Spatial and temporal variability in water quality characteristics of the Swartkops Estuary

By

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"Human interactions such as modifications of drainage basin, alterations of channel structures and water flows, and pollution have disturbed links between flow regime and water quality, and caused shifts in water quality above natural variability. Such changes started early, but their effects have become especially profound since the industrial era."

cf. Norris *et al.*, 2007

DECLARATION

I, Liaan Marie Pretorius, student number 205030998, hereby declare that the treatise/dissertation/thesis for Students qualification to be awarded is my own work and that it has not previously been submitted for assessment or completion of any postgraduate qualification to another University or for another qualification.

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SUMMARY

Water quality characteristics of the heavily urbanised and industrialised Swartkops River catchment in the Eastern Cape has been the focus of several studies since the 1970s. Overloaded and poorly maintained wastewater treatment works (WWTW), polluted stormwater runoff and solid waste have had a negative impact on the water quality status of the Swartkops River and estuary. Past studies have revealed that a distinct relationship exists between land use activities and the water quality of the Swartkops Estuary, which in turn has raised concerns pertaining to the ecological, economic, recreational, and cultural value of the estuary.

The Swartkops Estuary has a Present Ecological State (PES) of Category D[(1)] and a Recommended Ecological Category (REC)[(2)] of a C (Van Niekerk *et al*., 2014). It is known that effective management of anthropogenic impacts on coastal systems requires a thorough understanding of the system's biological responses to wastewater discharges and to hydrologic changes. For this reason, the objective of this study was to determine the current water quality status of the Swartkops Estuary, and to gain greater insight into factors controlling eutrophication. This was important as outdated water quality information was used in the Swartkops Integrated Environmental Management Plan (Enviro-Fish Africa, 2011) due to a lack of current data. To determine the current water quality status of the Swartkops Estuary the present study investigated spatial and temporal variability in physico-chemical parameters, nutrients, phytoplankton biomass and community composition, faecal bacteria, and "where possible" related this to historical water quality data. In general, points of entry into the estuary were investigated for their impact on nutrient enrichment and the bacteriological status of the estuary. Water quality surveys were completed in September 2012, November 2012, February 2013, May 2013 and August 2013.

The present study found evidence to suggest that water is not flushed as efficiently from the estuary as was previously the case, and that the natural hydrology of the estuary has been modified. These changes appear to be the effect of large volumes of wastewater discharges from the wastewater treatment works (WWTW), which has led to the additional stresses of increased vertical stratification and reduced vertical mixing. A build-up of dissolved inorganic nutrients has given rise to persistent eutrophic conditions and phytoplankton blooms occurring from the middle reaches to the tidal limit of the estuary. These findings were associated with a generally well oxygenated estuary; however, bottom water hypoxic conditions were recorded in the upper reaches of the estuary on two occasions and were generally associated with bloom-

(1) Largely modified estuary: A large loss of natural habitat, biota and basic ecosystem functions and processes have occurred.

(2) Moderately modified estuary: A loss and change of natural habitat and biota have occurred but the basic ecosystem functions and processes are still predominately unchanged.

forming flagellates. Elevated concentrations of inorganic nutrients stimulated phytoplankton to attain high biomass ranging from 0 to 248 μg l^{-1} (31.8 ± 6.56 μg l^{-1}). All nutrients displayed positive linear gradients from the mouth to the tidal limit, showed significant ($p < 0.05$) temporal and spatial variability, and were significantly ($p < 0.05$) correlated with phytoplankton biomass. Phytoplankton blooms (> 10 000 cells ml^{-1}) of several different groups were recorded from the middle reaches of the estuary to the tidal limit. Diatoms were the dominant group during increased freshwater inflow (at mean daily flow rate of 2.14 m^3 s^{-1}) and low DIP levels, whereas flagellates were generally the dominant group during reduced flow and under higher nutrient levels. Although the different tidal stages had no effect on phytoplankton biomass per se, it did support co-existance between phytoplankton groups. This was noted during the spring ebb tide in September 2012 (i.e. flagellates, diatoms and dinoflagellate) and in February 2013 (i.e. dinoflagellates, diatoms and chlorophytes). Phytoplankton blooms have become persistent in the middle to upper reaches of the estuary where chlorophyll-*a* was > 20 μg l^{-1} and cell density exceeded 10 000 cells ml^{-1}; a situation not reported in previous studies. The Motherwell Canal was and still is the main source of nitrogen (generally in the form of ammonium) to the estuary, whereas the Swartkops River is still the primary source of phosphorus to the estuary. Since the stormwater canal services the large residential area of Motherwell where leaks in the sewer system, the dumping of night soil buckets, and faulty pumps are often reported, polluted discharges from the Motherwell Canal can enter the canal at any given point. In contrast to the canal, DIP loading from the Swartkops River to the estuary generally occurred under conditions of low flow, whereas nitrogen showed no apparent relationship. Faecal bacteria originating from the Motherwell Canal had the most profound effect on the bacteriological status in the middle reaches of the estuary, whereas the Swartkops River had an intermediate effect due to bacteria die-offs occurring between the point of release from the WWTW to the riverine reaches and the tidal limit of the estuary. Nevertheless, *Escherichia coli* and enterococci levels are still high, especially in the summer months rendering the estuary unsafe for recreation during this season. Historical data on trace metals in the water column were limited and thus observations from the present study could not be concluded with much confidence. However, preliminary data suggest that levels of copper, zinc, iron and cadmium have increased by at least 90% in the estuary, at the tidal limit of the estuary and in the Markman and Motherwell canals. High inputs of nutrients, trace metals and faecal bacteria to the estuary from land-use activities indicate the necessity for remedial actions with the main objective being to conserve and protect the estuary's recreational, ecological and economic functions.

If urban runoff into the Motherwell Canal was better managed, nutrient removal methods at the three WWTW were improved and the hydraulic design capacities of the WWTW were increased then the persistent phytoplankton blooms and the health risks associated with eutrophication in the Swartkops River and estuary could be reduced.

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TERMINOLOGY AND ACRONYMS

ANZECC	Australia and New Zealand Environment and Conservation Council
BF	Brickfields
BN	Bar None
Chl-*a*	Chlorophyll-*a*
CR	Chatty River
CSIR	Council for Scientific and Industrial Research
D WWTW	Despatch Wastewater Treatment Works
DEAT	Department of Environmental Affairs and Tourism
DIN	Dissolved inorganic nitrogen
DIP	Dissolved inorganic phosphorus
DO	Dissolved oxygen
DWA	Department of Water Affairs
DWAF	Department of Water Affairs and Forestry
DWEA	Department of Water and Environmental Affairs
E. coli	*Escherichia coli*
EC	Electrical conductivity
EMP	Estuary Management Plan
EQC	Environmental quality objectives
FCB	Frans Claasen Bridge
FIB	Faecal indicator bacteria
GD	Groendal Dam
ICP-MS	Inductively coupled plasma mass spectrometry
ISO	International Standards Organisation
KJ WWTW	Kelvin Jones Wastewater Treatment Works
KN WWTW	KwaNobuhle Wastewater Treatment Works
MAP	Mean annual precipitation
MAR	Mean annual runoff
MMC	Markman Canal
MWC	Motherwell Canal
NB	Nivens Bridge
NMBM	Nelson Mandela Bay Municipality
NMMU	Nelson Mandela Metropolitan University
NWA	National Water Act
PES	Present Ecological State
POE	Permanently open estuary
ppt	Parts per thousand

PS	Perseverance
PSB	Perseverance Bridge
REC	Recommended Ecological Category
RSA DEA	Republic of South Africa Department of Environmental Affairs
RYC	Redhouse Yacht Club
SANS	South African National Standard
SB	Settlers Bridge
SD	Standard deviation
SKV	Swartkops Village
SRP	Soluble reactive phosphorus
STPP	Sodium tripolyphosphate
TC	Tippers Creek
TOCE	Temporarily open/closed estuary
TOxN	Total oxidised nitrogen
TSS	Total suspended solids
US EPA	United State Environmental Protection Agency
VSB	Van Schalkwyk Bridge
WHO	World Health Organisation
WIO	West Indian Ocean
WMS	Water Management System
WRC	Water Research Commission
WWTW	Wastewater treatment works

Chapter 1: Introduction

1.1. **Background**

One of the most critical problems that developing countries are faced with is improper management of waste generated through various anthropogenic activities (Palaniappan, 2010; Abbaspour, 2011; Kanu and Achi, 2011) as well as aged and limited infrastructure, unskilled wastewater plant operators and financial constraints (CSIR, 2007, 2010; WRC, 2013). Furthermore, rapid urbanisation has led to increased stormwater due to expanding urban areas and thus increased paved areas. Consequently, runoff of water that would previously have infiltrated the soil is discharged directly into aquatic ecosystems with potential to adversely impact the water quality and ecosystem health status of the water body (US EPA, 2003). Poor or inadequate stormwater management systems have also led to stormwater infiltrating the sewer system thereby resulting in pump station overflows and/or overloaded sewage treatment plants that do not have the required capacity.

In South Africa, water quality concerns have received a prominent position in the media, although initially, this was only related to the country's history of mining (Stander *et al*., 1970) and some industrial activities (Munnik *et al*., 2011). However, more recently, rapid deterioration of the country's water resources has been the result of other anthropogenic activities (Munnik *et al*., 2011). For example, during the 1970s concerns grew regarding the water quality of estuaries in South Africa as coastal aquatic ecosystems increasingly became the primary repositories for the disposal of waste, polluted urban runoff/stormwater, and effluents from industries and wastewater treatment works (WWTW) CSIR, 1983; Orr *et al*., 2008; CSIR, 2010). Additionally, changes in flow characteristics and related impacts on water quality characteristics have become increasingly important in developing sustainable water management strategies.

The Swartkops Estuary (33.8604° S, 25.6221° E) in the Eastern Cape is a prime example of an urban estuary where the water quality and health of the system has been noticeably impacted by land-use activities. The Swartkops River and the estuarine area meander through a highly urbanised and industrialised region of the Nelson Mandela Bay Municipality (NMBM), thus subjecting it to numerous anthropogenic activities. Water quality surveys and studies dating back as far as the 1970s (Bruggeman, 1972; Saegner, 1973; Grindley, 1974; Oliff, 1976; Dye, 1978; Erasmus *et al*., 1980; Day, 1981; Talbot, 1982) indicate that activities associated with industrial and urbanisation developments have adversely impacted the biological diversity and productivity of the estuary (Colloty *et al*., 2000).

Industrial activities in particular are a major land-use within the immediate estuarine area and within the upper reaches of the system in the vicinity of Uitenhage and Despatch, with land-use activities including numerous saltpans, sand/clay mining, brickworks, tanneries, the motor industry, wool industry, extractive/beneficiation processes, and marshalling railway yards and depots (Baird *et al*., 1986). Major water quality issues have also been attributed to sewage derived waste (i.e. faecal bacteria) from three WWTW located above the tidal limit and polluted urban/stormwater runoff from the Motherwell and Markman canals and the Chatty River in the lower to middle reaches of the estuary (Enviro-Fish Africa, 2009). The discharge of partially treated or untreated sewage wastewater, and runoff from industrial and agricultural areas and also from the high-density, low-income settlements located on the banks of the Swartkops River and estuary, are sources of nutrients and faecal pollution. The impact of these pollution types and sources have led to eutrophic conditions of the estuary and river and growth of *Eichhornia crassipes (*water hyacinth), *Phragmites australis* (common reed)*,* and *Salvinia molesta* (kariba weed) (CSIR, 1993; Enviro-Fish Africa, 2009), thereby effecting the recreational and aesthetic value of the Swartkops system. To add to this, the impact of anthropogenic activities on the deterioration in water quality has become so pronounced that it has resulted in the Redhouse River Mile swimming event being relocated to the Sundays River due to high bacteria counts detected in the water (see Appendix A: Articles 1 to 3). Furthermore, a number of sites above the tidal limit are used for cultural ceremonies by local sangomas but due to poor water quality participants can no longer drink the river water as part of their cleansing ceremonies and now depend on tap water carried to the river side.

Efficient monitoring and management of anthropogenic impacts on any given natural environment requires reliable, sufficient and consistent historical data to allow for comparisons with current water quality data. It has been noted that water quality data that was reviewed at a symposium in 1987 at the University of Port Elizabeth (now the Nelson Mandela Metropolitan University) (Baird *et al*., 1988) and other water quality data recorded during the 1980s and 1990s have been referred to and duplicated in the Swartkops Integrated Environmental Management Plan: Situation Assessment report (Enviro-Fish Africa, 2009) due to a lack of current data. In essence, historical information has been non-continuous and has been regularly rehashed without any recent measurements.

In an effort to better manage and monitor the water quality of the Swartkops Estuary and to gain a clearer perspective on the current water quality status, the objective of the present study was thus to determine the spatial and temporal water quality characteristics of the Swartkops River and estuary with special reference to the inorganic nutrient enrichment, trace metals, phytoplankton species composition and biomass and levels of faecal pollution. Historical nutrient and faecal bacteria data were collated from several sources, including the Department of Water Affairs, academic institutions, non-academic research organisations (i.e. Water Research Commission and the Council for Scientific and Industrial Research) and environmental consulting companies and thereafter, compared to water quality characteristics of the Swartkops Estuary recorded during 2012 and 2013. Water quality data were recorded at seven sites along the length of the Swartkops Estuary and at three points of entry into the Swartkops Estuary, namely Chatty River, Markman Canal and Motherwell Canal. Of the seven channel sites, the mouth and the head of the estuary were also considered entry points into the estuary. Water quality surveys were conducted in September 2012, November 2012, February 2013, May 2013 and August 2013 and correlated with rainfall and freshwater inflow. The following water quality variables were measured:

- Physical variables – temperature, salinity, pH, dissolved oxygen (DO), total suspended solids (TSS)
- Chemical variables – trace metals: aluminium, arsenic, cadmium, chromium, copper, cyanide, fluoride, iron, lead, mercury, selenium and zinc and nutrients: ammonium (NH_4^+), total oxidised nitrogen (TOxN; sum of nitriate and nitrite), dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP)
- Biological variables – phytoplankton: biomass, cell density and dominant taxa; and faecal bacteria: *Escherichia coli* and enterococci

1.2. **Aim and significance of the study**

The Swartkops Estuary is one of thirty-nine estuaries in South Africa that was given a high priority status for rehabilitation and one of the requirements for rehabilitation was water quality (pollution). Furthermore, the Swartkops Estuary is classified as one of 14 permanently open estuaries in South Africa with a Present Ecological State (PES) of Category D, meaning the system is moderately modified (Van Niekerk *et al.*, 2014) and has also been described as a Critical Biodiversity Area (Enviro-Fish Africa, 2011) by the Nelson Mandela Bay Municipality's (NMBM's) Conservation Plan. The extent and severity of the PES can only be addressed through a dedicated monitoring programme and assessments with historical data to determine trends over time. This study investigated the present water quality characteristics of the Swartkops Estuary and the pollutants that enter the estuary. Through water quality monitoring and collation of historical water quality data, this study investigated spatial and temporal water quality trends in both the estuary and river and identified water quality concerns.

To determine the suitability of the water for a particular water use, the water quality parameters were compared with the South African Water Quality Guidelines produced and published by the Department of Water Affairs in 1996 and the water quality guidelines as stipulated by the

Environmental Quality Objectives and Targets in the Coastal Zone of the Western Indian Ocean (WIO) Region (UNEP/Nairobi Convention Secretariat and CSIR, 2009). Furthermore, the present dataset was compared with historical data collated from various sources to determine whether the water quality status of the Swartkops Estuary had improved or deteriorated with time.

The main objectives of the study were to:

1. Obtain a new water quality data set for the Swartkops Estuary and five points of entry into the estuary,
2. Collate historical water quality data for both the Swartkops River and the estuary to allow for comparisons with current water quality data, and
3. Perform water quality compliance assessments on the water quality data.

It is important to note that since the study placed a great deal of emphasis on historical water quality; for each water quality variable measured, the current data were discussed, and only thereafter were differences over time concluded. Therefore, the section on ‘available literature on the Swartkops catchment’ (Chapter 3) does not include a review of water quality concerns, but instead focuses on catchment description and hydrodynamics, water and land uses, ecosystem goods and services, and applicable legislation.

1.3. **Expected results**

The overall hypothesis tested was that there has been deterioration in water quality due to an increase in nutrients, trace metals, faecal bacteria, phytoplankton biomass and phytoplankton cell density.

