, and the level of BOD during short term investigation fluctuated between 2 and 6 mg/L within an average of 3mg/L except at the Pomona Docks where the average was just over 2mg/L. Despite the reduction in BOD, DO remain low below Pomona Docks throughout the 1980s. The reason is the very high sediment oxygen demand (SOD) resulting from the deposition of particulate contaminants facilitated by canalisation changing the hydraulic regime resulting in a marked reduction in flow and hence a higher retention time (Williams et al., 2010). An SOD of 1,000 mg m^2/hr has been recorded (HR Wallingford, 1999) which is representative of heavily organically polluted sediments. These sediments exert a SOD on the overlying water column that combined with the high retention time and the deep (up to 9m) water column, resulted in stratification and bottom water anoxia during the warmer months prior to the installation of oxygen injection followed by Helixor mixers from 2001. Similar effects of SOD on the overlying water column oxygen demand have been observed in other urban waterbodies (Ellis and Hvitved-Jacobsen, 1996). The high DO record at these sites in this study is a result of the water column mixing mentioned above (see also Williams et al., 2010) which satisfies both the BOD and SOD. The importance of the deposition of suspended particulates for DO is discussed further below. In addition to organic effluent, phytoplankton primary production is a potential addition factor contributing to the BOD in the lower reaches (White, 1993) following cell death.

The water clarity in the lower Irwell and the MSC sites is very poor and hence the Secchi depth is quite low, fluctuating mostly between 0.5m and 1.5 m according to the historical dataset. The average Secchi depth for the more recent seasonal and episodic data is similar in that it varied from a low of 0.5m to 2.0m. The most significant reason for the reduced water clarity in the system is very likely to be the suspended solids (SS) load which commonly reached concentrations of 80mg/L (maximum 436mg/L) during the historical survey. The level of suspended solids also exceeded 80 mg/L during the storm events survey of July/September 2014. Moreover, although average SS were often within the acceptable value of <25mg/L under the Surface Waters Regulation, DFO-2000 and EIFAC, many researchers suggest that these standards should be reconsidered (WFD-UK, 2007) and this study shows that the <25mg/L standard does not adequately describe the relationship between water clarity and suspended solid concentration. Chlorophyll concentrations as an indication of algal biomass plus phytoplankton counts do not indicate the phytoplankton was a major reason for reduced light penetration. In common with other urbanised rivers (Chebbo et al., 1992; Rossi et al.,2005; Lawler et al., 2006) the often high level of suspended solids is mainly due to the River Irwell receiving run-off and discharges from the heavily urbanised catchment within Greater Manchester, including upstream of the first site at Adelphi Weir. The main tributaries rivers, the Irk and Medlock, are also affected by urbanisation, industrial effluents and sewage treatment works and hence contribute to the high level of suspended solids within the Irwell. SS levels of over 50mg/L have been recorded in the Medlock and sometimes reached 125mg/L (Williams et al., 2010; Medupin pers. comm). Isolation of most of the dock basins from the MSC resulted in an improvement in water quality and water clarity (Struthers, 1984; Williams et al, 2010; Mansfield et al., 2013). The steady increase of water clarity in, for example, the small enclosed basin where the average Secchi depth was 0.8m in 1990 and 5.0m in 2007 (Williams et al., 2010), and 2017; Qiao, pers. comm.) indicate that effluent is the main source of the constantly elevated levels of suspended solids in the Irwell. The isolated basins have improved significantly since isolation from the MSC and the water clarity is now within the range of natural lakes such as Rosthern Mere (Dean, 2004). Water clarity in this study is greater as measured by Secchi depth at the upper sites, ranging from 0.5m-2.0m.

The adverse effects of SS on photic depth and hence primary production (discussed below) are compounded by the hydraulic regime of the Irwell below the Mark Addy site. Canalization and deepening of the river slows the flow and in combination with the consequent high residence result in SS settling readily onto the sediment. It has been estimated that 0.5m of fresh sediment is deposited in the Turning Basin each year (APEM 1996). Sediment depth varies from 1.2m to 4.2m between the Woden Street site and Mode Wheel Locks downstream of the Turning Basin site and the total quantity of accumulated sediment is estimated at 460,000m^3 (APEM 1996). These sediments exert a SOD on the overlying water column as discussed above. The level of chlorophyll-a in the slow flowing reaches is commonly low which can be due to low concentrations of nutrients that restricts primary production; however this is not the case in the lower Irwell where concentrations of phosphorus for example are indicative of eutrophication (discussed below). The reason for the low level of chlorophyll-a is highly likely to be due to the effect of SS which reduces water clarity and which in turn affects the phytoplankton population that is considered the major source of chlorophyll a in slow-flowing lower River Irwell sites, as elsewhere (Larned and Schallenberg, 2006). The calculation of Carlson's trophic state index (Carlson, 1977) helped to resolve this question and is further discussed below. The high level of sedimentation of SS will also smother benthic algae and macrophytes (Dodds, Smith, and Zander, 1997) although the impact of sediment deposition on these primary producers was not examined in this study.

One of the most important chemical parameters considered in this study are the nutrients phosphorous and nitrate, plus ammonia which is both a plant nutrient but also potentially toxic. The River Irwell, in common with the whole Mersey catchment, is mostly within a heavily urbanised area that is affected by a wide range of anthropogenic activities such as industry, plus agriculture in the middle and upper reaches. However, the level of nitrate at both the upstream and downstream site between 2000 and 2012 and at these and the additional sites in 2013 and 2014 were low and within the accepted GQA and WFD standards of >5 to 10mg/L and indicate 'Low' nitrate and hence good water quality (APEM, 2005). Seasonal concentrations of both dissolved and total nitrogen in the system fluctuated around an average of 5mg/L while during the episodic discharge study nitrate did not exceed 5 mg/L. One source of high nitrate is underperforming WwTWs (Rees and White, 1993) which suggests that the works on the Irwell and its tributaries are performing adequately (see also Manchester City Council, 2008). Other sources of nitrate to groundwater and ultimately to urban rivers that again are presumably not significant contributors to nitrate levels in the Irwell is leaky sewers and landfill (Wakida and Lerner 2005).