Based on available literature the following results were expected:

<u>Physical variables</u>

1. The Swartkops Estuary is generally well oxygenated (~7.0 mg l^{-1}).

<u>Chemical variables</u>

1. All nutrients (NH_4^+, TOxN, DIN, and DIP) will increase with distance from the mouth.
2. NH_4^+ concentrations (< 0.10 mg l^{-1}) will show localised increases in the middle reaches of the estuary due to stormwater discharges from the Motherwell Canal.

3. The Motherwell Canal and to a lesser extent the Markman Canal will be the greatest sources of nitrogen compounds to the estuary, and contributions from the Chatty and, Swartkops rivers will be smaller in comparison.
4. TOxN will be highest at the Motherwell Canal (< 2.0 mg l^{-1}).
5. DIP will increase with distance from the mouth (0.1 to 5.0 mg l^{-1}) with the primary source of phosphorus being the Swartkops River.
6. Trace metal concentrations in the water column will be spatially variable and elevated at sampling sites located within close proximity to diffuse sources (i.e. the Motherwell and Markman canals, Tippers Creek and the Swartkops River). A general increase in trace metal concentrations is expected due to increases in industrial and urban areas.

Biological variables

1. An overall increase in phytoplankton biomass is expected, with the highest concentrations occurring in the upper reaches of the estuary and in the lower Chatty River. Bloom-forming densities (> 10 000 cells ml^{-1}) will be frequently observed in the Swartkops Estuary.
2. Temporal and spatial variability in faecal bacteria will be seen with higher counts during summer months than winter months.

Chapter 2: Literature review

2.1. The role, functioning and importance of estuaries

While estuaries have been defined and described in different ways by many people across the globe, there remains no single definition that easily fits all types of estuaries. International literature defines an estuary as “a semi-enclosed coastal body of water which has a free connection with the open sea and within which sea water is measurably diluted with freshwater derived from land drainage” (Cameron and Pritchard, 1963; Pritchard, 1967). However, the most comprehensive definition for a South African estuary is defined as “that portion of a river system which has, or can from time to time have, contact with the sea. Hence, during floods an estuary can become a river mouth with no seawater entering the formerly estuarine area. Conversely, when there is little or no fluvial input an estuary can be isolated from the sea by a sandbar and become a lagoon which may become fresh, or hypersaline, or even completely dry” (CSIR, 2009). This definition is still in agreement with the current international understanding of these complex aquatic ecosystems as evidenced by Wolanski (2007) who defines estuaries as, “a semi-enclosed body of water connected to the sea as far as the tidal limit or the salt intrusion limit and receiving freshwater runoff, recognising that the freshwater inflow may not be perennial (i.e. may occur only for part of the year), that the connection to the sea may be closed for part of the year (e.g. by a sand bar) and that the tidal influence may be negligible.”

A number of legal definitions also exist in South Africa. For instance, the National Water Act (No. 36 of 1998) defines an estuary as “a partially or fully enclosed water body that is open to the sea permanently or periodically, and within which the seawater can be diluted, to an extent that is measurable, with freshwater drained from land”. The Department of Water and Environmental Affairs (DWEA) defines the geographical boundaries of an estuary as follows; “the seaward boundary is the estuary mouth and the upper boundary the full extent of tidal influence or saline intrusion, whichever is furthest upstream, with the 5 m above mean sea level contour defined as the lateral boundaries.” Furthermore, estuaries have been classified into permanently open estuaries (POEs), temporarily open/closed estuaries (TOCEs), estuarine lakes, estuarine bays and river mouths based on size of the tidal prism, mixing process and salinity profile of the water body (Whitfield, 1992) and according to their geomorphological and climatic features classified into cool temperate (west coast), warm temperate (Cape Point to Mbashe River) and subtropical (east coast) estuaries. From the list of definitions provided above, the definition given by the NWA provides the most appropriate legal definition of an estuary, as it makes provision for the several geomorphological and climatic features which are used to differentiate South African estuaries from one another (Van Niekerk and Taljaard, 2003).

Estuaries are recognised as being among the most productive ecosystems on earth (Barbier *et al*., 2011), capable of providing (free) goods and services (uses) that would otherwise have been paid for. It is estimated that a healthy estuary can produce four times more plant matter than a good ryegrass pasture, is 20 times more productive than an equivalent area of open sea and is capable of supporting up to five times as many bird species as an equivalent area of native bush. In South Africa, an estimated 20 000 tonnes of plant material is traded annually to the value of R270 million (DEAT, 2009). Estuarine ecosystems carry out other important functions such as (1) supply of organisms (food supply), sediments and organic matter (nutrients) to the coastal environment; (2) erosion and deposition control; (3) detoxification and decomposition of wastes; (4) flood and sea storm protection; (5) carbon sequestration; (6) habitat creation for aquatic and terrestrial species; (7) provide medicinal plants for subsistence and commercial use; and (8) cultivation of plants for biofuels (Koch *et al*., 2009; Barbier *et al*., 2011; Van Niekerk and Turpie, 2012). Additionally, their shallow waters render them vital nursery areas for juvenile fishes and invertebrates and they also serve as safe recreational areas.

The ecosystem services provided by estuaries can be utilised by decision makers when establishing and maintaining management structures, conservation lands, or making development decisions; putting numbers to the impacts associated with those decisions and supplying valuable data when trade-offs are being debated. These services become particularly valuable when justifying water resource management strategies and grant funding (Beever III, 2013). A fundamental purpose of any water resource management plan would therefore emphasize the link between management actions to changes in water quality and ulitimately, the associated economic gains and losses.

A summary of the types of ecosystem goods and services associated with temperate South African estuaries are provided in Van Niekerk and Turpie (2012). Chapter 3 provides an overview of ecosystem goods and services provided by the Swartkops River and estuary.

2.2. **Water quality**

The term water quality describes the “physical, chemical, and aesthetic properties of water that determine its fitness for a variety of uses and for the protection of the health and integrity of aquatic ecosystems” (DWAF, 1996a); whereby the aquatic ecosystem is recognised as part of the water resource and not as a water user (DWAF, 1999). It is important to note that the term ‘health’ is used to describe an estuary’s condition, whereas ‘integrity’ is used in the assessment of river health. Additionally, the term ‘health’ is used to make known what extent an estuary’s state deviates from its pristine condition or otherwise known as “reference condition” (Turpie *et al*., 2012).

Many physical, chemical, and aesthetic properties of water are controlled or influenced by constituents that are either dissolved or suspended in the water column (DWAF, 1996a). At a specific point along a water course, the quality of a water body reflects several major natural and man-made influences, including bathymetry, climatic and atmosphereic conditions and anthropogenic inputs. Anthropogenic inputs can be fairly constant in time, such as industrial or municipal wastewater discharge, or highly correlated to climate, example, increased discharges via agricultural/urban runoff following rainfall events. Activities that cause stream flow reduction, such as forestry, can also have a negative impact on water quality by reducing the dilution capacity of the water course (Van der Laan *et al*., 2012). The addition of various kinds of pollutants and nutrients through treated or partially treated effluent from WWTW, industrial effluents, agricultural runoff etc. into a water course can bring about a series of changes in the physicochemical, biological and chemical characteristics of the aquatic ecosystem (Maheshwari *et al*., 2011). These characteristics are ultimately influenced by flushing regimes, residence time, sediment-water exchanges and biological processes (Buzzelli *et al*., 2013), which consequently determine the spatial and temporal water quality characteristics of the water course (Duarte *et al*., 2009).

2.3. **Natural processes affecting water column chemistry**

The water chemistry of an aquatic ecosystem is affected and characterised by a wide range of natural and human influences. The most important of the natural influences are the geomorphological, hydrological (i.e. flushing mechanisms and retention time) and climatic features, since these have an effect on the quantity and the quality of water available (Bartram and Ballance, 1996; Taljaard *et al*., 2009b; Tay, 2011). Furthermore, the interaction of the river flow regime with tidal flushing defines the estuarine circulation pattern and the extent of vertical and longitudinal stratification (Perillo and Piccolo, 2011). Additionally, systems with long residence times (and low flow regimes) are particularly susceptible to developing algal blooms under conditions of nutrient enrichment (Dauer *et al*., 2000) whereas estuaries with short residence times are characterized by increased flow which usually prevent eutrophication by flushing nutrients out of the system (Grall and Chauvaud, 2002). Both local and international literature have demonstrated the significance of freshwater flow in controlling phytoplankton spatial distribution (Hilmer and Bate, 1991; Snow *et al*., 2000a; Scharler and Baird, 2005; Valdes-Weaver *et al*., 2006; Arhonditsis *et al*., 2007; Kotsedi *et al*., 2012; Yang *et al*., 2014). In the eutrophic Hudson River Estuary, for example, persistent high nutrient loading has led to significant phytoplankton productivity during extended periods of low flow, reducing transport of phytoplankton via increased residence time and also increased stratification, thereby improving and sustaining the light environment for phytoplankton productivity (Howarth *et al*., 2000). Several studies have however noted a decrease in nutrient loading and primary productivity during extended periods of below average freshwater inflow (Boynton *et al*., 2008; Abreu *et al*., 2010; Phlips *et al*., 2010; Wetz *et al*., 2011). Studies in South Africa have shown that nutrient loading to the Kromme, Gamtoos, Swartkops and Sundays estuaries vary in response to different amounts of freshwater inflows in their respective catchment areas (Scharler and Baird, 2005). It was found that the lower reaches of these estuaries were similar in their inorganic nutrient content, yet differences were seen in the upper reaches and in the inflow river waters, which would impact primary production in those regions. Additionally, nutrient cycling patterns and storage of nutrients in estuaries are further characterised with respect to global geographical locations (i.e. northern and southern hemispheres).

In temperate estuaries of the northern hemisphere rivers have a consistent annual flow and although there is more rainfall during winter months than summer months, the rainfall may vary more from week to week than from season to season (Taljaard *et al*., 2009b). Consequently, the salinity levels within the estuary always fluctuate, although with a few extreme values. In contrast to the northern hemisphere, an extended period of dry weather during the summer months coupled with varying rainfall during the winter months are typically observed with estuaries in the southern hemisphere (McLusky and McIntyre, 1995) where inflowing river water is often erratic and may cease altogether for months at a time, sometimes giving rise to hyperhaline estuarine conditions (Whitfield and Elliott, 2011).

Primary production processes and nutrient fluxes within estuaries are also largely influenced by light and temperature and as such, are characterised by geographical locations. These processes are slower in colder northern hemisphere estuaries where mean temperatures range from 5 to 15 °C, while in the southern hemisphere temperatures may reach up to 45 °C (McLusky and McIntyre, 1995). In the temperate northern hemisphere (e.g. North America and Western Europe) annual runoff is much less variable than, for example, in subtropical and temperate regions of the southern hemisphere (Braune, 1985; Eyre, 1998). Typically the larger estuaries of the temperate northern hemisphere vary from well-mixed to partially stratified (Fisher *et al*., 1988). These systems receive regular nutrient loading from the catchment (e.g. during spring) and have a high nutrient retention efficiency (i.e. a significant proportion of the nutrient load is trapped and recycled to fuel subsequent production) (Eyre, 2000). In contrast, smaller systems of the southern hemisphere (i.e. Australia and South Africa) are generally characterised by low tidal ranges and high wave energy, rendering a wave-dominated coast (Cooper, 2001).

The rate of removal of nutrients and trace metals from the water column may also determine the magnitude of spatial and temporal variability, depending on hydrological conditions, upwelling events, and extent of the salt marsh. Most removal of phosphorus in estuaries takes place through burial in aquatic sediments (Slomp, 2011). Additional retention may occur in temporarily inundated soils of salt marshes with Fe-oxide-bound phosphorus, authigenic Ca–phosphorus, and/or organic phosphorus constituting the major long-term sinks for phosphorus (Mort *et al*., 2010).

2.4. **Sources of water pollution**

Pollution in the form of nutrients, trace metals and faecal bacteria can enter a water course as diffuse (non-point) pollution or point pollution. In essence, a point source of pollution discharges to the environment from an identifiable location, whereas a non-point source of pollution enters the environment from a widespread area. Table 1 provides a summary of examples and general characteristics of each.

Urban runoff can have a serious impact on the water quality of rivers and nearby coastal systems in South Africa. As with many other developing countries this issue is related to the rapid, largely uncontrolled, growth of low cost, high-density urban settlements and conflicting demands on water resources from agriculture, nature conservation and tourism, industry and domestic potable supply utilities (Campbell *et al*., 2006). Generally, urban runoff water quality depends on local climate, hydrology, topography, geomorphology, geology and soil conditions, extent of impermeable area, urban geography, existing storm water reticulation systems, land use, and available land area. Seasonal rainfall patterns and rainfall intensity determine the occurrence and severity of the first flush effect. Dallas and Day (1992) demonstrated the need for more control of urban runoff. When these authors compared general urban runoff water quality with that of treated sewage effluent in South Africa, they found the following:

- The concentration of non-filterable residues was twenty times higher for urban runoff than for treated sewage effluent,
- Biochemical oxygen demand of urban runoff was twice that of treated sewage effluent,
- The phosphate and total nitrogen concentrations were fifteen times higher for urban runoff than for treated sewage effluent, and
- Urban runoff contained higher concentrations of suspended solids than treated sewage effluent.

Table 1: Examples and general characteristics of diffuse and point sources of pollution.

	Diffuse sources (non-point)	Point sources
Examples	• Agricultural activities (e.g. irrigation and drainage, applications of pesticides and fertilisers, runoff and erosion) • Urban and industrial runoff • Erosion associated with construction • Mining and forest harvesting activities • Pesticide and fertiliser applications for parks, lawns, roadways, and golf courses • Road salt runoff • Atmospheric deposition • Livestock waste • Hydrologic modification (e.g. dams, diversions, channelization, over pumping of groundwater, siltation)	• Hazardous spills • Underground storage tanks • Storage piles of chemicals • Mine-waste ponds • Industrial or municipal waste outfalls • Runoff • Leachate from municipal and hazardous waste dumpsites • Septic tanks.
General characteristics	• Difficult or impossible to trace to a source • Enter the environment over an extensive area and sporadic timeframe • Are related (at least in part) to certain uncontrollable meteorological events and existing geographic or geomorphologic conditions • Have the potential for maintaining a relatively long active presence on the global ecosystem • May result in long-term, chronic (and endocrine) effects on human health and soil-aquatic degradation	• Easier to control • More readily identifiable and measurable • Generally more toxic

Source: Loague and Corwin (2005)

Discharges from WWTW into receiving water courses can serve as major sources of point pollution. In South Africa, increasing pressure of existing and poorly maintained WWTW has led to the discharge of inadequately treated effluent, emphasising the need to improve methods for monitoring and sustainably managing discharged effluent.

Socio-economic and environmental factors place even further stress on deteriorating water and sanitation infrastructure. Often the discharge of partially treated effluent from WWTW results in the deposition of excessive amounts of nutrients and microbes. Excessive nutrient loading can lead to eutrophication and temporary oxygen deficiencies, disrupting biotic community structure and function (Naidoo and Olaniran, 2014). Initially, all wastewater used to be discharged directly into natural waterways, where a dilution effect would occur in conjunction with the degradation of organic matter by existing microorganisms. However, due to rising population numbers and a rise domestic and industrial waste production, the pollution of surrounding environments and consequent deterioration of public health has escalated. This resulted in an increased need for the introduction of WWTW that would aid and accelerate the purification process prior to discharge into any natural waterway (Naidoo and Olaniran, 2014).

2.5. **Indicators of water pollution**

Physico-chemical variables

Salinity

Salinity is an indication of the concentration of dissolved salts in a body of water and is usually calculated from measured conductivity. It influences dissolved oxygen concentrations, such that oxygen is low in highly saline waters and vice versa. Increased salt concentrations allow alkali and alkaline earth metals to compete with trace metal ions, thus remobilising trace metals from suspended material and sediments (Dallas and Day, 2004). Salinity is also an inverse indicator of the availability of land-derived nutrients, with low salinities (high freshwater inflow) linked to high nutrient concentrations (Pollack *et al*., 2009). Because of the limited inflow of freshwater in most South African estuaries, salinity varies over the entire range from 0 to 35 ppt (parts per thousand), and sometimes beyond (Allanson and Baird, 2008). Freshwater input establishes longitudinal and vertical salinity gradients and drives non-tidal gravitational circulation, a major contributor to flushing. The salt concentrations impact circulation within estuaries by the density variation associated with salinity, where dense saline waters tend to flow under freshwater (Lawson, 2011) giving rise to vertical salinity stratification (Table 2). Under low flow conditions, salt wedges can form and can extend up the estuary due to a loss of flow induced currents and vertical mixing. The lack of mixing between the top (fresh) and bottom (saline) layers can create anoxic environments with depth leading to the release of nutrients and toxic compounds (e.g. heavy metals) from sediments.