The level of ammonia ranged between 4.0mg/L and 0.5mg/L, the general average was 1.5mg/L upstream at Adelphi Weir and 1.0mg/L downstream at the Turning Basin between 2001 and 2013, which is well within the GQA and WFD standards of 0.6mg/L and 2.5 mg/L for 'good to fair' water quality. Seasonal ammonia concentrations in 2013 and 2014 fluctuated between 0.25mg/L and 3.3mg/L with an average <1.0mg/L, and the range of ammonia was third lower during the short-term survey at between 0.25mg/L and 2.0 mg/L despite storm surges that can increase ammonia concentrations in urban rivers (McCutchan et al.,, 2012). These figures reflect that the system is classified as 'Good' to Fair' for ammonia according to chemical GQA water quality standard (APEM, 2005). However, such averages disguise the fact that there were many peaks in ammonia which resulted in the lower grade of 'Poor', particularly during the period 2003-2009. However more recently, including during the seasonal survey of 2013-2014 ammonia levels did not exceed 2.5 mg/l ('Fair'). Oxidation of high levels of ammonia to nitrate will contribute to oxygen depletion in slow flowing rivers (e.g. Mason, 2002) but is therefore unlikely to be an issue in the lower Irwell. The generally low concentrations of ammonia throughout the study reach in recent years indicate a clear improvement in the functioning of the WwTWs which are often the major source of ammonia (Haure, 2006) and which in the 1960s adversely impacted on water quality of the Irwell (Holland and Harding 1984). There are combinations of factors that contribute to the episodic high levels of ammonia. One factor is the temporary underperformance of WwTWs and an additional adverse impact arises from the overloading during storm events resulting in the release of partially treated effluent: one or both of these factors may have contributed to the episodic deterioration in water quality seen in the previous decade. Further investigation is also required examine the contribution of other parameters such as temperature and pH (which influences the amount of the more toxic unionised ammonia: e.g. Sigee, 2001) especially downstream where the system is considered as a linear lake as the interaction of these factors could raise the level of ammonia toxicity which will in turn will affect biodiversity (Wurts and Durborow, 1992).

Notwithstanding the major efforts to control the negative impact of the large inputs to the system from WwTWs, CSOs and upstream agriculture, concentrations of phosphate fell in the lower Irwell by only one degree on the scale from ‘excessively high’ to the category ‘high’ according to the GQA phosphate grade assessment between 2000 and 2010. The general trend was for a decrease in phosphate from 2001 to 2012, decreasing from an average of around 0.8 mg/L to just under 0.2mg/L. This compares with 0.4mg/L at the Turning Basin site which was therefore more than double that upstream. Levels of phosphate therefore remain ‘High’ to ‘Very High’ according to the Phosphate GQA and the WFD-2005 both on the basis of my work and from other studies (Salford City Council, 2012). Total phosphorous was much higher upstream, with an average of 1.0mg/L at Adelphi Weir compared the lower sites where the average was just under 0.5mg/L, presumably reflecting the breakdown to phosphate and entrapment in the sediments (Tournoud, et al., 2005).

A source of phosphorous is likely to be the WwTWs above the study reach as not all on the River Irwell include tertiary treatment to remove phosphorus. However, improvements to the WwTws in the upper reaches of the Irwell and its tributaries have resulted in its compliance with the requirements of the WFD of phosphorous. The rivers Irk, and Medlock are also significant sources of phosphorus; these tributaries are a reason for the increase in phosphorus between the upstream site and the Turing Basin. For example, the lower reaches of the Medlock contains an average of 0.501 mg/L of phosphate and contributes around 15.28kg/day to the Irwell (Medupin pers. comm.).

A range of phosphorous over 0.035mg/L is indicative of a possible algal bloom and hence eutrophication in the system (Williams, 2004). The average of long term Trophic State Index (TBI) was between 60 and 70 and with a mean of 67.9 indicating that the system is eutrophic with potential for cyanobacterial dominance and that surface algal scums are possible (Carlson, 1977, 1996). In 2001 and 2005 scores above 70 indicates hypereutrophy and hence the potential for heavy phytoplankton blooms throughout the summer. However no algal blooms were recorded in the lower reaches of the Irwell or in the Turning Basin and, as discussed above, chlorophyll concentrations were low. Similarly, phosphorus figures of both seasonal and short term trophic status, indicates a high level of eutrophication with score between 69 and 79 while that based on chlorophyll ranges from 41 to 51 which Carlson (1977) states indicates moderately clear water. The reason for the disparity is shown by the Secchi TSI which, with the exception of storm events, ranges from 57.1 to 62.0, again indicating eutrophic conditions. The reason for the low Secchi depth is the high SS load which reduces photic depth and high primary production and not a high level of primary production. Storm events do not markedly change the indices although the Secchi index decreases slightly; presumably due to run-off containing less SS. A similar pattern emerges to the seasonal survey as the trophic status indicated by chlorophyll concentrations is lower at an average of 51.9 with only one year exceeding 60 whereas the phosphorus TSI is commonly >80.

The TSIs support the above suggestion that SS are a key control on phytoplankton productivity in the lower Irwell and the upper MSC. Such control has important implications for the future management of the system as a reduction in SS resulting from improvements in run-off quality and CSOs that is not accompanied by the reduction in levels of phosphate are highly likely to result in aesthetically displeasing algal blooms, including of the potentially harmful cyanobacteria which, as discussed below, are present in relatively small numbers in the MSC. Such blooms are likely to be more prevalent in the lower Irwell/MSC than many UK urban rivers and canals due to the water depth and slow-flowing nature of the system. However, such blooms are observed in rivers of similar hydrology and water quality elsewhere, for example in China (e.g. Li et al., 2009)

Given that WwTWs and upstream agricultural run-off are key sources of phosphorus to the system, control of CSOs and other urban point and diffuse sources are unlikely to reduce phosphorus concentrations below the threshold to prevent algal blooms. Thus, tertiary treatment of all the major SwTWs effluent on the Irwell and its tributaries is required. Retro-fitting of a tertiary treatment plant has been undertaken at the Daveyhulme SwTWs downstream of the Turning Basin to reduce phosphorus levels in the MSC and hence amounts entering the Mersey Estuary (Chambers, 1991). However, sediments downstream of WTWs act as a sink for phosphorus and therefore introducing tertiary phosphorus removal is likely to make these sediments vulnerable to phosphorus release (Environment Agency, 2015). A further long-term source of phosphorus to the Irwell and MSC is therefore the sediments and phosphorus release will be enhanced as a result of disturbance due to scouring and also dredging to reduce flood risk. The change in the hydraulic regime resulting from the construction of the MSC will necessitate continual dredging on the Turning Basin into the foreseeable future (Williams et al., 2010) with the concomitant risk of long term release of phosphorus.