Under very low flow conditions, the flow of the freshwater layer may be slow-moving on top of the salt wedge. This shallow layer then also creates calm, still and warm conditions which favour the growth of phytoplankton blooms. Under high freshwater inflows, the force of the water pushes the salt wedge towards the estuary mouth, resulting in a larger body of freshwater in the estuary (Sinclair Knight Merz, 2013). If freshwater inflow is not sustained, hypersalinity at the head of estuaries are likely to occur. Hypersalinity can be detrimental for estuarine flora and fauna, mainly due to difficulties of coping with osmotic stress and lowered dissolved oxygen (Scharler *et al*., 1997). Salinity is also a critical factor affecting the survival of *Escherichia coli* (*E. coli)*, where decreasing salinity is accompanied by increasing survival (Rozen, 2001). Furthermore, it has been noted that salinity significantly affects the distrubituion of diatoms in various estuarine systems of the Eastern Cape (Minne, 2003). In the Great Fish Estuary, diatom species were associated with low salinity whereas in the Breede, Bushmans, Kowie, Mpekweni and Swartkops estuaries diatoms were associated with high salinity (Minne, 2003).

Table 2: Levels of salinity stratification based on the difference between surface and bottom salinity.

Degree of stratification	Difference between surface and bottom salinity (ppt)
Highly stratified	> 10
Partially stratified, strong	5 - 10
Partially stratified, weak	2 - 5
Vertically homogenous	< 2
Source: Livingstone (2003)	

Man-made structures, such as roads and rail bridges can also impact the magnitude of tidal and freshwater flushing and therefore influence vertical and horizontal salinity gradients. MacKay (1993) illustrated that the Wydle Bridge across the Swartkops Estuary restricts the exchange of marine water with the upper estuary. A consequence of this is that a sand bar has been created (by the levees used to support the bridge) which acts like a second mouth. Consequently, exchange of water between the upper (fresh) and lower (saline) regions of the estuary is limited at neap tide (MacKay, 1994).

The Palmiet Estuary in the Western Cape of South Africa is one example of a highly stratified estuary which is characterised by a coastal sand bar. However, unlike the Swartkops Estuary, the formation of the sand bar developed in response to naturally occuring conditions (i.e. wave-energy dynamics) of the system and not to flow induced changes due to man-made obstructions (Largier *et al*., 1992). In the Palmiet Estuary flushing times of 12 hours, two days and two weeks have been observed for the surface, intermediate and bottom waters respectively (Largier, 1986); whereas in the Swartkops Estuary, the average flushing time has been estimated to be 25 hours (Baird *et al*., 1988).

Temperature

Temperature is a limiting factor in aquatic environments as it affects physical, chemical and biological processes (Lawson, 2011) and, therefore, the concentration of many water quality variables. It also determines the availability of nutrients, toxins and the oxygen saturation level (Dallas and Day, 2004). As water temperature increases, the solubility of gases in the water decreases whereas the rate of chemical reactions generally increases together with the evaporation and volatilisation of substances from the water. The metabolic rate of aquatic organisms is also related to temperature, and in warm waters, respiration rates increase leading to increased oxygen consumption and increased decomposition of organic matter. Under combined conditions of increased temperature and nutrient levels, growth rates increase (this is most noticeable for bacteria and phytoplankton, which double their populations in very short time periods) leading to increased water turbidity, macrophyte growth and algal blooms

(Chapman, 1996). Stratification in estuaries can also occur in response to temperatures and light winds. For instance, elevated temperatures and light winds can cause thermal stratification in an estuary, where warmer fresh (less dense) water is separed from cooler (dense) seawater, which may lead to decreased mixing and dissolved oxygen, particularly at depth.

pH

The pH value of water is a measure of the acidity or alkalinity, which is an indirect measure of inflowing water sources, namely seawater and freshwater. The pH of seawater usually ranges between 7.9 and 8.2 (DWAF, 1995) while in freshwater are more or less neutral with a pH range of between 6 and 8 (DWAF, 1996a). Elevated pH values can be caused by increased biological activity in eutrophic systems. The pH values may fluctuate widely from below 6 to above 10 over a 24-hour period as a result of changing rates of photosynthesis and respiration, where very high pH values in standing waters results in extreme rates of photosynthesis (whether natural or as a result of eutrophication) (DWAF, 1996a). Changes in pH can also indicate the presence of certain effluents, together with the conductivity of the water body (Chapman, 1996). The pH of a water body influences the concentration of many metals by altering their availability and toxicity. Metals such as aluminium (Al), zinc (Zn), copper (Cu), mercury (Hg), lead (Pb), iron (Fe), cyanide (CN) and cadmium (Cd) are most likely to have increased detrimental environmental effects as a result of lowered pH (DWAF, 1996a). Changes in pH usually occur in response to discharges from chemical, pulp and paper, tanning/leather industries, mine drainage water and air pollution (Dallas and Day, 2004).

Dissolved oxygen

Dissolved oxygen (DO) refers to the amount of oxygen available to inhabitants of an aquatic ecosystem and is involved in, or influences, nearly all chemical and biological processes within water bodies. It is regarded as a good indicator of water quality, i.e. high concentrations usually indicate good water quality (Carr and Neary, 2008), although supersaturated DO usually indicate phytoplankton blooms.In the surface water, DO is often used as a proxy measure of primary productivity, as plants produce oxygen by photosynthesis, whereas DO concentration in stratified waters, below the photic zone, is an important variable for benthic organisms (Carstensen *et al*., 2011).

A decrease in DO is generally caused by the degradation of organic material through bacterial activity, which consumes available oxygen. Fluctuations in DO can occur in response to other natural processes such as temperature, salinity, turbulence, atmospheric pressure, photosynthesis by aquatic plants and algae and also land-use activities such as anthropogenic sources of nutrients and organic material such as fertilisers, deposition of nitrogen from the atmosphere, industrial and sewage treatment plant discharges, and erosion of soil containing nutrients) (Dallas and Day, 2004). Variations in DO can occur seasonally, or even over 24 hour periods, in relation to temperature and biological activity in response to photosynthetic processes and respiration (Chapman, 1996). The solubility of oxygen decreases as temperature and salinity increase and in unpolluted waters are usually close to, but less than 10 mg l^{-1}. Concentrations below 5 mg l^{-1} may adversely affect the functioning and survival of biological communities whereas oxygen concentrations less than 2 to 3 mg l^{-1} may give rise to hypoxic conditions and may cause organisms and fish to become stressed or die (Chapman, 1996). Anoxic conditions arise when there is a complete lack of oxygen (0 mg l^{-1}).

Water clarity

The immediate visual effect of a change in turbidity is a change in water clarity, in other words, water clarity refers to the depth to which light can penetrate in a water body (DWAF, 1996a). Light limitation can change the balance between nutrient uptake by autotrophs and heterotrophs, with consequences such as eutrophication. Water clarity is thus a key indicator of water quality and is closely linked to biological activities, the trophic status of the water column and other parameters including suspended particulates, chlorophyll-*a*, nutrient levels, etc. (Liu *et al*., 2013). For instance, in the Sundays Estuary, Kotsedi (2011) found a negative correlation between mean chlorophyll-*a* and mean Secchi depth, which meant that the water clarity was low and mean chlorophyll-*a* was high as the dense blooms (> 75 µg l^{-1}) decreased light penetration.

Total suspended solids

The total suspended solids (TSS) concentration is a measure of the amount of material suspended in water. An increase in the TSS may lead to a decrease in water temperature as more heat is reflected from the surface and less absorbed by the water. A variety of dissolved substances, including nutrients, traces metal ions, and organic biocides, may become adsorbed onto the surfaces of these particles. Subsequently, substances adsorbed onto particles become unavailable, and this may be advantageous in the case of toxic trace metal ions, but disadvantageous, in the case of nutrients (DWAF, 1996a).

Elevated TSS concentrations determine the degree of light penetration, hence vision and photosynthesis and can influence the transport and bioavailability of nutrients, and metals. TSS concentrations are influence by basic hydrology (e.g. flow regime, rainfall) and geomorphology, urban runoff, land erosion by wind and rain and land-based activities; i.e. domestic sewage and industrial discharge (including mining, dredging, pulp and paper manufacturing) (Dallas and Day, 2004). A long-term study on the water quality of the Knysna Estuary found a progressive decrease in the water clarity with distance from the mouth, suggesting that suspended particulate matter was primarily of fluvial origin rather than marine origin. However, in contrast, the concentration of TSS was generally higher near the mouth of the estuary compared to the middle and upper reaches. The author attributed this observation to high concentrations of planktonic organisms of marine origin which did not contribute to decreased water clarity as much as inorganic particulate matter of fluvial origin (Russell, 1996).

Chemical variables

Nitrogen

In water, nitrogen is present as the inorganic ions ammonium (NH_4^+) and nitrate (NO_3^-), with smaller amounts of nitrite (NO_2^-). These forms of nitrogen are readily available for uptake by plants and algae (i.e. they are bioavailable). Although, the ammonium ion dominates at low to medium pH values, ammonia is formed when the pH increases (Schubauer-Berigan *et al.,* 1995). Water quality reporting often refers to dissolved inorganic nitrogen (DIN), which represents the total amount of nitrogen present as ammonium, nitrate and nitrite. Increases in inorganic nitrogen loading can lead to eutrophication, with increased phytoplankton biomass, changes in species composition, the possible proliferation of harmful or toxic species and, in extreme cases, de-oxygenation of the water column (Officer and Ryther, 1980; Smayda, 1990). Nitrite and nitrate are often combined to give a total oxidised nitrogen value, with the assumption being made that the nitrite represents only a minor component of the total nitrate plus nitrite under oxic conditions (Statham, 2012). Oxygen concentrations and nitrogen fixing bacteria strongly regulate nitrogen transformation pathways including nitrification, denitrification, and dissimilatory nitrate reduction to ammonium under anoxic conditions (Bianchi, 2007). The process whereby nitrate is converted to ammonium conserves bioavailable nitrogen within the system (An and Gardner, 2002). Anoxic events are triggered by the isolation of bottom waters via stratification and subsequent depletion of oxygen through respiration of organic matter (Ritter and Montagna, 1999; Diaz, 2001). Alternatively, denitrification is an anaerobic process in which nitrate, obtained from the water column or through nitrification, is reduced to dinitrogen gas, reducing the bioavailable nitrogen load (Bianchi, 2007). Major sources of nutrients in surface waters are fertilisers, sewage and industrial effluent, stormwater runoff and dissolution of naturally occurring minerals. Nutrients originating from these sources may either enter a water body as diffuse sources of pollution or at specific sites (point source sites).

Eutrophication results from excessive amounts of nitrogen or phosphorus available for primary producers, such as planktonic algae, macroalgae and macrophytes. To successfully alleviate and manage eutrophication, identifying the nutrient(s) that is/are responsible for enhanced primary production is required. A well-studied topic in biogeochemical studies of estuaries is the change in nitrogen and phosphorus ratios in response to anthropogenic loadings, because of the impact that these ratios have on primary production (Statham, 2012). In marine systems, nitrogen has been identified as the growth limiting nutrient, whereas in estuaries phosphorus may be limiting in freshwater and nitrogen in the marine water (Howarth and Marino, 2006). Nitrogen limitation in marine systems has been attributed to the following scenarios: (1) stronger N removal in coastal waters e.g. by denitrification, (2) lower amount of N-fixing cyanobacteria ('blue-green algae') and (3) to higher rates of microbial SO_4 reduction that makes iron unable to sequester phosphorus (Addiscott, 2005; Howarth and Marino, 2006; Tammienen and Anderson, 2007). Table 3 hows inorganic nitrogen concentrations and corresponding trophic states.

Phosphorus

Phosphorus (P) is an essential nutrient element for all living organisms including phytoplankton and microorganisms, and so adequate concentrations within the water column are needed. However, excessive loading is not good either as this can overstimulate primary growth and cause a shift towards fast-growing macroalgae and phytoplankton thereby promoting bacterial growth, oxygen consumption through decomposition, and ultimately anoxic or hypoxic conditions. Apart from its application in fertilisers, phosphate is also used in detergents as a builder in the form of pentasodium tri-polyphosphate (STPP: sodium tripolyphosphate) which acts to soften hard water by complexing calcium, ferric and magnesium ions and to assist in the cleaning process by buffering the pH of the washing solution, preventing rust and corrosion and keeping dirt particles in suspension. Sodium tripolyphosphate is easily hydolysed in the presence of water to bioavailable orthophosphate (or dissolved inorganic phosphorus; DIP), an important nutrient which, when released into the aquatic environment contributes to eutrophication (Quayle *et al*., 2010). Processes responsible for DIP production include release from dissolved or particulate riverine organic matter and release of sorbed phosphorus from suspended matter with increased salinity (Froelich, 1988). In marine systems, nitrogen is the limiting nutrient, whereas phosphorus is limited in freshwater systems. However, unlike phosphorus, nitrogen can often be lost due to denitrification, which in some cases can exceed 50% of the total nitrogen input (Seitzinger *et al*., 1984). Removal of DIP from the water column may occur through biological uptake by phytoplankton (Sharp *et al*., 2009) and binding to sediment particles. Phosphate concentrations are typically lowest at the surface level as a result of the downward drift of organic debris and phytoplankton that only populate the photic zone. Additionally, DIP concentrations generally increase in the water-column with water depth due to uptake by biota in surface waters and regeneration from organic matter at depth (Conley *et al*., 2002; Luff and Moll, 2004). It is also known that, upwelling conditions can introduce phosphates to surface waters, while anoxic conditions will facilitate the return of phosphate from the sediment back into solution (DWAF, 1995). Table 4 shows inorganic phosphorus concentrations and the corresponding trophic state.

Table 3: Concentration of dissolved inorganic nitrogen (DIN) and corresponding trophic state.

Freshwater[1]		
DIN (mg l^{-1})	DIN (µg l^{-1})	**Water quality conditions**
< 0.5	< 500	Oligotrophic
0.5 – 2.5	500 – 2 500	Mesotrophic
2.5 – 10	2 500 – 10 000	Eutrophic
> 10	> 10 000	Hypertrophic
Marine water[2]		
DIN (mg l^{-1})	DIN (µg l^{-1})	**Water quality conditions**
< 0.5	< 500	N/A
Source: [1]DWAF (1996a), [2] UNEP/Nairobi Convention Secretariat and CSIR (2009)		

Table 4: Concentration of dissolved inorganic phosphorus (DIP) and corresponding water quality conditions.

Freshwater[1]		
DIN (mg l^{-1})	DIN (µg l^{-1})	**Water quality conditions**
< 0.005	< 5	Oligotrophic
0.005 – 0.025	5 – 25	Mesotrophic
0.025 – 0.25	25 – 250	Eutrophic
> 0.25	> 250	Hypertrophic
Marine water[2]		
DIP (mg l^{-1})	DIP (µg l^{-1})	**Water quality conditions**
< 0.05	50	N/A
Source: [1]DWAF (1996a), [2] UNEP/Nairobi Convention Secretariat and CSIR (2009)		

Trace metals

Metals are present in the environment in trace amounts and certain metals, such as iron (Fe), copper (Cu) and zinc (Zn), are essential for plant growth and a variety of other biological processes in aquatic species (Jackson *et al*., 2009). However, their existence in aquatic environments can be related to both natural and anthropogenic sources. Those that are naturally introduced into a water body primarily originate from soil erosion, rock weathering or the dissolution of water-soluble salts and move through aquatic environment independently of human activities, usually without any detrimental effects. Trace metals can be introduced into water bodies from multiple anthropogenic sources, namely, urban stormwater runoff, industrial effluent discharge and human waste. Industrial contributions to pollution are varied, depending on the industrial process, but can include poisonous and hazardous chemicals, nutrients, elevated salinity and increased sediments. Once introduced into surface waters, metals undergo an array of biogeochemical processes, including rapid adsorb to suspended sediment, clay minerals and organic matter and are in this manner ‘scavenged’ from the water column through flocculation, coagulation and sedimentation (Mwanuzi and De Smedt 1999; Hatje *et al*., 2003). Additionally, the toxicity and mobility of metals are continuously altered in response to fluctuations in water column salinity (Ellwood *et al*., 2008). The concentrations and bioavailability of several nutrients are also influenced when exposed to specific metals (Allanson and Baird, 2008). For example, cadmium shows a highly significant positive correlation with phosphate and nitrate at all depths, as does cyanide, copper, arsenic and phosphorus. Zinc for instance, increases the toxicity of cadmium to aquatic invertebrates, whereas a decrease in pH increases the toxicity of cyanide and in anoxic waters, the solubility of cadmium decreases (DWAF, 1995). Metals such as zinc, copper and lead have an effect on micro-organisms that recycle organic material, particularly the nitrifying bacteria, i.e. those involved in nitrogen cycling. Talbot (1988) showed that zinc concentrations below 500 $\mu g\ l^{-1}$ led to increased ammonia in experimental sediments and decreased nitrate production and related this to inhibitory effects of the metal on the micro-organisms involved. Overall, the study found zinc to have the greatest impact on nitrogen cycling, followed by copper and then lead. Additionally, metals generally do not degrade, and with continued input and limited sediment redistribution can accumulate in depositional zones to concentrations high enough to cause toxic effects to benthic and epibenthic organisms (Chapman, 1989).

In essence, adverse biological effects of trace metals are dependent upon a metal's bioavailability, solubility and mobility and subsequent uptake by organisms. These factors are dependent upon the prevailing environmental conditions (e.g. salinity, sediment redox potential, and pH) and, in estuaries, are variable both throughout the tidal cycle and on a seasonal basis. For these reasons, it is thus important to ensure that metals do not exceed normal concentrations, as they may then have detrimental long-term effects on both aquatic species and on human health, as described by Jackson *et al*. (2009) and Kanu and Achi (2011).

Biological variables

The type and biomass of biological species present in an aquatic environment reflects the quality of the ecosystem (Carr and Neary, 2008), and may include studies on heterotrophic and autotrophic protists such as phytoplankton, periphyton, zooplankton, macro-invertebrates, fish and birds. However, more commonly associated with water quality assessments, are changes in primary productivity, i.e. phytoplankton biomass, and phytoplankton community composition and succession.

Phytoplankton biomass

Measurements of chlorophyll-*a* are used as an index of phytoplankton biomass in aquatic ecosystems. Other measures that are used to evaluate the status of phytoplankton populations include the abundance and species composition of the phytoplankton community and changes in the frequency and duration of blooms (> 20 μg l^{-1} chlorophyll-*a*). In essence, the first thing to occur with nutrient enrichment of coastal waters is uptake by and stimulation of phytoplankton growth. Their horizontal and vertical distribution in the water column is controlled primarily by the degree of stratification and water motion (as either mass flow or turbulence) (Allanson and Baird, 2008; Domingues and Barbosa, 2011; George *et al*., 2013). Excessive amounts of nutrients may, under the right conditions result in overgrowth of phytoplankton (eutrophication problems) leading to low dissolved oxygen conditions as the bloom dies and the biomass decays, as well as reduced water clarity that may lead to losses of seagrasses. Anthropogenic inputs of excess nutrients to coastal water bodies have repeatedly indicated changes in ecosystem structure and function, with consequences such as major effects on phytoplankton dynamics. This phenomena is particular apparent in shallow estuaries where phytoplankton blooms are linked to fluctuations in river inflow, stratification of the water column, grazing pressure by zooplankton, and light availability (Biswas *et al*., 2009). Additionally, nutrient additions from wastewater discharge for example, may cause changes in natural nutrient ratios and/or speciation leading to blooms of opportunistic phytoplankton species, many of which are harmful or toxic (Borja *et al*., 2012). As such, some phytoplankton group species are good indicators of nutrient-related impacts.

Field data collected over two decades in San Francisco Estuary (Cloern, 1991) revealed that tides cause temporal patterns during each spring phytoplankton blooms occurred with more intense blooms during neap tide than during spring tide. Eyre (2000) illustrated phytoplankton growth in nine river-dominated subtropical east Australian estuaries and highlighted several differences compared to well mixed temperate systems. The author found that the timing and magnitude of hydrological factors to be the major feature that determined the differences in the temporal patterns of phytoplankton growth between subtropical and temperate regions. Nutrient loading and subsequent phytoplankton growth in the nine estuaries appeared to act in phases, suggesting that stored and recycled nutrients may play a smaller role in maintaining phytoplankton growth in these systems compared to temperate systems.

Phytoplankton community composition

The succession, composition, distribution and abundance of freshwater and marine phytoplankton species in the water column is regulated by environmental gradients of salinity, temperature, light availability, nutrient levels, grazing, freshwater flow and tidal stage (Lancelot and Muylaet, 2011; Shen *et al*., 2011). Phytoplankton can be classified into functional taxonomic groups (flagellates, dinoflagellates, diatoms, chlorophytes, cryptophytes and cyanobacteria, and diatoms) that play major roles in coastal production, nutrient cycling, and food web dynamics (Pinckney *et al*., 2001). General characteristics of these groups make them useful indicators of ecosystem function and change (Shen *et al*., 2011), where each phytoplankton group responds differently to nutrient availability and physico-chemical characteristics of the water column. For example, diatoms are good indicators of rapid changes in water quality chemistry (Bate *et al*., 2004) since (1) they reproduce and respond rapidly to environmental changes, providing early measures of pollution impacts and habitat restoration; (2) they have one of the shortest generation times of all biological indicators (~2 weeks) and (3) respond rapidly to eutrophication (Harding *et al*., 2005). It has also been found that waters with low nutrient levels are dominated by diatoms, whereas dinoflagellates are typically the dominant group when nutrient levels increase and during stable, stratified conditions. Moreover, in estuaries where turbidity is low, diatoms tend to be dominant in spring, when light levels are relatively low and stratification does not occur (Marshall *et al*., 2006) but dinoflagellates and cyanobacteria are usually more abundant in summer during periods of low discharge, long residence time and minimal flushing rates and turbidity (Valdes-Weaver *et al*., 2006). Several authors have found flagellates to occur under conditions of low nutrients and stratification (Margalef, 1978; Adams and Bate, 1999; Lee *et al*., 2003), whereas Legendre (1990) found that flagellates dominate under low phosphate, high ammonium concentrations and also during stratification. Conversely, Heisler *et al*. (2008) hypothesised that a shift from a diatom-dominated community towards flagellates may occur under conditions of phosphorus enrichment. Other characteristics of phytpoplankton groups are summarised in Table 5.

Theoretical explanations concerning co-existence of phytoplankton groups range from symbiotic relationships and niche-based mechanisms (Hutchinson, 1961 and Clark *et al*., 2007), emergent neutrality (co-existing species arise as the result of ecological and evolutive interactions) (Hubbel, 2001) to the presence of a predator which can disrupt competition between two species enough to promote co-existence. However, if the environment changes sufficiently through time, no single competitor can remain superior long enough to exclude other species (Hutchinson, 1961). It has also been noted that short-term physical and chemical fluctuations allow many phytoplankton species to co-exist, whereas in a rapidly changing environment; a specific phytoplankton group can dominate (Moser *et al*., 2012). In the Swartkops Estuary, several phytoplankton groups can thus be expected to occur in response to fluctuations in nutrient availability, tidal stage and freshwater flow.

Since the Swartkops Estuary is highly impacted by industrial discharges, intermittent stormwater runoff, and large volumes of treated sewage effluent (which are likely to show daily and diurnal variability in volumes and nutrient loads), so are fluctuations in nutrient availability expected.

Table 5: Phytoplankton taxonomic groups and their general characteristics.

Group	Habitat and responses to environmental conditions
Flagellates	• Stratified conditions when nutrients become depleted • Populate photic zone • May have a grazing impact on bacteria numbers • Changes in phytoplankton dominance from diatoms to flagellates constitutes as a symptom of eutrophication
Dinoflagellates	• Mostly marine • Stable, stratified conditions but when nutrients are still high • Usually outnumbered by diatoms when nutrients are high • Dense blooms during neap tides during strong vertical salinity stratification • Populate nutrient-rich water at night, take up nutrients and then swim back to the surface to utilise the abundant sunlight for photosynthesis during the day • Can form 'red tides' – rusty to dark brown coloured blooms • Benthic cysts may germinate under favourable conditions, i.e. elevated nutrients
Diatoms	• Turbulent conditions and nutrient-rich waters • Usually during spring tides, although exceptions have been reported (Kotsedi *et al*., 2012) • Growth response is directly affected by changes in prevailing nutrient concentrations and light availability • Respond quickly to changes in water quality (i.e. flowing water and cold temperatures restrict rapid algal growth) • Good indicators of total dissolved solids
Cyanobacteria	• Freshwater with high nutrient levels • Periodically stratified, long residence times (> 30 days) • Exception when N and P are elevated with increased flushing, short residence times • Low light and high turbidity • May bloom in summer (> 25 °C) (which is higher than optimum temperatures for chlorophytes and diatoms)
Chlorophytes	• Surface waters • Stable conditions when nutrients are in excess • Mostly in freshwaters
Euglena	• Usually freshwater • Anoxic conditions in sewage with high organic loads • High biomass density at elevated concentrations of ammonium

Given that the onset and occurrence of eutrophic conditions are associated with decreases in diatoms and increases in flagellates (Stoermer and Smol, 1999), phytoplankton dominance by diatom cells can be least expected in the Swartkops Estuary, whereas growth of flagelletes would be favoured. Since *Euglena* cells are biological indicators of domestic waste (Person, 1989) where both nutrients and faecal bacteria are elevated, their growth at the tidal limit of the Swartkops Estuary can also be expected as this region of the estuary receives sewage-derived effluent from wastewater treatment works. *Euglena* can also be expected to correlate with high levels of faecal bacteria.

Faecal bacteria

The monitoring of microbiological water quality is based on the concept of faecal indicator bacteria (FIB), where the abundance is related to the risk of pathogens being present. Coliform bacteria normally occur in the intestines of all warm-blooded animals and are excreted in great numbers in faeces. Faecal coliforms such as *E. coli* are non-conservative, meaning levels can change independently of how much was originally added to the surface water (DWAF, 2002). *Escherichia coli* is regarded as a good indicator of faecal contamination in water and wastewater for several reasons, (1) *E. coli* is always present in the faeces of humans and other animals such as birds, whether the individual is healthy or sick (approximately one hundred million to one billion *E. coli* cells per gram of human faeces), (2) the bacteria does not grow in the environment, such as on plants, in soil or in water, (3) *E. coli* die slowly when excreted in faeces, but survive in water at least as long as the bacteria that cause typhoid fever, cholera, and dysentery, (4) *E. coli* is relatively easy to detect by simple procedures that result in unambiguous identification of the faecal-coliform group and (5) when present above certain levels, the probability of other disease-causing organisms in the water is made known and so a potential threat to human health is revealed, and (6) it shows increased resistance to disinfectants as opposed to pathogens[(3)] (Elmund *et al*., 1999).

Sources of faecal contamination to aquatic ecosystems include effluent discharges from wastewater treatment plants, on-site septic systems, domestic and animal manure, and stormwater/urban runoff. The fate and transport of microbes in water is affected by numerous physical, chemical and biological factors (McCarthy *et al*., 2012). The disappearance of faecal bacteria in aquatic environments results from the combined actions of various biological (predation by protozoa, virus induced cell lysis and autolysis) and physico-chemical conditions (stress due to osmotic shock when released into marine water, nutrient depletion, sunlight and temperature decreases) and adsorption/desorption processes (Rozen and Belkin, 2001; McCarthy *et al*., 2012). Urban stormwater runoff has been shown to contain large quantities of both faecal bacteria and sediment (Napieralska and Goldyn, 2012) where concentration changes of both have been best described by adjusted rainfall intensity (McCarthy *et al*., 2012). Severe rainfall events can lead to a sudden initial increase in faecal bacteria levels as well as turbidity. Increased levels of nutrients and suspended solids can be problematic because nutrients adsorb onto particle surfaces, leading to increased faecal coliform growth rates. Higher temperatures also increase growth rates while extreme pH conditions increase the rate at which they decay (DWAF, 2002).

(3) Disinfectants can effectively kill pathogenic microorganisms (bacteria, viruses and parasites). Some bacteria, such as *E. coli* for example are more resistant to disinfectants than other bacteria and are therefore regarded as better indicator organisms of faecal contamination in water.

2.6. **Water quality characteristics of selected national and international estuaries**

Nutrients and freshwater flow

Land use activities define the magnitude and type of nutrients that are present in a water body. Scharler and Baird (2005) compared the nutrient dynamics of four Eastern Cape estuaries, namely, Kromme, Gamtoos, Swartkops and Sundays, each with different land uses in their catchment areas (Table 6). The highest DIP concentrations were measured in the Swartkops Estuary, and the lowest in the Gamtoos Estuary. Different land use activities gave rise to these diferences; the Swartkops systems is an urban estuary impacted by numerous industries, whereas agricultural activites prevail in the Gamtoos Estuary, where nutrients enrichment is attributed to fertiliser from agricultural return flow (Snow, 2007). Conversely, the highest DIN concentrations were measured in the Gamtoos and Sundays estuaries, due to impacts from agricultural runoff (Scharler and Baird, 2005; Snow, 2007) with the lowest measured in the Kromme Estuary (Scharler and Baird, 2005). It is clear from these studies that a high incidence of agriculture fueled elevated levels of DIN, rather than DIP in the estuarine systems.

Apart from differences in land-use activities, the Kromme, Gamtoos, Swartkops and Sundays estuaries also receive different amounts of freshwater inflows; impacting residence times, nutrient levels and thus phytoplankton biomass and cell densities would also be impacted. Scharler and Baird (2005) found that mean annual freshwater inflow rates varied to a great extent, which corroborated the variations in nutrient levels observed in the upper reaches of these systems. Inflow was the lowest and least regular in the Kromme Estuary (mean inflow rate: 1.16 m^3 s^{-1}; SD = 3.07). Excluding one high discharge event of 8.75 m^3 s^{-1} during June 1993, the mean freshwater inflow dropped to 0.07 m^3 s^{-1} (SD = 0.14). Freshwater inflow was highest and least variable in the Sundays Estuary (mean inflow rate: 2.74 m^3 s^{-1}; SD = 1.03). The mean inflow rate into the Swartkops Estuary was 1.52 m^3 s^{-1} (SD = 2.14) and in the Gamtoos freshwater inflow ranged from 0.40 to 1.60 m^3 s^{-1} under base flow conditions (Scharler and Baird, 2005), with an estimated average inflow into the estuary of less than 1 m^3 s^{-1} (Snow *et al*., 2003; Schumann and Pearce, 1997). The lower nutrients levels recorded in the Gamtoos Estuary were related to frequent flushing of the estuary thereby preventing persistent eutrophic conditions. The study also found that only in the Swartkops system was DIP concentrations in the inflowing freshwater higher compared to the upper estuarine reaches, whereas DIN was higher in the freshwater inflow in the Swartkops, Sundays and Kromme rivers. Several years later Kotsedi (2011) found that freshwater flow ranged from 0.06 to 0.08 m^3 s^{-1} (0.07 ± 0.002 m^3 s^{-1}); flow measurements far lower than those recorded during the study by Scharler and Baird (2005). DIP levels increased with distance from the mouth and ranged from below detectable limits to 0.27 mg l^{-1}; whereas DIN ranged from 0 to 1.89 mg l^{-1}, also increasing with distance from the mouth (Kotsedi *et al.,* 2012).

Table 6: Mean values of nutrient measurements for the Sundays, Kromme, Gamtoos and Swartkops estuaries.