There was a lack of data with regard to heavy metal concentrations in the long-term surveys carried out by the Environment Agency at Adelphi Weir and APEM Ltd at the Turning Basin between 2000 and 2012. However other studies revealed historically high level of contamination by metals in the lower part of the River Irwell, in part due to the suspension of the sediment disturbed mainly by dredging but also by local construction. Studies carried out in 1987 by Hendry et al (1988) revealed very high concentrations of Cu, Zn, Pb, Cr, Ni Cd, As and Hg in the sediments of the Turning Basin, all of which exceeded the Dutch guidelines for sediment contamination. Chromium, Pb and Ni in the MSC were found at higher concentrations than in sediments from a number of dock basins in the UK, whilst concentrations of Cu, Zn and Pb were exceeded by those at only a single site (Newcastle, Glasgow and London docks, respectively). Concentrations of these metals in the sediments of the lower Irwell remain high due to the legacy of past contamination (APEM Ltd, pers. comm.). In the 1970s concentrations of heavy metals in the sediments were elevated throughout much of the River Irwell (Eyres and Pugh-Thomas, 1978) but have since decreased (White, 1993; Williamson et al., 2010) with the reduction in industrial pollution due to water quality improvements and a decline in manufacturing industry.

Analysis carried out between 2013 and 2014 in this study divided the heavy metal contaminants into two categories; high concentration (Fe, Mn and Zn), and low concentration (Cu, As, and Pb). The concentration of the first group was mostly quite high relative to standards and guidelines, especially Fe and Zn during both the seasonal and the episodic surveys. For instance, the level of Fe according to the (WFD-UKTAG.2015) should be around 1µg/L, whereas the average within the system is around 200µg/L. The level of Zn was also exceeded by far the normal range of 10.9µg/L, reaching a maximum of 500µg/L and with an average of over 50µg/L. The level of Mn was mostly within the normal range of 123 µg/l bioavailable (Environment Agency, 2015). On the other hand, the level of Cu, As and Pb was far below the normal range. For example, the normal concentration of Cu according to the WFD (WFD-UKTAG.2015) is 50µg/L whereas the concentration of Cu recorded during both the seasonal and episodic studies was <10µg/L. This figure reflects a decrease in contamination by Cu in the system as the level was by far higher in the early 1980s at nearly 200 µg/L (Dixit and Witcomb, 1983).

The level of several heavy metals thus remains high, reflecting that there are continuous sources of metal pollution entering to the system from mobilization from contaminated sediments upstream and run-off from contaminated land – both in part a legacy of past industrial pollution. Many of the contaminating heavy metals are within the black list; for instance, the level of mercury and cadmium in previous a study exceeded by far fresh water quality standards within the Dangerous Substances Directive (Williams et al., 2010). It is difficult to determine the exact sources of heavy metal in urban rivers and streams because of the wide variety of potential sources, from wash-out from contaminated air to the contribution of waste discharges via the streets, industrial, domestic, storm overflows and WwTWs (Duda, *et al*, 1982). As the lower Irwell is also highly affected by run-off of effluent from WwTWs and CSOs that are considered to be the second main source of non-historical heavy metals in urban streams (Walker, Mcnutt, and Mash, 1999). Drainage from mine-workings, either active businesses or abandoned mines, is considered to be within the most important resources of heavy metals in many streams and rivers (Harding, 2005). The old coal mine workings at Worsley, 8km above Adelphi Weir were a source of Fe pollution but concentrations have since decreased (Salford City Council, 2012). Despite the significant reduction in the mining industry (including coal mining) over the last three decades, the legacy of heavy metal pollutants that remained in fresh water ecosystems, in particular in river channels and sediments, is considered as a major factor in preventing many rivers achieving the ‘Good’ grade under the European Union’s Water Framework Directive in respect of the ecological and chemical status of surface waters (Byrne, Reid, and Wood, 2010).

4.4 Ecological status

Although freshwaters may be severely impacted by urbanization there are of course still many niches available and the lower Irwell, in common with other such systems (e.g. Beaven et al., 2001; Paul and Meyer 2001; Gurnell et al., 2007), supports a diverse but degraded biotic community. The uppermost site at Adelphi Weir is shallow and erosional and is therefore dominated by benthic algae and invertebrates. The system is too fast flowing to support a submergent macrophyte community while emergent plants such as the bulrush *Typha* common further upstream are absent, in part due to embankment, but also due to the episodically high discharge and water flow. Further down-stream reduced flow and the deep-water column supports a plankton community while the sediments and the vertical walls provide niches occupied by benthic macroinvertebrates.

The shallow nature of the Adelphi Weir site allows convention nick-net sampling of the benthic invertebrates. Downstream at the Mark Addy and beyond the system deepens and flow decreases resulting in a more homogenous substrate dominated by sand and silt but again subject to the effects of changes in water velocity. Sampling of the benthic invertebrate community is difficult at these sites and cannot be carried out safely from the bank, so colonisation samplers were employed to obtain some indication of the community, in particular diversity and degree of dominance. Although it is recognised that this approach will not fully reflect the community structure, in particular the relative numbers of individuals, it did provide an indication of the degree of biodiversity and the extent to which the community is dominated by organisms tolerant of pollution and substrate disturbance. In addition, hand net is considered more subjective and the colonisation sampler which allows standardizing the sampling effort, and hence provides more accurate assessment of differing freshwater ecosystems (De Pauw et al., 1986). It was possible to quantitatively sample the phyto- and zooplankton communities and hence obtain an indication of the degree of plankton biodiversity and dominance in the depositional sites downstream of Adelphi Weir.

The benthic invertebrate community is widely considered as a key indicator of the health of freshwater systems and a long term water data-set exists at the Turning Basin as a result on the monitoring programme by APEM Ltd. Long-term changes in phytoplankton were assessed although there is a lack of data with regard to the zooplankton community. However, both the phyto- and zooplankton were examined by myself during the seasonal and short-term study. Therefore, in this study the three major groups of fresh water biota could be examined; benthic macroinvertebrates, phytoplankton and zooplankton.

There are many differences between fresh water systems between lake, pond or stream habitats, especially in terms of the macrophyte and macroinvertebrate communities but in all cases pollution and other anthropogenic stressor results in a reduction in biodiversity (Williams, 2004). Overall, there was a lack of biodiversity in the Irwell due to pollutants discharged from the upper reaches which is subject to effects of urbanisation, including release of industrial effluents, WwTWs and run-off from CSOs. Pollution from the two tributaries – the rivers Irk and Medlock – are also an important influence on the ecology of the lower Irwell. Finally, there is the legacy of past industrial pollution directly to the lower river Irwell and Upper MSC (Wood and Lee, 1974; Williams et al., 2010).

That water quality is a key influence on the phytoplankton and benthic invertebrate community in the Irwell/MSC is shown by the marked improvement in the isolated dock basins from the 1990s following a reduction in pollution resulting from their isolation from the MSC. Installation of artificial mixing devices was an additional factor as prevented stratification and hence bottom water anoxia that would have resulted from the still high water column BOD plus the high SOD from the contaminated sediments (removal of the sediments was rejected as being too expensive). Isolation and mixing increased significantly the level of DO to >90% and hence decreased the negative impact of anoxia within the system (Hendry 1993). In addition, aeration reduced the level of phosphorous because of the elimination of sulphides from the surface sediments (Walker 1993) and reduced mobilisation from the oxic surface layer (Sas, 1989). Furthermore, oxygenation plus elimination of ingress of organic pollution form the MSC which resulted in a decrease in the level of ammonia (average of 0.12mg/L, and not exceeding 1.0mg/L) also had a positive impact on biodiversity. The improvements in water quality led to a very large increase of zooplankton community from 1/L in Basin 9 in May 1988 to 500/L by July (White *et al*, 1993). Such top-down control plus bottom up due to the reduction in water column phosphorus markedly reduced algal biomass that initially reached ‘bloom’ proportions following the reduction in SS following isolation (Mansfield et al., 2014). In parallel the number of benthic invertebrate taxa increased from 8 in 1986 to 56 in 2008 (Williams et al., 2010) and 95 in 2016 (APEM Ltd pers. comm.), including many clean water organisms such as dragonflies (*Odonata*). Of course, the Irwell and MSC cannot be isolated but interesting comparisons can be made with both the open basin 6 and the adjacent Turning Basin. The work at Salford Quays also reveals which stressors are likely to constrain the biotic community in the Irwell/MSC.

Both the MSC Turning Basin and semi-enclosed basin 6 are subject to poor water quality resulting in a lack of biodiversity and a reduction in biomass of phytoplankton. A key factor is the episodic increase in suspended solids as a result of discharge of effluents and possibly suspension of sediment during storm events which in turn affects the photic zone as indicated by the low Secchi depth. A lack of water clarity is considered as a common constraint on phytoplankton communities in systems subject to high suspended solids (e.g. Williamson et al., 2010). In contrast the isolated dock basins show Secchi depths of >5m (Williams 2010, White pers. comm.) which also allow the growth of rooted macrophytes. A further factor that will influence the phytoplankton community at both sites, plus the enclosed basins, is the absence of thermal stratification due to water column mixing resulting from natural water movement plus the presence of Helixors in the Turning Basin and the open basin dock 6. Summer stratification profoundly influences algal community structure in deep (>5m) temperate standing waters (e.g. Sigee, 2005) and, despite the water depth, Salford Quays and the Turning basin show similarities to natural shallow standing waters in that they are subject to regular mixing; albeit artificial rather than wind-induced. This continual disturbance probably accounts for the absence of the characteristic sequence of algal community's characteristic of deep temperate lakes (e.g. Sigee, 2004): a diatom-dominated community responding to increased light and temperature in the spring and a still mixed water column followed by green algae and dinoflagellates adapted to higher summer temperatures and stratification and then cyanobacteria (blue-green 'algae') that can tolerate lower light from self-shading. This group is also able to take advantage of nitrate limitation following uptake by other groups as they are able to fix atmospheric nitrogen.

There were six common phytoplankton genera, all of which were present at the downstream sites, with the same three occurring at the highest densities: the diatoms

(Bacillariophyceae) *Synedra, Nitzschia* and *Asterionella*. Other diatoms recorded were *Navicula, Diatoma* and *Melosira*. All these genera are indicative of eutrophic conditions (Reynolds, 1990) and reflect the high amounts of phosphorus in the Irwell/MSC. These diatoms occurred throughout the spring, summer and autumn and were the dominant group at all times during the seasonal survey. The historical data likewise showed that diatoms were the commonest group throughout the year over the period 2001-2012 in the Turning Basin. Diatoms were also the dominant group throughout the year in Hollingworth Lake, a naturally well-mixed system on account of its shallow nature. In contrast diatoms were only dominant in the stratified Rostherne Mere during the spring (Dean, 2004; see also Reynolds, 1990). The historical data did show a cyanobacterial peak in the Turning Basin the autumn that is characteristic of stratified waterbodies although numbers were still lower than diatoms at this time. Interestingly green algae (Chlorophyta) also show a summer peak typical of deep stratified lakes on the basis of the historical data-set and in this case numbers did exceed that of diatoms at this time. No such cyanobacterial peak was observed during the seasonal survey of 2013-2014 however.