Estuary	**Mean flow (m^3 s^{-1})**	**Total exchange times (~days)**	**Nutrients (mg l^{-1})**	**Mouth**	**Lower**	**Middle**	**Upper**	**River**
[1]**Sundays:** June 1993 to June 1994[1]	2.74 m^3 s^{-1} (SD = 1.03)	42	DIP	-	0.01	0.02	0.02	0.02
			TOxN	-	0.32	0.36	0.49	0.95
			NH_4^+	-	0.09	0.10	0.12	0.07
			DIN	-	0.41	0.46	0.61	1.02
[1]**Kromme:** June 1993 and March 1995[1]	1.16 m^3 s^{-1} (SD = 3.07)	87	DIP	0.02	0.02	0.02	0.02	0.02
			TOxN	0.24	0.13	0.13	0.13	0.04
			NH_4^+	0.08	0.07	0.08	0.08	0.13
			DIN	0.32	0.20	0.22	0.21	0.17
[2]**Gamtoos:** November 1996 and November 1998[2]	0.4 – 1.6 m^3 s^{-1} (base flow)	26	DIP	-	0.01	0.01	0.01	0.01
			TOxN	-	0.15	0.31	0.89	1.27
			NH_4^+	-	0.09	0.15	0.11	0.08
			DIN	-	0.24	0.47	1.01	1.35
[1]**Swartkops:** June 1993 to June 1994[1]	1.52 m^3 s^{-1} (SD = 2.14)	34	DIP	-	0.02	0.08	0.12	0.21
			TOxN	-	0.18	0.21	0.18	0.34
			NH_4^+	-	0.10	0.11	0.09	0.10
			DIN	-	0.28	0.31	0.27	0.43

Note: Mean inflow into the estuary of less than 1 m^3 s^{-1} has been observed by Schumann and Pearce (1997) and Snow et al. (2003). "-": No data available.
Source: [1]Scharler et al. (1997), [2]Scharler and Baird (2005), Baird (2001)

According to Scharler *et al*. (1997), the Swartkops Estuary has an intermediate position in terms of freshwater inflow between the Sundays and Kromme estuaries and together with different land-use activities gives rise to differences in nutrient dynamics within each system. However, it is hypothesised that nutrient dynamics in the Swartkops Estuary in relation to those of the Sundays and Kromme estuaries have been modified due to increased discharges from wastewater treatment works in the riverine reaches of the Swartkops system. Perissinotto *et al*. (2004) noted that return flows from municipal sewage works can increase normal estuary flow. This scenario was well illustrated in a hydrological study carried out by Ninham Shand (1994), where it was predicated that domestic and industrial water requirements in the Swartkops catchment will increase by between 40 x 10^6 m^3 y^{-1} and 55 x 10^6 m^3 y^{-1} by the year 2020 and that 65% of the water used will most likely be returned to sewage treatment works as effluent. The increase in the volume of treated effluent generated in the Uitenhage, KwaNobuhle and Despatch areas was estimated at between 6 x 10^6 m^3 y^{-1} and 16 x 10^6 m^3 y^{-1}. The study concluded that if this were all to discharge to the Swartkops River, it would increase the base flow by between 0.2 and 0.5 m^3 s^{-1}. Impacts on the Swartkops Estuary would be those of increased nutrient loading and persistent eutrophic conditions.

Compared to their permanently open counterparts, TOCEs are far more threatened by residence times (during times of mouth closure), with consequences such as over-enrichment of the waters, and eutrophication. These effects are exacerbated in systems that are at the receiving end of return flows from sewage plants, where steady flows of high nutrients levels enter the water bodies. In TOCEs, especially, this can result in nuisance algal growths that are unsightly and even cause smelly and toxic waters and result in fish kills. Several studies have illustrated the effect of a temporary/open closed mouth regimes on phytoplankton biomass and have found concentrations to be much greater than permanently open systems (Adams *et al*., 1999; Snow *et al,*. 2000 a, b; Thomas *et al*., 2005; Gama, 2008; Kaselowski, 2012) and even greater in systems which receive direct inputs of treated sewage effluent (Taljaard *et al*., 1992; Perissinotto *et al*., 2004) compared to those that don't. For example, the Mdloti and Mhlanga systems receive 8 and 20 Ml d^{-1} (megalitres per day) of treated sewage waters respectively (Lyer, 2004); a situation that has enhanced eutrophication in both estuaries. This compares to a combined daily mean wastewater volume of 29 Ml d^{-1} which is discharged into the Swartkops River from the three wastewater treatment works (DWAF, 1999). Nozais *et al*. (2001) found that in the Mdloti Estuary, DIN concentrations were lowest during a winter closed mouth state, with levels ranging from 0.01 to 0.20 mg l^{-1} and elevated during the open phase with DIN concentrations ranging from 0.09 to 2.9 mg l^{-1}. Several years later Thomas *et al*. (2005) re-studied the water quality of the estuary and found phytoplankton biomass to range from 0.9 to 111 μg l^{-1}.

Iyer (2004) found that in the Mhlanga Estuary, DIP concentrations increased during closed mouth conditions and increased during open mouth conditions, with a range of 0.7 mg l^{-1} (bottom water) to 2.7 mg l^{-1} (surface water). Conversely, DIN concentrations decreased during closed mouth conditions, with a maximum concentration of 6.4 mg l^{-1} (surface) recorded during open mouth conditions and 1.4 mg l^{-1} (bottom) recorded during close mouth conditions. Nutrient concentrations of this magnitude have been found to support a phytoplankton biomass in the range of 0.7 to 303 μg l^{-1} (Thomas *et al*., 2005). A review of South African literature has indicated that this maximum chlorophyll-*a* concentration has been the highest value recorded in any estuary in South Africa thus far; a clear illustration of the impact that large volumes of treated sewage effluents can have on the water quality status of an estuary (Perissinotto *et al*., 2004). According to the Ohlanga-Tongati Local Area Plan and Coastal Management Plan of 2007 (Ferugson, 2007), the discharge of treated wastewater effluent from WWTW at the head of Mhlanga Estuay was noted as one of the key catchment issues that has potential to impact on estuary quality, ecology and recreation. These same concerns have been raised for the Swartkops Estuary. For example, a sewage spill in September 2009 from the Kelvin Jones WWTW in Uitenhage led to a massive fish kill along the Swartkops River.

Phytoplankton blooms and dominant groups

Hilmer and Bate (1991) found that in the upper reaches of the Sundays Estuary, phytoplankton blooms formed when there was a residence time of 6 to 7 tidal cycles, where stable conditions gave rise to a bloom of dinoflagellates because the water was present for longer than the doubling time requirements to produce the bloom (Allanson and Baird, 2008). Kotsedi *et al*. (2012) found that in the Sundays Estuary different groups of bloom-forming phytoplankton dominate the water column in response to environmental changes and nutrient availability. For example, it was shown that diatoms occurred in blooms during warm, calm conditions whereas wind-mixing and reduced temperature promoted the dominance of flagellates throughout the estuary, although they were present at all times. Dominant diatom species (*Cylindrotheca closterium, Cyclotella atomus* and *Cyclostephanus dubius*) indicated brackish, nutrient-rich water. It was also observed that under low to medium flow conditions, diatoms and dinoflagellates (at a chlorophyll-*a* biomass of 99.5 $\mu g\ l^{-1}$) were dominant, whereas under high flow conditions flagellates and green algae were the dominant groups (Kotsedi *et al*., 2012). Other studies have illustrated that phytoplankton biomass relates better to nitrate concentrations and that the salinity gradient determines the species distribution (Adams and Bate, 1994). For example, in all the estuaries (Berg, Palmiet, Goukou, Gourits, Great Brak, Keurbooms, Gamtoos and Sundays) that were studied by Adams and Bate (1994), the phytoplankton community was dominated by flagellates. Although the levels of nitrate varied greatly among these estuaries, all the estuaries displayed horizontal and vertical salinity gradients. Therefore, nutrient levels alone could not have determined the species composition, but rather the stratified conditions of the water column. The study concluded that phytoplankton biomass related better to nitrate concentration and the salinity gradient determined the species distribution (Adams and Bate, 1994). However, in earlier studies, Margalef (1978) and Hilmer and Bate (1991) illustrated that dinoflagellates form dense blooms during neap tides in the Sundays Estuary when strong vertical salinity stratification had developed, where the difference in surface and bottom water salinity was greater than 5 ppt.

International literature has also illustrated the influence of fluctuations in nutrients and phytoplankton assemblages in estuarine waters (Rothenberger *et al*., 2009; Dugdale *et al*., 2012; Parker *et al*., 2012 a, b; Senn and Norvick, 2013; Egerton *et al*., 2014). For example, Egerton *et al*. (2014) found that dinoflagellates bloomed when DIN concentrations were at their lowest during a 34 day study on phytoplankton abundance and community composition in the eutrophic Lafayette River, a tidal tributary within Chesapeake Bay's estuarine complex. Positive correlations between DIP concentration and dinoflagellate abundance were also identified, but were only significant four days prior and five days after the bloom likely due to uptake by dinoflagellates during growth and regeneration after the bloom. Furthermore, diatom abundance 1 to 4 days after the bloom was significantly negative correlated with dinoflagellate abundance; as dinoflagellate abundances decreased, diatom abundances increased. In another study, it was hypothesized that high ammonium levels contribute to the low biomass and infrequent phytoplankton blooms in Suisun Bay (the northern region of the greater San Francisco Bay estuary) by inhibiting primary production, in particular growth of diatoms (Dugdale *et al*., 2012; Parker *et al*., 2012 a, b).

Phytoplankton biomass and freshwater input

Several international studies have illustrated a correlation between phytoplankton biomass and freshwater input (Dugdale *et al*., 2012; George *et al*., 2013) and in South Africa, such studies have been illustrated in the Kariega, Great Fish, Keiskamma (Allanson and Read, 1995); Swartkops and Sundays estuaries (Hilmer, 1984; Hilmer, 1990; Hilmer and Bate, 1990); in the Berg, Palmiet, Goukou, Gourits, Great Brak, Keurbooms, Gamtoos and Sundays estuaries (Adams and Bate, 1994); the Kromme, Swartkops and Sundays estuaries (Scharler *et al*., 1997) and Sundays Estuary (Kotsedi *et al*., 2012). Phytoplankton in South African estuaries have been found to be dominant in large channel-like estuaries such as the Sundays and Gamtoos estuaries, that are characterised by large catchments and high annual runoffs. These conditions allow strong river inflow to introduce nutrients to the estuaries, creating stratified conditions (Adams and Bate, 1999) that favour the growth of some phytoplankton groups (i.e. flagellates and dinoflagellates). In the Kromme Estuary, Snow and Adams (2006) found low chlorophyll-*a* due to the low nutrient freshwater inflow it receives. Average water column chlorophyll-*a* concentrations ranged from 0.6 ± 0.1 µg l^{-1} to 5.6 ± 0.3 µg l^{-1}, which was found to be consistently lower than nearby eutrophic estuaries, such as the Gamtoos. The authors further attributed the low chlorophyll-*a* levels to freshwater flow impairment following the construction of the Mpofu Dam and thereafter very few or no releases for a number of years (2002 to 2005). In the study carried out by Kotsedi *et al*. (2012) in 2007, a chlorophyll-*a* maximum of 237 µg l^{-1} was recorded in the lower reaches of the Sundays Estuary, 4.1 km from the mouth, where a freshwater flagellate (*Chlamydomonas* sp.) was the dominant phytoplankton species. In the Gamtoos Estuary, it has been found that the highest chlorophyll-*a* concentrations occur in the zone of 10 to 15 ppt; also referred to as the River Estuary Interface (REI) zone (Snow, 2000). In this region of the estuary, elevated nutrients have typically been found in high enough salinity waters that can still support marine microalgae, along with extended residence times.

In many estuaries, the residence time is primarily influenced by river discharge; hence, the development of phytoplankton blooms is often, inversely correlated with river discharge (Strayer *et al*., 2008). Because phytoplankton cells are passively transported along with the water currents, it can only increase within the estuary when net specific growth rates (i.e. the balance between phytoplankton growth and losses by lysis and grazing) exceed the residence time of the water (Lucas *et al*., 2009). In both the Sundays (Bate and Adams, 2000) and Gamtoos (Snow, 2000) estuaries, elevated phytoplankton biomass have been found to be dependent on the river flow rate and a "residence time" of 3 spring tidal cycles (or 42 days). In international literature, similar studies have been conducted by Cloern (1991; South San Francisco Bay Estuary, U.S.), Jordan *et al*. (1991; Chesapeake Bay, U.S.), Hamilton *et al*. (2000; Swan River Estuary, Australia), Acharyya *et al*. (2012; Godavari Estuary, India) and Maier *et al*. (2012; Taw Estuary, England). For instance, in the latter study, concentrations of chlorophyll-*a* were highest during low river flow and neap tides. Increased river flows resulted in a maximum chlorophyll-*a* in the outer regions of the estuary, whereas under highest river discharges, chlorophyll-*a* concentrations were further reduced. This feature was even more pronounced when spring tides coincided with high flows, with blooms generally consisting of diatoms (Maier *et al*., 2012).

Water column chlorophyll-*a* values measured in several permanently open, and also temporarily open/closed estuaries of South Africa for comparison are provided in Table 7. The table shows that phytoplankton biomass in South African estuaries are extremely variable. It also shows the combined effects of mouth closures and thus accumulated nutrients in temporarlly open/closed estuaries such as the Mdloti and Mhlaga estuarine systems which both receive large volumes of treated sewage effluents (Perissinotto *et al*., 2004). This scenario is similar to that of the Swartkops system; however, unlike the Mdloti and Mhlanga estuaries, tidal intrusion and a continuous flow of freshwater to the Swartkops Estuary ensures that the system is flushed, at least to some extent.

Metals

Studies on trace metals in surface waters of South African estuaries are limited and also highly variable due to different catchment activities and geology. In the late 1970s and early 1980s several papers were published on metal concentrations in surface waters of South African estuaries (Watling and Watling, 1976 to 1983; Watling, 1988; Table 8), providing a comprehensive set of baseline data. The assessment concluded that the Eastern Cape estuaries were not polluted in terms of trace metals concentrations. However, more recently, Orr *et al*. (2007) found that cadmium and copper concentrations in the Kariega Estuary had increased with time (Table 8). Additionally, the study found that the mean cadmium and lead concentrations were 66-fold and 19-fold higher in the dry season, respectively. In other words, the mean concentration of cadmium decreased from 3.32 ± 4.09 µg l^{-1} in the dry season to 0.05 ± 0.99 µg l^{-1} in the wet season and the mean concentration of lead decreased from 34.13 ± 42.56 µg l^{-1} in the dry season to 1.75 ± 1.01 µg l^{-1} in the wet season. This also meant that both the mean concentrations of cadmium and lead recorded during the dry season were above the target values recommended for South African coastal waters (DWAF, 1995).

Table 7: Water column chlorophyll-*a* concentrations ($\mu g\ l^{-1}$) published for permanently open estuaries and temporarily open/closed estuaries of South African estuaries.

		Province	Minimum	Maximum	Reference
POES	Berg	Western Cape	0.3	6.6	Snow and Bate (2009)
	Berg	Western Cape	-	20	Adams and Bate (1999)
	Bushmans	Eastern Cape	2.1	9.0	Jafta (2010)
	Gamtoos	Eastern Cape	1.6	115	Snow (2000)
	Gamtoos	Eastern Cape	6.5	Bloom	Adams and Bate (1999)
	Goukou	Eastern Cape	< 0.5	0.5	Adams and Bate (1999)
	Gqunube	Eastern Cape	5	15	Campbell *et al*. (1991)
	Great Fish	Eastern Cape	0	52	Allanson and Read (1995)
	Great Fish	Eastern Cape	> 100 (bloom)		Lucas (1986)
	Great Fish	Eastern Cape	1	23	Grange and Allanson (1995)
	Kariega	Eastern Cape	1	8	Allanson and Read (1995)
	Kariega	Eastern Cape	0.2	1.1	Grange and Allanson (1995)
	Kariega	Eastern Cape	0.3	9.4	Vorwerk *et al*. (2008)
	Keiskamma	Eastern Cape	0	19	Allanson and Read (1995)
	Kromme	Eastern Cape	4.7	5.5	Scharler (2000)
	Kromme	Eastern Cape	1.8	5.5	Snow (2000)
	Kromme	Eastern Cape	0.6	5	Snow and Adams (2006)
	Nahoon	Eastern Cape	1	6	Campbell *et al*. (1991)
	Olifants	Western Cape	1.7	10.3	Bate (2006)
	Palmiet	Western Cape	2	8	Branch and Day (1984)
	Sundays	Eastern Cape	> 100 (bloom)		Hilmer and Bate (1990)
	Sundays	Eastern Cape	12	23	Hilmer and Bate (1991)
	Sundays	Eastern Cape	5	35	Jerling and Wooldridge (1995)
	Sundays	Eastern Cape	-	29	Adams and Bate (1999)
	Sundays	Eastern Cape	8.6	22.8	Scharler (2000)
	Sundays	Eastern Cape	8.2	237	Kotsedi (2011)
	Swartkops	Eastern Cape	4.1	8.6	Scharler (2000)
TOCES	Great Brak	Western Cape	< 1	13.5	Adams and Bate (1999)
	Kasouga	Eastern Cape	1.49	7.72	Froneman (2006)
	Goukamma	Western Cape	0.3	112	Kaselowski (2012)
	Maitland	Eastern Cape	5.3	138	Gama (2008)
	Mdloti	KwaZulu Natal	0.09	8.6	Nozais *et al*. (2001)
	Mdloti	KwaZulu Natal	0.09	8.6	Perissinotto *et al.* (2003)
	Mdloti	KwaZulu Natal	0.87	111	Thomas *et al*. (2005)
	Mhlanga	KwaZulu Natal	0.73	303	Thomas *et al*. (2005)
	Van Stadens	Eastern Cape	0.8	13.9	Gama (2008)

Notes: permanently open estuaries (POES), temporarily open/closed estuaries (TOCEs)
Source: adapted from Kotsedi (2011)

Table 8: Metal concentrations ($\mu g\ l^{-1}$) in surface waters of Eastern Cape estuaries.