There were few phytoplankton peaks indicative of bloom conditions of between 300/L, 500/L and 700/L respectively between June and August 2013. Likewise, mean numbers of phytoplankton, in particular diatoms exceeded 500/L over the period 2001-212. This compares with 425 cells/L of *Asterionella* at Rostherne Mere during spring of 2000 (Dean, 2004). During the summer and autumn phytoplankton numbers were generally very low and mostly less than 50/L with few peaks fewer than 200/L between March and August 2014. With regard to low density genera *Diatoma*, *Navicula* and *Melosira*, the highest numbers were again in March, and September 2013. *Diatoma* and *Navicula* were present in larger numbers at the upstream sites for reasons that are unclear but could relate to their ability to tolerate disturbance (Reynolds, 1990).

PCA indicate that BOD, ammonia and phosphate are the major contributors of the phytoplankton (largely diatom) community during the season survey. Whether these parameters directly influence the community is however unclear. Ammonia is unlikely to be toxic but may actually stimulate phytoplankton growth (Donald et al., 2011). Phosphate is unlikely to be limiting but along with BOD may reflect changes in other parameters that directly influence the community that were not measured such as flow.

Unfortunately, there was a lack of zooplankton data from 2000 to 2012 in the Turning Basin and Basin 6 due to an incomplete dataset. In this research I used cubic meter as a unit for better description of zooplankton community in the system due to the relatively low densities. With regard to the seasonal changes, from my 2013-2014 dataset there are two main groups; the first includes taxa with relatively high density such as Cyclopoida and the Cladocerans, *Macrothrix* and *Eurycercus*. These genera fluctuated between 0 and as high as 5,000/m^3. The other group consists of low density taxa such as; the cladocerans *Daphnia* and Sidae, plus diptera. Sidae varied from zero to occasionally around 2,000/m^3. Densities and composition of the zooplankton community are generally a little lower than other temperate waters with similar phytoplankton densities (Dean, 2004; Pinto-Coelho., 2005). Possible reasons for this difference are discussed below.

As expected the largest numbers of zooplankton were present during summer due to greater availability of phytoplankton. Generally, the largest densities of zooplankton were found downstream at the Turning Basin; this may reflect more stable conditions due to the deeper and, possibly, more stable water column but further work is required to test this hypothesis. The overall average downstream densities for all the zooplankton taxa increased in numbers during storm events. This may be a result of either shifting of zooplankton community within the surface water or by providing additional source of food such as organic detritus in the form of SS (Hillbricht-Tlkowska, 1977; Czerniawski, Sługocki, & Kowalska-Góralska, 2016).

An investigation over 72 hours was carried out at the deepest achievable location between the Turning Basin and Basin 6. The figures showed that there was clear evidence of vertical movement of zooplankton with different trends among different species between day and night. Such Diel Vertical Movement (DVM) of zooplankton takes place mostly between before sunrise, with movement to deeper water, and return back to the surface after sunset (Ringelberg, 1999). DVM also occurs in phytoplankton and relates to the interaction between light and chlorophyll to enhance photosynthesis (Gerbersdorf and Schubert 2011); however, DMV of phytoplankton was not observed in this study. DVM in zooplankton can reflect changes in phytoplankton plus predator avoidance (Bayly, 1986) although in this study the former was not a factor. DVM could indicate explain the lack of zooplankton in the samples during seasonal sampling events as horizontal plankton samples were taken that will only sample the first metre of the water column. Smaller numbers of zooplankton at the surface may also be in response to predation pressure of sticklebacks (*Gasterosteus aculeatus*) and young roach (*Rutilus rutilus*) as some zooplankton genera depend on vertical migration as a strategy to avoid predation (Jack et al., 2006).

It is clear that the zooplankton community shows a strong relationship with the phytoplankton community; this is very obvious from the coincidence between both communities during the short-term survey as any increase or decrease of phytoplankton was mirrored by a clear trend in zooplankton numbers. Likewise, Hendry (1991) found that total zooplankton density in both in the enclosed and open basins in Salford Quays was positively correlated with algal biomass measured as chlorophyll-a concentration. This is to be predicted as of course phytoplankton is a key source of food for herbivorous zooplankton such as the numerous cladocerans and many copepods. Organic SS (detritus) may be a further source of food for zooplankton. In a review of the trophic relationships of zooplankton in standing waters, Hillbricht-Tlkowska (1977) concluded that in many cases, and especially in eutrophic lakes, more energy is obtained by zooplankton through detritus pathways than by direct grazing on living phytoplankton. In eutrophic lakes 20% of energy was from detritus and a further is derived from 70-85% bacteria. Both these potential sources are available in the lower Irwell/MSC, including large amounts of organic SS. In support of the role is detritus is the observation of significant amounts of organic material in the gut of large apparently healthy *Daphnia* from the MSC compared to (green) phytoplankton in the same genera form the enclosed basins of Salford Quays (Keith White, pers. comm.).

There are various other possible reasons for the overall low zooplankton biomass such as predation by fish such as the stickleback (Hendry et al, 1997). Predation by the roach is also likely as this species is now commonly found throughout the lower Irwell (APEM, 2004; Williams et al., 2010). Fish are well known to be a key control on zooplankton, including in eutrophic freshwaters (Christoffersen et al., 1993; Perrow et al 1999). Predation of plantivorous fish such as the stickleback and roach by piscivorous species such as the perch *Perca fluviatilus* will reduce predation pressure on the zooplankton (Perrow et al., 1999). However, such species are currently relatively uncommon in the lower Irwell and upper MSC although numbers are increasing with improved water quality (APEM, 2004), and this change may reduce predation pressure on the zooplankton in the future. If so, this may reduce phytoplankton numbers through top-down control as has been observed in the Norfolk Broads, UK (Phillips et al., 1999).

Water quality parameters other than suspended solids such as conductivity and heavy metals may affect the zooplankton as well as the macroinvertebrate community in the MSC and the associated dock basins (White et al., 1993). That the industrial pollution legacy was a barrier to a diverse zooplankton community as shown by the improvements at Salford Quays following the isolation of all but one of the dock basins which have facilitated an increase in the zooplankton community within the isolated basins (Williams et al., 2010) although the role of piscivores introduced in the 1990s in controlling predation by plantovorus fish may have been more important. The limited data on historical zooplankton densities at the Turning Basin site indicate small numbers although data was only available before 2000; and pollution could remain a factor accounting for the low zooplankton densities observed in this study during 2013 and 2014. PCA reveal that DO, ammonia, nitrate, phosphate and discharge influence the zooplankton community. The effects of nutrients are likely to be indirect via phytoplankton but as stated above, discharge is highly likely to influence the community via changes in flow and mixing. Occasionally low DO and high ammonia concentrations could have adverse effects on zooplankton although further work is required to confirm this supposition.

One of the most responsive biological indicators of the health of freshwater ecosystems are benthic invertebrates because they are reasonably sedentary and their response to a range of pollutants, in particular organic pollution, is well understood. The hydrology and topography of each site will also influence the distribution and abundance of benthic invertebrates (Minshall, 1988) including flow rate that also affects the sediment and suspended solids concentration. These factors were therefore taken into account in the selection of the sampling stations and the positioning of the invertebrate colonisation samplers. Sampling of the colonisation units was over periods of one month to allow the benthic invertebrate assemblages to reflect the current environmental conditions.

Distribution and abundance of benthic invertebrate assemblages depend on many variables in lotic systems including stream size, water flow (Oswood, 1989) and a range of physical and chemical characteristics, including nutrient levels as these will influence benthic algal composition and hence grazers such as many Ephemeroptera (Bjorn Malmqvist & Maki, 1994), including Baetidae. In addition to direct pollution of the water course, the benthic invertebrate assemblage is markedly affected by other anthropogenic factors such as canalisation and changes to the catchment that will influence the volume as well as the quality of the run-off (Paul and Meyer, 2001; Molnar et al., 2002). It is for the above reasons that symptoms of the urban stream syndrome include reduced biotic richness and increased dominance of tolerant species (Walsh et al., 2005). It was suggested by Harper (2000) that MSC is the 'sump' of the Mersey catchment and this could be extended to the rest of the lower Irwell below Adelphi Weir. For this reason, and despite the water quality improvements discussed above, the most common benthic invertebrate taxa in the system were still largely among the list of the most pollution-tolerant organisms (Williams et al., 2010). However, studies of other urban rivers suggest that pollution has a greater impact on benthic invertebrates than channel modification (Beaven et al., 2001) although the MSC is an extreme case given the degree of re-engineering. Clearly further studies are required as to the relative importance of pollution and channel modification in the lower Irwell.

In previous studies from the early 1990s of the lower Irwell and upper MSC the most dominant benthic invertebrate genera were detritivorous and pollution-tolerant taxa such as Oligochaeta (including Lumbriculidae), the leech *Erpobdella*, the midge larvae Chironomidae and the isopod crustacean *Asellus aquaticus* (White et al. 1993). With regard to the more recent long-term data provided by APEM between 2000 and 2010, benthic invertebrates were more abundant in the semi-enclosed Basin 6 than the Turning Basin and can be categorised into a high-density group composed of *Asellus aquaticus*, Oligochaeta and Chironomidae within a range between 0 - 4,000 per sampler. In the Turning Basin, these high-density taxa were considerably lower than in Basin 6 as the range was between 0 and1,500 individuals per sampler. Taxa that were in relatively low numbers in Basin 6 were Erpobdella that ranged between from 50 to over 300, A second genera of leech *Helobdella* and the amphipod *Gammarus* were mostly below 40 individuals per sampler. The general trend was a decrease of both high and low-density genera between 2000 and 2007 despite a few peaks that occurred between 2001 and 2003. There was a lack of data between 2008 and 2013 when colonisation samplers where introduced at a number of sites along the lower Irwell/MSC between 15th of September and 15th of December 2014. Samplers distributed at each of the deeper sites (Mark Addy, Regent Road Bridge, Old Trafford Road Bridge and Turning Basin). The most common taxa found in these colonisation samplers were again worms of the family *Lumbriculidae*, the crustacean *Asellus* (Isopoda) and the leech *Erpobdellidae*. *Gammaridae* (*Amphipoda*) were also found is significant number during my study. Other taxa that were occasionally present were the mayfly (Ephemeroptera) Baetidae, and the gastropods Planorbiidae, and Physidae. General numbers fluctuated between 0 and up to 300 per sampler with just only one exceptional figure where Lumbriculidae reached between 800 and 1000 per sampler between November and December 2014.

It is apparent that all the taxa are indicative of poor water quality being characterised by four or five pollution-tolerant detritivorous taxa typical of polluted environments. The pollution-tolerance of the taxa is illustrated by their BMWP scores on a scale of 1 (very tolerant) to 10 (highly intolerant) – Oligochaeta: 1; Chronomidae: 2; Erpobdellidae: 3; Asellidae: 3; Baetidae: 4. Only Gammaridae score higher at 6. Moreover, there has not been a marked improvement over the study period of 2000-2015; in fact, there was been little improvement since the early 1990s when invertebrate community was also dominated by Oligochaeta, leeches, *Chironomidae* and *Asellus aquaticus* (White et al. 1993). Numerous studies of benthic invertebrates in urban rivers have revealed similar patterns characterised by few or absent pollution sensitive taxa and dominance by detritivorous pollution- and disturbance-tolerant taxa (see references in Walsh et al., 2005), including in the adjacent River Tame catchment (Beavan et al., 2001). In their discussion of the urban stream syndrome Walsh et al. (2005) state that ‘Globally, streams in urban areas are characterized by species-poor assemblages, consisting mostly of disturbance-tolerant taxa’ and ‘Assemblages of highly degraded streams within urban catchments are numerically dominated by a few species of oligochaetes and chironomids. We know of no studies where any other pattern has been reported.' Correlation and ordination analysis support the supposition regarding the role of disturbance and pollution in degrading the invertebrate community as the key influences were discharge and suspended solids. Conductivity, pH, ammonia, DO and phosphate were highlighted although to what extent these reflect changes in other parameters are unclear. The small change in pH for example is unlikely to influence the community although short-term deterioration in ammonia and DO may contribute to the elimination of pollution intolerant taxa (e.g. Xu et al., 2014).

The presence of *Gammarus* in the colonisation samplers at all sites is interesting. Gammaridae requires a highly oxygenated river system, in particular for reproduction but occurs, often in large numbers, in moderately polluted rivers (Hynes, 1970). Possible reasons include feeding plasticity in that Gammaridae can act as an herbivore and/or predator in different environments depending on food availability. It will therefore survive in the lower Irwell as is a generally well-oxygenated environment containing organic detritus and containing allochthonous leaf litter plus the associated microbial community (MacNeil et al, 1997).