Estuary	Metal concentrations				
	Zn	Cd	Cu	Pb	Fe
Kromme	0.4	0.08	0.9	0.13	132
Gamtoos	1.6	< 0.1	0.8	0.7	550
Papenkuils	154	0.1	8.3	21.0	3 100
Swartkops	2.7	0.2	2.7	1.4	150
Sundays	2.9	0.04	4.0	1.0	378
Bushmans	0.5	0.13	1.8	0.2	302
Kariega	0.5	0.07	1.5	0.2	170
Kowie	0.6	0.05	1.7	0.2	214
Great Fish	3.0	0.07	2.2	1.1	1 446
Buffalo	12.0	0.2	5.0	41.6	166
Nahoon	4.6	0.06	0.3	97.6	91
Mossel Bay	1.8	0.8	1.8	0.3	45
St Francis Bay	1.4	0.3	1.3	0.3	275
Algoa Bay	2.2	0.2	2.2	0.9	101

Source: Watling (1988)

The authors attributed these observations to the net effect of reducing the concentrations of cadmium and lead in the surface water by flushing estuarine water into the marine environment, or via dilution with large volumes of freshwater. Amigo *et al*. (2012) found distinct relationships between salinity and trace metals in the waters of the Nerbioi-Ibaizbal River estuary which is situated in the south-east of the Bay of Biscay in Northern Spain. The estuary is located in a highly industrialised area, and is impacted by mining activities and urban wastewater. The study found that an increase in salinity led to an increase in arsenic, aluminium and iron.

As one of the most impacted estuarine systems in South Africa (Van Niekerk and Turpie, 2012), elevated levels of trace metals can be expected in the water column of the Swartkops Estuary, especially those which are associated with tanning, textile, wool processing, electroplating industries.

Faecal bacteria

Stormwater discharges can represent a large source of *E. coli* to receiving aquatic ecosystems, especially in highly urbanised settings, though impacts are dependent upon stormwater loads and the flow of the receiving water body. Daly *et al*. (2013) investigated stormwater loads that discharge into the Yarra River Estuary (Australia), including the relationship between *E. coli* concentrations and other pollutants, and the importance of stormwater drains in determining the magnitude of *E. coli* levels in the estuary.

The study found that median values of *E. coli* concentrations increased further downstream with concentrations showing larger variability in the estuary. The study also found that *E. coli* concentrations along the river were positively correlated with phosphorus and zinc, suggesting similar sources and causes, whereas salinity was negatively correlated with *E. coli*. Additioanlly, the lower concentrations in the estuary were suggested to be related to a dilution effect and to high salinity (osmotic shock). A weak trend was observed between *E. coli* levels in the stormwater drains and the entrances/downstream sampling sites in the estuary which implied that the influence of dry-weather stormwater contaminated on the estuary was limited. The low effect of the drains on the *E. coli* loads in the estuary was also explained by the large difference between the flow rate within the Yara River and the stormwater drains (Daly *et al*., 2013). This effect would typically not be observed in the Swartkops Estuary, as it receives a continuous and relatively high stormwater flow from the Motherwell Canal (MacKay, 1994).

In another study, De Brauwere *et al.* (2011) assessed the importance of tides, river discharge, and point sources on *E. coli* concentrations in the tidal Scheldt River and estuary. The river flows from the north of France to the Belgian-Dutch border and represents an extreme case of surface water pollution arising from industrial developments, intensive agriculture and animal farming. The study found that tide is crucial to explaining increased concentrations upstream of discharge sites and that it had a significant influence on the *E. coli* concentrations; both the median value and the range. However, considering the estuary as a whole, wastewater treatment plants discharging into the system did not seem to have a significant impact due to the dilution effect.

As with *E. coli*, enterococci are also used to evaluate recreational water quality and associated human health risks. In addition to their occurrence in faeces of warm blooded animals, they are also common epiphytes of the marine environment (Mote *et al*., 2012). Most countries (including South Africa and the United Sates) are finding enterococci to be the most suitable indicator for marine waters due to a number of drawbacks with the use of thermotolerant coliforms (i.e. faecal coliforms) as indicator organisms of health risks in marine waters. Epidemiological studies have showed poorer relationships between thermotolerant coliform densities and illness rates than are obtained using intestinal enterococci. Enterococci are more closely associated with human faecal matter than animal faeces and also survive longer in marine environments than faecal coliform bacteria (i.e. mortality due to salinity), making the enterococci group easier to detect (Vasconcelos and Swartz, 1976; Cabelli and Levin, 1983). Faecal pollution may thus go undetected in marine waters if *E. coli* levels are measured instead of enterococci counts.

2.7. **Water Resource Management in South Africa**

South Africa has been at the international front line of water management improvement and transformation efforts and was one of the first to engage in significant water reform. This movement included the constitutionalisation of human and ecosystem water rights and then passing it as a comprehensive new National Water Act (Act 36 of 1998) (NWA), four years after the end of apartheid in 1994 (Pollard and du Toit, 2011). The Act provides for a classification system, the “Reserve” (quantity and quality requirements for basic human needs and the aquatic ecosystem) and resource quality objectives generally referred to as Resource Directed Measures (RDMs) aimed at providing water resources the necessary level of protection in order for the resource to remain fit for use by other users (Oelofse *et al*., 2004). The Department of Water Affairs (DWA) is the official government department or “public trustee” responsible for the formulation and implementation of policies and programmes related to water resource management in South Africa.

The national government through the DWA ensures that “water is protected, used, developed, conserved, managed and controlled in a sustainable and equitable manner for the benefit of all persons” or water users (NWA, Chapter 1:3(1)). Although aquatic ecosystems are not considered to be "users" of water (in competition with other users), water, within certain quality ranges, is required to protect and maintain their health in such a way that water resources can be utilized for recreational and cultural purposes, whilst sustaining their economic and subsistence values (DWAF, 1996a). A system that is heavily impacted and under-protected is at risk of losing system resilience and as such requires maintaining a certain base level of ecological integrity and function. In order to find the most suitable level of protection (i.e. Resource Base), the NWA provides for the Ecological Reserve, which comprises descriptive and quantitative definitions of the physical structure, water quality, and water quantity required by aquatic ecosystems to maintain a desirable level of integrity (Palmer, 1999; Palmer *et al.,* 2004) thereby securing ecologically sustainable development and use of the relevant water resource (DWAF, 2006; DWA, 2010).

2.8. **Water quality monitoring and status assessments**

The National Water Act requires that monitoring of water quality constitutes an integral part of water resources management in South Africa. Numerous definitions of water quality monitoring have been provided in published literature. Chapman (1996) defines water quality monitoring as “the actual collection of information at set locations and at regular intervals in order to provide the data which may be used to define current water quality conditions”. The International Standards Organisation (ISO) defines water quality monitoring as “the programmed process of sampling, measurement and subsequent recording or signalling, or both, of various characteristics, often with the aim of assessing conformity with specific objective" while the South African Strategic Framework for National Water Resource Quality Monitoring (DWAF, 2004b) defines water resources quality monitoring rather than water quality monitoring as the acquisition of data, management and storage of data and the generation and dissemination of information on the physical, chemical, biological and ecological attributes of the water resource (DWAF, 2004a). The specific objectives of a monitoring programme depend on the information required, which generally includes one of the following: 1) compliance auditing (including legal), (2) resource status and trend reporting, (3) assessment of fitness for use, (4) water quality objectives auditing, and (5) special studies (Van Niekerk *et al.*, 2002). An estuary must be managed to avoid, minimise or mitigate significant negative impacts such as reduced water flows and loss of habitat or species in response to land-based activities. To achieve this, the PES of an estuary must be determined, and the REC must be set. Table 9 and the description below provide a definition of each.

Present Ecological State

The Present Ecological State a measure of the health of a resource based on a comparison between the Reference Condition and the Present State (DWAF, 2008) and provides the point of departure for the development of any management objectives. The PES is assessed in terms of the degree of similarity to reference conditions. This helps to identify what may be desirable or achievable as a future management class. Chemical and biotic response data are linked to a class (Natural, Good, Fair, or Poor), where data from one to three years prior to the assessment of the PES are used. If the data record is poor (e.g. less than monthly sampling frequency), then data from up to, but no longer than five years prior to the assessment can be used (Palmer *et al*., 2007).

Recommended Ecological Category

The Recommended Ecological Category is one of the first four ecological categories (A to D) utilized in identifying the present status. This category is the target for protection and management of the resource which could be the same as the Present Ecological Status, or higher if an improvement in resource condition is desired. The ecological components to be assessed include water quantity (i.e. the magnitude, duration, timing and reliability of the flow), water chemistry (total dissolved solids, pH, dissolved oxygen, temperature, total suspended solids, nitrogen and phosphorus, and toxic substances), and ecological state (bioassessment and geomorphology). The classification system provides the guidelines and procedures for classifying different classes of water resources. Each class in the classification system needs to state what kinds of impacts on the water resource are acceptable and what kinds of impacts are not acceptable in order to protect the resource.

Table 9: Relationship between Present Ecological Status and Ecological Category.

Present Ecological State (PES)	Description	Ecological category	Corresponding Management Class*
A	Unmodified or or approximates natural condition	A	Natural (Class I) – Minimally used/impacted
B	Largely natural with few modifications	B	Good (Class II) Moderately used/impacted
C	Moderately modified	C	Fair (Class III) Heavily used/impacted
D	Largely modified	D	
E	Seriously degraded	E	Poor
F	Critically degraded	F	

** The Management Class is the desired state selected after various social, economic and ecological implications have been considered for a set of use and protection scenarios. The management class must capture the most desirable balance between use and protection. It is determined in relation to the present state, but at a level which represents a goal of no further degradation for water resources which are slightly to largely modified, and at least a move towards improvement for water resources which are critically modified.*

Source: modified from DWAF (2008).

The class also needs to state how much water can be used from the water resource. Furthermore, the classification system must satisfy the water quality requirement of users without significantly altering the natural water quality characteristics of the water resource and it must also take into account the use of water for particular activities that need to be controlled in order to protect the water resource. The classes permit the national government to group water resources into three management classes, namely; natural (class I), good (class II), fair (class III) and poor, while the Ecological Categories (which indicate the potential management target for a water resource) range from; Category A (unmodified, natural) to Category F (extremely degraded). The Ecological Category is allocated on the basis of the importance score, using the PES (DWAF, 2008).

2.8.1. Estuary health assessment

The general consensus is that the conservation status of an estuary is expected to lead to a recovery of ecosystem health and the supply of ecosystem services that requires an understanding of its current health including abiotic and biotic components (Turpie *et al*., 2012). The accelerating deterioration of estuaries and coastal waters has led to an increased demand for monitoring different properties of the marine ecosystem. A baseline study and health assessment of estuaries provides a description of the estuary in its present state and quantifies its health in terms of the Estuary Health Index (Turpie *et al*., 2012). It also provides a description of the characteristics and functioning of all major abiotic and biotic aspects of the system and their relationships to one another, and the flow and non-flow related pressures and impacts on the system. Although baseline studies and long-term monitoring programmes have different purposes, it is important that long-term monitoring programmes follow on from similarly structured baseline studies. In essence, the monitoring activities selected for the long-term monitoring programme should be derived from the monitoring activities conducted as part of the baseline studies, but implemented on less intensive spatial and/or temporal scales (Turpie *et al*., 2012). Long-term monitoring data should then be used to (1) determine the impacts of industrial, agricultural, and other human activities; (2) quantify the effectiveness of policies and management plans; (3) develop water-management models; (4) prioritize where management effort should be concentrated (i.e. natural flows); and (5) communicate to key stakeholders about pollution, human health concerns, and degraded ecosystems (Palaniappan *et al*., 2010).

2.8.2. Water quality guidelines

To determine the effectiveness of management actions and achievement of water quality objectives (with specific reference to Resource Directed Measures), a set of performance indicators are required, best described as Water Quality Guidelines. The South African Water Quality Guidelines serve as the primary source of information for determining the water quality requirements of different water uses and for the protection and maintenance of the health of aquatic ecosystems (DWAF, 1996b). The guidelines are used for making informed decisions concerning the physical, chemical, biological and aesthetic properties of water. Essentially, the guidelines consist of water quality criteria, in particular, the Target Values, defined as the 'level of a particular water quality/constituent at which no detrimental impact should occur' (DWAF, 1995). There are currently no South African nutrient target values (not standards) for estuarine waters. Grobelaar (1992) once stated that over-simplified models of nutrient determinations are inadequate for coastal marine ecosystems where hydrodynamic factors, freshwater inflow and volumes, and high turbidity can modify the effects of nutrients within an estuarine system. The concern here is that such models may have the potential to exacerbate the magnitude of nutrient loads detected in the water column. Even if standards from other countries (e.g. Australia) were adopted, then virtually all of South Africa's estuaries would be classified as eutrophic (Harrison *et al*., 2000). This is because fluvial nutrient concentrations of Australian coastal waters are naturally high. Coastal ecosystems of KwaZulu-Natal illustrate this scenario well, where nutrient concentrations, derived from detrital sources, result in relatively high background nutrient levels. However, according to the Department of Water Affairs, international marine water quality guideline documents can in fact be considered where South Africa does not have the recommended environmental target values (DWAF, 2004b). These documents include the guidelines for Australia and New Zealand (ANZECC, 2000), Canada (Environment Canada, 2002) and United States Environmental Protection Agency (US EPA, 2002). Alternatively, water quality target values set for coastal zones of the West Indian Ocean Region may also be used (UNEP/Nairobi Convention Secretariat and CSIR, 2009). Generic environmental quality objectives (EQO) proposed for the West Indian Ocean regions include, the protection of coastal aquatic ecosystems, recreational use, marine aquaculture and industrial use, where target values for coastal aquatic ecosystems, constitute, objectable matter (i.e. aesthetic criteria), physico-chemical variables, nutrients, and toxic substances (i.e. trace metals).

With reference to recreational waters, suitable drinking water quality guidelines (e.g. SANS (South African National Standard) 241:2011; World Health Organisation (WHO), 2011) may be consulted to make preliminary risk assessments where toxic substances could be present at levels posing a risk to human health (UNEP/Nairobi Convention Secretariat and CSIR, 2009). However, South African water quality guidelines for suitable levels of faecal bacteria in recreational waters are available. These guidelines were revised in 2012 (RSA DEA, 2012) and recommend target values for enterococci and *E. coli* based on an evaluation of a long-term dataset. Not only indicated in the revised South African water quality guidelines, but also internationally, the accepted norm for recreational water compliance in terms of faecal bacteria (i.e*. E. coli*) is based upon percentage compliance levels, typically 90% and 95 % compliance levels (i.e. 90% and 95% of the number of samples collected over an extended period of time, must lie below specific values in order to meet the standard). Table 10 shows the relevant water quality guidelines for both nutrients and trace metals.

Microbiological assessments must be an evaluation of data collected over a fixed period of time, typically five years and on a monthly basis (or ideally every two weeks) (RSA DEA, 2012). In some instances (i.e. baseline studies) the grading cannot be applied due to limited data. Additionally, bacteria loads may from time to time exceed water quality guidelines, and be regarded as infrequent "spikes" while in other instances; elevated bacteria levels appearing above the guideline may persist for extended periods of time. Therefore, a single sample count value has been introduced and may be used for compliance analyses. The revised South African recreational water quality guidelines of 2012 proposed that for enterococci counts, a 'single sample target value'(4) approach be used (RSA DEA, 2012), where; no action is required where enterococci counts are less than 240 per 100 ml (based on a single water sample), sampling must be increased to daily (using first sample to confirm problem) if enterococci counts exceed 240 counts 100 ml^{-1} (based on a single water sample) and finally, action is required if the two consecutive samples exceed 380 counts 100 ml^{-1} (RSA DEA, 2012).