The impact of poor water quality and the potential for improvement is illustrated by the changes in the benthic invertebrate community that occurred within three years of isolation and artificial desertification of the dock basins at Salford Quays. In addition to the above taxa genera less tolerant to pollution such as the snail *Lymnaea peregra* were commonly found, plus the water bug *Corixa* and the caddis flies *Agraylea* and *Phryganea bipunctata* within a five-year following isolation and artificial de-stratification (White et al., 1993). The invertebrate community in the lower Irwell/MSC is therefore still within the category of 'bad' and the hydrology and sedimentation behaviour are the most likely factors that support this specific community (Williams et al., 2010). While these taxonomic groups tolerate low dissolved oxygen and organic pollution it has long been recognised that their dominance is also due to disturbance, including to the deposition of silt and sand (Macan, 1962; Chutter, 1969). Therefore, the reworking of the substrate in the lower Irwell will also favour their occurrence. Benthic invertebrate assemblages are also influenced by drifting following increases in flow (Hildrew, 1976) and therefore storm events will also affect distribution. Disturbance may be a factor in the in the lower Irwell as some slightly higher scoring taxa were found at Adephi Weir including the caddis fly Hydropsychidae (BMWP

Score of 5) although the community was still severely degraded with an average score per taxa of 3-3.5 which was of the same order (3-5.5) as the lower sites.

Improvements in water quality, in particular reductions in suspended solids and less variability in the flow regime is therefore likely to result in a significant improvement in benthic invertebrate biodiversity in the deeper reaches of the Irwell and the MSC Turning Basin, as occurred in the isolated dock basins. However, the homogeneous nature of the river and lack of refuges and macrophytes would likely inhibit further diversification. Habitat diversification carried out in the enclosed dock basins from the mid-1990s (see APEM, 2008; Williams et al., 2010) has resulted in a marked further increase in diversity, with over 50 macroinvertebrate taxa recorded in 2006 (APEM 2008) and over 90 in 2015 (APEM pers. com.), compared to only nine immediately following isolation of the dock basins (APEM 2008). To fully diversify the invertebrate community in the lower river Irwell would require similar habitat diversification. Such diversification could include the use of existing artificial structures (Francis and Hoggart, 2008).

5 CONCLUSION, FUTURE WORK AND RECOMMENDATIONS

5.1 Conclusions

In summary, it is apparent that, in common with many other urban rivers both in the UK and elsewhere, the lower Irwell/upper MSC suffers from the urban stream syndrome as defined by Walsh et al., 2005. Firstly, it is characterised by a flashy hydrograph due to run-off from an urban catchment characterised by reduced soil infiltration and hence rapid response to rainfall events. It is also subject to elevated concentrations of nutrients, in particular phosphate, and a variety of contaminants. Sources of nutrients will include WwTWs (in particularly those that are not fitted with tertiary treatment to remove phosphate) and CSOs plus agricultural inputs from the upper less urbanised part of the catchment. Contaminants include suspended solids (including a significant organic component), that are likely to originate from CSOs, other point sources such as surface run-off plus diffuse sources including soil erosion. Organic contaminants including sewage from CSOs but other sources such as surface run-off contribute to the BOD and ammonia load in the river. Other contaminants include heavy metals from past and current industrial activity, including a legacy of past contamination of the sediments. The channel morphology of the entire studied reach has been markedly modified, at the first site by embankment but then as a result of canalisation resulting in an increasingly deep (2-7m) vertical-sided waterway as one approaches to Turning Basin. One difference to many other urban systems, at least in the UK, is the degree of channel modification resulting a wide (up to100m) and deep system resulting in the upper (and lower) MSC behaving like a ‘linear lake’ characterised by often very low flow and water column stratification – at least prior to installation of water column mixing devices.

Biotic richness is markedly reduced with increased dominance of tolerant species. The latter is particular apparent with regarding to the benthic invertebrate community that is characterised by pollution- and disturbance-tolerant taxa. The slow flow, vertical channel profile and water depth characteristic of the lower Irwell/MSC results in a plankton community that is the main contributor to autochthonous production in the system. The plankton community is of course the result of the channelization and would not have existed in anything like its present form in the early Nineteenth Century as records show that the River Irwell was a meandering shallow (~1m) system. The phytoplankton community is however restricted by disturbance and episodically high suspended solids despite the high phosphate concentrations that would otherwise allow a significant increase in phytoplankton biomass of ‘bloom’ proportions. Phytoplankton biomass plus, possibly, organic detritus in the form of suspended solids are adequate to support a zooplanktonic community.

In common with many other urban rivers the lower Irwell is affected by various factors relating to water quality problems, physical modification for flood prevention and flashy flows which restrict benthic invertebrate biodiversity such that a few stresses- and pollution-tolerant taxa dominate and clean water taxa such as odonata are completely absent. Other studies (e.g. Beavan et al., 2001) indicate that pollution often has a greater impact than re-engineering on invertebrate communities in urban rivers. Pollution undoubtedly has a significant impact on the benthic invertebrate community in the Irwell although the extreme channel modification of the lower reaches is an important factor in restricting the community on account of sediment deposition and highly restricted number of niches.

One aim of the study was to ascertain if water quality and ecology of the lower Irwell/MSC has improved since 2001. The results are mixed. Amounts of phosphate for example remain high and flux has increased. DO levels are good but occasional oxygen sags are observed and conductivity remains elevated. Ammonia concentrations have however decreased pointing to a continued improvement in the WwTWs. More marked improvements in water quality (albeit from a lower baseline), including from organic pollution and trace metals had however occurred over the preceding decades resulting high levels of DO, low BOD and ammonia. A key reason is the marked improvement in the quality of the effluent from WwTWs reflecting the large investment of such facilities from the late 1950s that continue to the present. In addition, oxygenation followed by artificial mixing of the water column in the Turning Basin that commenced in 2001 has contributed to this improvement in the MSC. The decrease in heavy metals is probably because of the reduction in industrial discharges that mirror the de-industrialisation of the catchment. Phosphate concentrations remain very high and indicative of eutrophication although as stated above primary production is limited by the high concentrations of suspended solids resulting from run-off and sediment resuspension at high discharge. Sources of phosphorus from fertilizers and lack of tertiary treatment (phosphate stripping) of WwTWs include the two tributaries the rivers Irk and Medlock.