(4) A "single" bacteria count is compared against a bacteria compliance target as apposed to a mean bacteria count which is based on microbiological data collected over five years. The "single sample target value" allows for a timeous response and implementation of appropriate management actions to any day-to-day situation that could pose potential risk to human health (RSA DEA, 2012).

There are currently no single sample target values for *E. coli* in marine or estuarine waters as enterococci bacteria are the preferred indicator of faecal pollution.

Escherichia coli counts in estuarine waters are normally orders of magnitude greater at the tidal limit than at the seaward end and tidal movements and variations in freshwater inflows can produce continual changes. This phenomenon causes the bacteria counts to vary throughout the estuary and the complexity creates difficulty in assessing the bacteria counts (i.e. difficulties arise when deciding if the bacteria counts for estuarine water samples are within an acceptable range relative to their corresponding salinities). Table 11 and Table 12 show the risk-based criteria for *E.coli* and enterococci counts respectively.

Table 10: Relevant water quality guidelines for recreational water use and aquatic ecosystem health, including guidelines for freshwater and coastal aquatic ecosystems.

	South Africa			West Indian Ocean
Water users	**Recreational**	**Natural Environment (marine)**	**Aquatic cosystem (freshwater)**	**Coastal aquatic Ecosystems (marine)**
Guideline source	**SANS 241:2011 (2011)**	**DWAF (1995)**	**DWAF (1996a)**	**UNEP/Nairobi Convention Secretariat and CSIR (2009)**
DIN ($\mathrm{mg\ l^{-1}}$)			0.5	0.5
DIP ($\mathrm{mg\ l^{-1}}$)			0.005	0.05
CN ($\mu\mathrm{g\ l^{-1}}$)	≤ 70	12		
As ($\mu\mathrm{g\ l^{-1}}$)	≤ 10	12		
Hg ($\mu\mathrm{g\ l^{-1}}$)	≤ 6	0.3		
Se ($\mu\mathrm{g\ l^{-1}}$)	≤ 10	-		
F ($\mu\mathrm{g\ l^{-1}}$)	≤ 1 500	5 000		
Fe ($\mu\mathrm{g\ l^{-1}}$)	≤ 2 000	-		
Al ($\mu\mathrm{g\ l^{-1}}$)	≤ 0.3	-		
Cd ($\mu\mathrm{g\ l^{-1}}$)	≤ 3	4		
Cr ($\mu\mathrm{g\ l^{-1}}$)	≤ 50	8		
Cu ($\mu\mathrm{g\ l^{-1}}$)	≤ 2 000	5		
Pb ($\mu\mathrm{g\ l^{-1}}$)	≤ 10	12		
Zn ($\mu\mathrm{g\ l^{-1}}$)	≤ 5 000	25		

Table 11: Risk-based range for *Escherichia coli* for recreational waters in the coastal marine environment.

	Estimated risk for exposure	***E. coli* (counts 100 $\mathrm{ml^{-1}}$)**
Excellent	2.9% GI illness risk	≤ 250 (95 percentile)
Good	5% GI illness risk	≤ 500 (95 percentile)
Sufficient or Fair (minimum requirement)	8.5% GI illness risk	≤ 500 (90 percentile)
Poor (unacceptable)	> 8.5% GI illness risk	> 500 (90 percentile)

Note: 'GI' = gastrointestinal
Source: RSA DEA (2012)

Table 12: Water quality assessment criteria for single sample assessments of enterococci counts in recreational waters of the marine environment.

Enterococci single count value	**< 240 enterococci 100 ml^{-1}**	**> 240 enterococci 100 ml^{-1}**	**> 380 enterococci 100 ml^{-1} *(resample as soon as possible)***
Assessment mode	**Survellance (green) mode**	**Alert (amber) mode**	**Action (red) mode**
Action required	No action, continue routine monitoring programme	Increase sampling to daily (using first sample to confirm problem)	Increase sampling to daily (using second sample to confirm problem)

Source: RSA DEA (2012)

Chapter 3: Available information on the Swartkops catchment

3.1. **Catchment description**

Over the past five decades various aspects of the Swartkops catchment have been researched and the area described in detail by numerous authors including MacNae (1957), HKS (1974), Reddering and Esterhuyzen (1981), Baird *et al*. (1986), Fromme (1988) and Haigh (2002). The Swartkops Estuary is located approximately 15 km north of the Port Elizabeth CBD (Central Business District) in the Eastern Cape, South Africa (Baird, 2001) and has its origin in the Groot Winterhoek Mountains (Reddering and Esterhuysen, 1981). The system is characterised by a permanently open connection with the sea where riverine water enters into Algoa Bay in the Indian Ocean. The estuary itself is approximately 16.4 km long and the total length of the Swartkops River is 155 km from the mouth to its origin (Baird *et al*., 1986).

The Swartkops River catchment is mostly forested (Reddering and Esterhuysen, 1981) and includes the municipal areas of Uitenhage, KwaNobuhle, Despatch and Ibhayi Township (see Chapter 4: Figure 2; Binning and Baird, 2001). The catchment spans an area of 1 360 km^2 (Reddering and Esterhuysen, 1981), is about 120 km long and 42 km wide in its greatest dimension (Baird *et al*., 1986) and is considered to be small in comparison to the Sundays and Gamtoos river catchments that lie on either side (Binning, 1999). The Swartkops valley is densely populated in the lower reaches of the catchment and around most of the estuary, comprising of both formal and informal settlements and numerous industrial activities such as clay mining, salt works, sewage treatment works, wool washeries and tanneries (Enviro-Fish Africa, 2009). The catchment area consists of four tertiary sub-catchments: the river system consists of two main (Elands and Kwazunga rivers) and two subsidiary tributaries (Brak and Chatty rivers) (Klages *et al*., 2011), the former originating in the Winterhoek Mountains and joining just above Uitenhage in an area known as Kruisriver to form the Swartkops River and the latter, originating in the plains north of the Nelson Mandela Bay and joining the Swartkops River after confluence of the Elands and KwaZunga rivers. The Elands River has two main tributaries, the Sand River in the north and the Bulk River located in the south, both originating in the Elandsberg (Haigh, 2002). The Brak River forms a confluence with the Swartkops River in Uitenhage, while the Chatty River (see Chapter 4: Figure 1) flows through an informal settlement and enters the Swartkops Estuary in the lower reaches, upstream of the Swartkops Village area. Three dams occur in the upper reaches of the Swartkops River. The Groendal Dam (see Chapter 4:), which is located approximately 35 km upstream of the estuary, was built to completion in 1939 and has a storage capacity of ~12 x 10^6 m^3. The dam retains approximately 16% of the mean annual runoff (MAR) and reduces freshwater inflow by 5%. The

Sand and Bulk River dams are small and have little effect on freshwater inflow (Baird *et al*., 1986; Fromme, 1988). After the confluence of the Elands and the KwaZunga rivers, the Swartkops River meanders through a wide alluvial floodplain past the industrial area of Uitenhage and residential areas of Despatch to the tidal limit of the estuary. The channel cuts deeply into the sedimentary deposits and is severely disturbed both upstream and downstream of Uitenhage (Haigh, 2002).

The Swartkops Estuary is described as a small, shallow, turbid, well-mixed(5) temperate estuary (Emmerson, 1985; Baird *et al*., 1986; MacKay, 1993) surrounded by residential and industrial developments. The estuary experiences significant tidal water exchange and is maintained by strong currents, rather than high run-off (Binning, 1999). The estimated area covered by the estuary is 4 km^2 with a tidal prism of approximately 2.9×10^6 m^3 and a capacity of 5.1 and 2.2×10^6 m^3 for high and low tide respectively. Furthermore, the average flushing time is about 22 hours (Baird *et al*., 1986; Winter and Baird, 1991) and tidal differences within the estuary range from 0.5 m on a neap tide to 2 m on a spring tide with circulatory patterns strongly influenced by variations in spring and neap tides. Consequently, flushing of the estuary in the lower region is rapid and complete over each tidal cycle; however circulation in the upper region is limited due to flow constriction caused by sand banks positioned behind the rail/road bridge that crosses the estuary (Goschen and MacKay, 1992). McLachlan (1972) divided the estuary into four regions: the mouth, lower, middle and upper reaches. The mouth region extends for a kilometre upstream and includes the rocky stretch in front of the Amsterdamhoek residential area (Hanekom, 1980). Large intertidal mudflats, islands and salt marshes increase in coverage near the mouth and give way to large sandbanks and a rocky embankment north and south of the river mouth. The lower reaches of the estuary stretches from Amsterdamhoek to Brickfields and is characterised by extensive supra-tidal salt marshes (Melville-Smith and Baird 1980; Winter 1990) of 1.8 – 2.4 km^2, a substrate comprising of a mixture of sand and mud as well as extensive inter-tidal flats. In the middle reaches, from Brickfields to Redhouse, the estuary widens up to 350 m and the steep banks flatten out. In this area, the relief changes from low to high, with finer and more compact sediments (Hanekom, 1980), steep banks and slightly wider and less convoluted channels. The upper reaches of the estuary from Redhouse to Perseverance are narrow (~90 m wide) with steep muddy sandbanks (DWAF, 1999) and winding narrow channels with steep banks. This region of the estuary is also characterised by coarse sand mixed with various amounts of silt (Jooste, 2003). The tidal limit of the Swartkops Estuary is positioned approximately 16 km from the mouth in the area known as Perseverance.

(5) There have been no records of hypoxic or anoxic events in the lower reaches of the Swartkops Estuary suggesting that the lower reaches are well mixed.

Artificial disturbances to the geomorphology and flow to the estuary include a series of bridges constructed for road and rail use. Although these features restrict the flow to a certain extent, they seem to have minimal tidal effects (DWAF, 1999). The estuary is home to six different plant community types, namely supra- and intertidal salt marsh, submerged macrophytes, reeds, sedges, phytoplankton and benthic microalgae. The only plant community types not found in the estuary include mangroves and swamp forest (Colloty *et al.,* 2000). In 2000, Colloty *et al.* (2000) estimated that the supratidal salt marsh once covered an area of 40 ha before the rise in industrial and residential developments occurred, however, presently only 5 ha remains. The intertidal salt marsh has diminished at a slower rate, from 215 to 165 ha.

The Markman and Motherwell canals and the Chatty River are regarded as three entry points of diffuse pollution of their respective catchment areas, especially for nitrate and ammonia (Watling, 1982; MacKay, 1993; Berry and Robertson, 1996). The Motherwell and Markman stormwater canals drain residential and industrial township areas respectively, and the Chatty River drains an area where informal settlements have been established. All three sources are located in the lower and middle reaches of the Swartkops Estuary.

3.2. **Hydrodynamics, circulation and climate**

The Swartkops River catchment falls in a transitional region between the summer rainfall of the KwaZulu-Natal and Transkei coasts and the winter rainfall of the Western Cape with summer months being the driest (Klages *et al*., 2011), and the highest rainfall usually occurring in June and October. The mean annual runoff is about 84.2 x 10^6 m^3 (Reddering and Esterhuysen, 1981) and the mean annual precipitation (MAP) is roughly 636 mm, with a range of 500 – 1000 mm (Baird, 2001). The monthly average is approximately 55 mm but measurements as high as 200 mm per month have been recorded. Typically, the maximum rainfall occurs at the headwaters of the Swartkops River system and decreases towards to the coast (DWAF, 1999). Occasionally, a single thunderstorm can contribute towards one third of the annual rainfall of the catchment (Haigh, 2002). The flow pattern in the catchment is characterised by low baseflows, with minor floods of 40 to 80 x 10^6 m^3 (DWAF, 1999). The most severe floods (120 to 160 x 10^6 m^3) ever to be recorded occurred in 1879, 1912, 1914, 1971 (when an overflow of 425 m^3 per second was recorded at the Groendal Dam spillway; HKS, 1974) and 1979 (Eastern Province Herald, 1979). A flood event with a recurrence probability of 1 in 100 years occurred in 1981 (Ninham Shand, 1994). Five causeways span the Swartkops River below the Groendal Dam and act as weirs that impede freshwater flow. Apart from a retaining wall at a bridge near Despatch (which also acts as a weir) the other bridges on the Swartkops River do not impede flow. The Wylde Bridge and railway bridge at Swartkops Village obstruct floodwaters but do not appear to significantly affect normal tidal flows, although a region of Tippers Creek has been blocked. At Redhouse, saltpans hold back floodwaters with localized erosion resulting in downstream sand deposition. The Settlers Bridge (N2 Bridge) confines flow to the northern bank at the estuary mouth and the southern causeway has impeded the natural migration tendencies of the main channel (Fromme, 1988). According to Baird *et al.* (1986) the only major obstruction is the Groendal Dam, however, due to discharges from wastewater treatment works, the contribution of flow from the Kelvin Jones, KwaNobuhle and Despatch treatment plants, ensures that there is a nett gain in flow in the lower reaches of the Swartkops River, although this results in a higher than natural flow, especially under low flow conditions (DWAF, 1999). Alteration of flow within the Swartkops catchment area has been associated with four key issues (DWAF, 1999), namely:

1. Agricultural developments, dams and alien vegetation in the surrounding areas of the Elands River have resulted in alteration of flow in the Elands sub-catchment.
2. Water abstraction in the KwaZunga, Elands and Swartkops rivers have reduced the flow.
3. Construction of dams and weirs have obstructed/altered flow throughout the catchment.
4. Construction of dams in the Elands and KwaZunga river catchments have prevented surface runoff from reaching the river system.
5. Wastewater treatment works in the lower reaches of the Swartkops River have resulted in high flow due to the discharge of effluent.

The Swartkops Estuary is characterised by its predominately saline waters, which would imply that it is marine-dominated and that high salinity results from direct influence of marine intrustion. However, this is infact not the case, but instead the higher salinity is due to relatively low freshwater inputs and a gradual dispersion of salt in the upper reaches (MacKay, 1994). Vertical and longitudinal salinity gradients are very small under dry-weather conditions, allowing vertical mixing of nutrients or bacteria to occur. In contrast, longitudinal mixing and dispersion in the upper reaches is restricted by a bar-built formation upstream of the Wylde Bridge. Residence times of between 10 and 14 days have been estimated for the region upstream of Bar None and localised trapping of water may also occur in the estuary. These include Tippers Creek, the channels between the Wylde Bridge and Brickfields and the region between Bar None and Perseverance. It has been noted that pollutants discharged into these regions of the estuary are likely to remain there for extended periods (MacKay, 1994).

The climate is generally warm and temperate with large fluctuations in temperature occurring on a daily and seasonal basis. The mean daily maximum temperature in the low lying areas is approximately 32 °C in January and 18 °C in July, with maximum temperatures of 45 °C and 31 °C respectively being recorded. The mean daily minimum temperature is 15 °C in January and 5 °C in July with maximum temperatures of 5 °C and -3 °C respectively recorded (Haigh, 2002). Mean daily temperatures of about 6 °C and 27 °C have been recorded for July and January respectively.

3.3. **Land use and water use**

The Swartkops River and estuary is impacted by recreational, industrial and residential land uses and water users of varying degrees due to its proximity to the heavily urbanised and industrialised regions of the catchment area. Land use in the upper catchment is predominantly pristine and forested. Although the Swartkops River flows through natural and agricultural areas for most of its length, significant portions of the lower catchment are highly urbanized and industrialised, with major urban areas including Despatch, Uitenhage, Perseverance and KwaNobuhle. The Swartkops catchment area was first populated following the eastward migration of farmers from the Western Cape and by 1776, a number of farms were already established. It was not until 1859, when the first bridge across the Swartkops River was built. In the lower reaches of the catchment, excluding the estuary, domestic use, irrigation, livestock watering and industrial use of surface water occur on a localised level. The Swartkops River is in a highly degraded state due to severe water quality problems (although not unique to developing countries), alien vegetation and fish, and physical manipulation of the channel as well as increased low flows (Enviro-Fish Africa, 2009). The water quality status of the system is typified by localised impacts resulting from industrial activities and both urban and agricultural runoff which is reflected in the absence of sensitive fish and macroinvertebrate species, especially in the lower Swartkops catchment areas, including the Brak and Chatty rivers (DWAF, 1999). Waters in the lower regions of the Elands, Brak, Chatty and Swartkops rivers are generally unsuitable for domestic or broad agricultural use (Haigh, 2002). The main industrial areas within the lower catchment area include:

- the Uitenhage Riverside industrial area (Riverside, Cape Road and Alexander Park),
- the Perseverance industrial area,
- an industrial area in KwaZakele,
- a small industrial area in Despatch, and
- the Deal Party and Markman industrial areas on the boundaries of the catchment. Most industries located in these areas discharge their wastewater to the Fishwater Flats WWTW, to be treated and discharged to the Papenkuils River located south of the estuary. Depending on the tide and wind and wave conditions, effluent discharged from the Paapenkuils River into the surf zone can extend to the mouth of the estuary.