5.2 Future work and recommendations for water quality Improvements

This research has identified key issues relating to the water quality in the lower River Irwell and upper MSC that decision makers need to take into consideration in order to allow the system to be improved and hence categorised as having good water quality and ecology in relation to current fresh water quality standards, in particular the WFD. This project has highlighted several research questions that need to be addressed for a better understanding of the system and hence targeted and cost-effective amelioration strategies.

- Sources of suspended solids. Elevated suspended solids arise from point and diffuse sources largely from upstream and from the tributaries, including material remobilised following deposition at low flow. This includes the effect of 'legacy pollution' that rises from dredging and resuspension of pollutants such as trace metals. Further work could include measurement of pollutant levels in the sediments plus their contribution to water column deoxygenation through measurement of sediment oxygen demand (SOD). An estimate of deposition and resuspension could be made under different hydrological regimes by using sediment traps (review: Bloeschand and Bums, 1980)

- Sources of phosphate need to be more clearly identified. This would require an extensive survey of the River Irwell and its tributaries to identify and quantify the impact of specific point sources such as WwTWs plus diffuse sources such as from agriculture. This would require quantification of amounts (flux) to take into account dilution, including under lower discharge. Agricultural sources are is likely to show seasonality due to variations in the application of fertilizers. Resources could then be focussed to dealing with the most contaminated sources, in particular point sources.

- There are many factors that contribute to level of nitrate in the river Irwell system in particular the upstream reaches. Despite the normal range of nitrate in the system, total loading is high due to the large volume of water (flux range between 20,000 and 25,000 mg/sec). Therefore, further investigation of the water pathway will provide a better understanding and provide accurate data to assess the contribution of nitrate sources, in particular at the lower parts of the system where river modification and canalization significantly affect the flow rate (Heppell et al., 2014).

- The level of heavy metals in the water column and sediments should be monitored and the sources defined to minimize the potential harmful effects of trace metals, in particular those within the black and grey list such as Hg, Cd, Cu and Zn. Of particular importance is the potential to metal remobilisation from the sediments. Such studies could include measurements of metal levels in the biota, in particular the infauna of the contaminated sediments below Adelphi Weir.

- Consideration of water conductivity and hence sources of soluble salts delivered to the system by either point source runoff or drainage from the catchment. Point sources are likely to include CSOs. The installation of further gauging facilities will allow a more accurate assessment of water conductivity through continuous monitoring, as well as assisting in identifying point sources in relation to, for example, nutrients (Ort and Siegrist, 2009).

- Intensive sampling of phytoplankton, zooplankton and benthic invertebrates is required to identify and define the most crucial factors that drives both horizontal and, in the case of plankton, vertical migration.

  Mesocosm experiments using polyethylene cylinders open to the atmosphere and to the river bottom (e.g. Romo et al., 2004) in the Turning Basin would be instructive to manipulate the environment to examine the effect on the plankton community. Of particular interest would be to monitor the effect of the decline in suspended solids through natural sedimentation on the phytoplankton community. Studies could also include the potential for grazing zooplankton to control the resulting likely 'bloom' resulting from the increased light penetration.

  Zooplankton populations in the lower Irwell and MSC can reach significant densities and would be instructive to look at the relative contribution of detrital material and phytoplankton to zooplankton nutrition given the likely decrease in suspended solids with improved water quality in the future, including the degree to which even small numbers of phytoplankton can be important in regulating phytoplankton growth in the presence of large amounts of detrital carbon (e.g. Muller-Solger et al., 2002). Mesocosm experiments would contribute to such a study as detritus levels would decline.

  Increased flow and possibly decreases in water quality are key causes of drift of benthic macroinvertebrates and it would be instructive to examine the relative importance of these factors through an examination of colonisation samplers under different flow and water quality regimes.

As stated above, key water quality problems identified in the study are high levels of phosphate and episodically high suspended solids. Discharge and flow are affected by the urbanised catchment which results a 'flashy' hydrograph due to surface run-off and the operation of CSOs. In addition, canalization has resulted in a paucity of niches due to the vertical stoned/brick-lined channel and a homogeneous substrate of silt. Some suggestions for improvements are summarised below although many will require the commitment of substantial resources.

- Levels of phosphate could be reduced by retro-fitting phosphate stripping technology to existing WwTWs plus improving agricultural practices in the upper rural catchment of the Irwell and tributaries.

- Suspended solids could be controlled by reducing the number of storm drains and/or diverting to the combined sewer system. However, this would need to be combined with the retention and storage of storm water during periods of high rainfall for subsequent treatment.

- Habitat diversification could include the installation of planted gabions positioned just below the surface above Trafford Road Bridge to the Mark Addy as interference with navigation is not now an issue. Gabions have been found to be effective in habitat diversification in the enclosed basins at Salford Quays (Williams et al., 2010). Tethered floating islands similar to those in Salford Quays are probably not a viable option due to the variable water depth and water flow. Habitat diversification could include the use of existing artificial structures (Francis and Hoggart, 2008). Previous studies have shown that a connected riparian zone is crucial to river health and water quality (Gurnell et al., 2007), and so a planning framework to facilitate such connectivity in the lower Irwell is key to habitat diversification (see APEM, 2004).

- Artificial aeration and water column mixing currently employed in the Turning Basin is effective at maintaining levels of DO above a critical threshold that could lead to anoxia (Williamson et al., 2010). Installation above Trafford Road Bridge is a possible option (APEM, 2004; Gurnell et al., 2007).

Although continued water quality improvements in the lower Irwell and upper MSC require a significant input of human and financial resources, case studies elsewhere (see for example Everard and Moggridge, 2012) plus the experience of Salford Quays provide unequivocal evidence that enhanced or restored river ecosystems are not a matter of altruism, but deliver tangible and quantifiable economic and social benefits locally and nationally.

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