Three residential hubs are located adjacent to the estuary, namely (1) Swartkops Village, Redhouse and Amsterdamhoek/Bluewater Bay, (2) the townships of KwaZakele and Motherwell, and (3) ten major industrial areas are located within close proximity to the estuary. Three important pollution point sources into the Swartkops Estuary are the Motherwell and Markman canals, and the Chatty River, all of which enter the estuary in the middle reaches. Although the Kelvin Jones (in Uitenhage), Despatch and KwaNobuhle WWTW are located above the tidal limit (see Chapter 4: Figure 2), their impacts are seen further downstream with increased levels of nutrients and faecal bacteria. Numerous informal settlements on the banks of the Chatty River, such as the formal high-density townships of Zwide, New Brighton, Missionvale, and Veeplaas contribute substantially to the faecal bacteria present in the river when adequate services are not provided. Stormwater runoffs from these areas are thus a major concern and require improved stormwater drainage and/or sanitation systems. With residence times possibly longer than 14 days and inefficient mixing, the estuary has been shown to be susceptible to degradation as a response to input of pollutants in the middle and upper reaches (Lord and MacKay, 1993). The section below describes the water users of the Swartkops catchment and how they contribute to the physical, chemical, biological and/or aesthetic properties of the river system.

The use of surface water directly from the Swartkops catchment is limited to the inhabitants of the informal developments near the river banks. The surface water quality is however not suitable for domestic use, but water from the Elands (upper), KwaZunga, Sand/Bulk and Brak (upper) rivers may be fit for long term water use with mild health effects. The only recognized water use in the estuary is recreation, with activities mainly including angling and boating, while activities such as swimming, boating, fishing, and hiking take place in the upper reaches of the estuary and in the Swartkops and KwaZunga rivers. The health risk to recreational use is high in the Swartkops River, especially at the Nivens Bridge area and Bullmer Drift, where faecal counts exceed requirements for recreational use and occur as a result of urban runoff and stormwater (DWAF, 1999; Enviro-Fish Africa, 2011). Contamination of the estuarine water by faecal bacteria is also high and originates from urban runoff (from Chatty River), polluted stormwater (from the Motherwell canal) and sewage discharges from wastewater treatment works upstream above the tidal limit. These facilities are associated with health risks, such as gastrointestinal illnesses and are thus considered a serious health risk to recreational users.

Industrial activities are a major land-use within the immediate estuarine area and include saltpans, Fishwater Flats WWTW (see Chapter 4:), sand/clay mining, brickworks, a motor industry, wool industry, tanneries, extractive/beneficiation processes, and railway yards and depots, with only limited agriculture taking place (Scharler and Baird, 2003; Enviro-Fish Africa, 2009), however industrial water use of surface water is considered to be minimal. DWA once issued permits to Gubb & Ingss (wool processing factor), and Perseverance Wool Pullery to abstract water from the Swartkops River, although this no longer occurs. Most industries within the catchment are supplied with municipal water and/or treated wastewater from wastewater treatment works (see Appendix B: Table 28). The industrial area in Uitenhage has the greatest potential to influence water quality in the catchment. Although nearly all industries discharge their wastewater into sewers to be treated by Kelvin Jones WWTW, salts pass through the plant and are discharged to the river (Haigh, 2002). Additionally, the combination of electrical faults and poorly maintained sewer pump stations and wastewater treatment facilities is often responsible for the direct discharge of untreated or partially treated industrial and domestic wastewater into the aquatic system. Industrial activities, especially at the tannery and wool processing plants, are also associated with increased concentration of salts, some trace metals and phosphates and have a high potential to impact the quality of surface and ground water through infiltration and surface runoff. The tanning industry converts almost all water used into a wastewater with a high pH, total dissolved salts and chromium. This wastewater is difficult and expensive to treat, and therefore water is discharged to open and evaporation ponds. MacKay (1994) found that seepage from these ponds had a significant impact on water quality downstream. Two wool processing plants, namely, Gubb & Inggs and Cape of Good Hope, and the wool-pulling plant at Perseverance are located within the catchment. Much of the wastewater is difficult to treat and thus evaporation ponds are used. The wastewater from the wool-washing process is high in phosphates, chlorides, potassium, sodium, total dissolved solids, and carbonate, while wastewater from the scouring process has very high organic and inorganic loads.

Surface waters for irrigation purposes are mainly abstracted from the Sand, Bulk and Groendal dams and supplied by the respective municipalities. All other uses of surface water are non-consumptive (DWAF, 1999). Agriculture occurs mainly in the Kruisriver area, above the confluence of Elands and KwaZunga rivers where mainly potatoes, lettuce, cabbage, carrots, beetroot, sweet potatoes, beans, peppers and brinjals are grown. Cultivated pastures and citrus are also irrigated in this area. Stormwater runoff from irrigated areas is a potential source of nutrients as a result of fertiliser use. However, the impact from agricultural runoff is considered to be minimal due to the low rainfall and limited extent of farming as only 0.7% of the local economy is attributed to the agriculture sector (Klages et al., 2011). Surface water is considered suitable to water of all livestock, however below Bullmer Drift the water quality is not suitable, with some health effects reported in poultry, as well as young and vulnerable animals. The presence of faecal coliforms at Bullmers Drift, Nivans Drift and Perseverance render water unsuitable for livestock watering. Stormwater runoff from farms in the catchment is a potential source of nutrients and microbiological indicators, although the impact is considered to be small (DWAF, 1999).

Numerous inhabitants of the catchment rely on subsistence fishing and bait collection. An estimate of 20 boats on any given weekday and at least twice this number on weekends and holidays can be found in the estuary, with numerous shore-based fishermen seen throughout the week. Additionally, many unlicensed collectors are regularly seen digging up the mudbanks for mudprawns, which ultimately results in habitat loss and affected breeding and recruitment success. The concern is that too much bait is being collected; much of which is not used and discarded. Moreover, the estuary has been ranked as the 11th most important estuary in South Africa in terms of biodiversity with an overall importance score of 92 out of a possible 100. Therefore, the Swartkops Estuary requires a water resource management and biodiversity plant to ensure that its biodiversity status is maintained. Ecosystem pressure is further amplified by the increased demand for bait as the bait that is collected and sold at Swartkops is used extensively in other estuaries such as the Gamtoos, Kromme and Sundays rivers (Enviro-Fish Africa, 2009). On Fridays the bait fishermen are allowed to use spades to collect bait, turning large areas of intertidal sediment over and potentially releasing nutrients and minerals – ammonium and phosphates in particular – to the water column. Additionally, oysters and mussels were also once harvested commercially from an area known as the Blue Hole and also an adjacent area near the estuary mouth, however this is no longer taking place.

Forestry was started in the Elandsberg in 1918. The total area under commercial and natural forest in the Swartkops Catchment is relatively small (4.23 % of the total catchment area) (Haigh, 2002). Although forestry activities within the catchment have a low pollution potential, they can provide a source of organic debris, suspended solids and nutrients to the nearby rivers and cause a reduction in the surface runoff to rivers. The removal of sand and river gravel from river beds in the Swartkops catchment occurs at the following sites; lower KwaZunga River, at Springfontein; in the Swartkops River above Uitenhage, and above and below Perseverance (Haigh, 2002). Quarry mining within the catchment alters the river channel and banks, reducing the habitat integrity and water quality (suspended solids) of those sections of the river. Salt mining also occurs within the floodplain of the Swartkops River at Chatty River, Redhouse and Bar None. Although the Chatty River saltpan is used for feeding and roosting seabirds, it together with the other pans is a source of brine (with a high salt concentration) to the aquifer.

3.4. **Ecosystem goods and services**

A healthy and viable aquatic ecosystem provides numerous functions and ensures moderate, year round flows and degradation of pollutants and pathogens thereby functioning as natural purification system. However, the capacity of a water course to function as a detoxifying system is often exploited through the continual or sporadic discharge of untreated or partially treated wastewater into the water course. For the Swartkops catchment, this is an ongoing concern and one that is associated with economic and recreational losses for the city. For example, in 2010 the Redhouse River Mile swimming event, which takes place on the Swartkops River was relocated to the Sundays River due to water quality concerns.

For many inhabitants of the Swartkops catchment area, health status of the estuary is of particular importance, as the system provides a number of goods and services. Estuaries provide goods and services that generate a range of economic values (Turpie and Clark, 2007; Hosking, 2011). Table 13 provides economic values attributed to the Swartkops Estuary. According to a botanical rating system (area covered by each plant community type, its association with the estuary, its condition and the plant community richness), the Swartkops Estuary is rated 18th overall with a score of 170 (normalised score of 41 out of 100). With a rich and diverse botanical profile, the Swartkops Estuary serves as an important nursery habitat for the ichthyofaunal community (Baird *et al*., 1986; Marais, 1987) and bird species (Martin, 1988). Various regions of the Swartkops catchment are also utilised for cultural and religious purposes. For example, the area adjacent to the Motherwell Canal and 100 m from Swartkops Nature Reserve is utilised by the Zion Christian Church for the baptism of members of its congregation. Similarly, another congregation of the Zion Christian Church uses the area beneath the rail and road bridges on the Old Grahamstown Road for baptisms. The western side of the Swartkops River near Redhouse and the area beneath the Nivens Bridge are also used by traditional healers to perform cleansing ceremonies and to harvest medicinal plants.

3.5. **Legislation and management structures**

Numerous governmental structures required to manage and maintain the water quality and quantity of the Swartkops catchment are in place (Enviro-Fish Africa, 2009). However, it is the implementation of these laws that will protect and conserve the Swartkops River and estuary and promote the goods and ecosystem services associated with it. A Co-management Forum has been established to oversee issues of mutual interest, implement procedures that ensure the preservation of the natural resources, identify risks and non-compliance, and report these to relevant authorities. Management groups associated with the forum include:

- Nelson Mandela Metropolitan University (NMMU) – Developmental Studies
- Subsistence bait collectors and fishermen
- Cape Action for People and the Environment (C.A.P.E.)
- Zwartkops Trust, now referred to as the Zwartkops Conservancy: (The Trust has undertaken the task of cleaning the Swartkops River and Motherwell Canal through joint initiatives with corporations such as South African Breweries)

Table 13: Economic values attributed to the Swartkops Estuary.

Type of value	**Value provide by the estuary**
Subsistence	Ranked 1st amongst temperate systems with a value of R808 953 per annum
Property	Ranked 19th amongst temperate systems in terms of property value related to estuaries with a value of R155 million
Tourism	Ranked 7th amongst temperate systems in terms of tourism value attributed to estuaries with a value of R50 million per year.
Nursery for fish	Ranked 5th amongst temperate systems with a value of R32.8 million per annum.
Existence	Does not rank amongst the top 40 temperate estuaries

Note: Existence value is the value of simply knowing that an estuary and its biodiversity are protected.
Source: Turpie and Clark (2007)

Chapter 4: Spatial and temporal variability in water quality

4.1. Introduction

On a global scale, considerable attention has been given to the assessment of water quality for spatial and temporal trends, as only recently has there been enough data available to make such analyses feasible (Sonier *et al*., 2006; Sprague *et al*., 2008; Ballantine and Davies-Colley, 2010). With the appropriate use of statistical tests, long-term data can serve as a warning, alerting environmental managers of potential of increasing surface water related problems, such as (1) changes in flow characteristics due to runoff alteration, wastewater discharge volumes, abstraction or infrastructure developments and (2) water chemistry changes due to polluted effluent from wastewater treatment plants, stormwater canals and industrial areas (Russell *et al*., 2011). Such data can also confirm the effectiveness of newly enforced water resource management strategies and restoration initiatives. Long-term water quality data should inevitably bring to light socio-economic and infrastructural aspects of the catchment area that influence the water quality of the water body. In this manner, factors that influence the water quality of the aquatic system can be understood and managed accordingly, the ecosystem health can be maintained or improved and as a result, ecosystem goods and services provided by the estuary can be sustained or enhanced. Long-term data sets can also assist in determining whether a newly enforced water resource management action plan and/or anthropogenic impacts have adversely or favourably influenced the water quality and ecosystem integrity of a system (Carr and Neary, 2008). In essence, long-term monitoring is meaningless if the following pre-requisites are not adhered to: (1) consistent data collection (i.e. set temporal and spatial scales and record of tidal stages), (2) correlation of water quality variables with freshwater inflow measured at each monitoring site, (3) safe and secure storage of data and (4) standardised laboratory methods.

Over the last three decades there has been a dramatic increase in low-cost housing, informal settlements, industrial and agricultural developments along the banks of Eastern Cape estuaries and within their catchment areas. This has led to a deterioration in water quality as they increasingly become the repositories of human, domestic, industrial and agricultural waste (Binning and Baird, 2001; Scharler and Baird, 2003). The Department of Water Affairs (DWA) together with the Nelson Mandela Bay Municipality and other research groups such as Rhodes University, the Nelson Mandela Metropolitan University (NMMU), the Council for Scientific and Industrial Research (CSIR) and environmental consulting companies have been monitoring the water quality of the Swartkops Estuary since the 1970s.

Efficient monitoring and management of anthropogenic impacts requires reliable and consistent historical data to allow for comparisons with current water quality data. Moreover, management actions for improving the health of the Swartkops Estuary and achieving the recommended ecological status are required. The Estuary Management Plan (EMP) for the Swartkops Estuary has identified eight key areas for management actions for the Swartkops Estuary, one of which concerns water quantity and quality (Enviro-Fish, 2011).

To improve the health of the Swartkops Estuary from a “D” to a “C” would require an improvement in the water quality and volume of ‘treated’ sewage wastewater and better control of stormwater input. In order achieve this a comprehensive reserve study is needed to the Swartkops Estuary. Moreover, Water Resource Quality Objectives neeed to be identified based on reliable flow data. This study has contributed to this assessment by providing detailed water quality data for 2012 and 2013, including an overview of historical quality quality data.

4.2. **Materials and Methods**

Water quality parameters were measured on five occasions between September 2012 and August 2013 to provide an assessment of the health of the estuary. Thereafter, the current status of the water qualiy was compared with historical water quality to determine the extent of temporal and spatial variability in water quality patterns.

4.2.1. Sampling sites

Although the Department of Water Affairs has been collecting long-term water quality data since the 1970s, data collection has not been continuous. Consequently only a subset of the Department of Water Affairs water quality monitoring sites was selected for long-term data analysis (obtained from: http://www.dwa.gov.za/iwqs/wms/data/M_reg_WMS_nobor.htm). Nevertheless, historical water quality data from DWA supported the bulk of obtainable historical data and these data were thus compared against data from other sources. In total, 16 monitoring sites were selected for spatial and temporal analyses of historical water quality data. Estuarine and freshwater monitoring sites of the Swartkops catchment area are indicated in Figure 1 and respectively. Additionally, historical water quality data for the three wastewater treatment works, namely Kelvin Jones, Despatch and KwaNobuhle were obtained upon request from the Department of Water Affairs (DWA, 2013).

Water samples were collected at three point sources of entry into the estuary, namely Chatty River, Markman Canal and Motherwell Canal and then assessed for their impact on seven sites within the estuary, namely Settlers Bridge, Tippers Creek, Swartkops Village, Brickfields, Redhouse Yacht Club, Bar None and Perseverance. Tippers Creek is not located within the main channel but is regarded as a popular bathing area and hence was included in this study. All estuarine and freshwater study sites, including specific sources of pollution to the estuary, namely wastewater treatment works and stormwater canals, as well as the Chatty and Elands rivers are listed in Table 14**.**

Estuarine sites

The Settlers Bridge (SB; N2 Highway) (Figure 3: Plate A) was built to completion in 1974 (Jacot Guillard, 1974) and is situated approximately 100 m upstream of the estuary mouth (Figure 3: Plate B). The bridge confines flow to the northern bank at the mouth and the southern causeway has impeded the natural migration tendencies of the main channel (Fromme, 1988). This area of the estuary is bordered by the residential area of Amsterdamhoek and is regarded as a popular area for fishing and boating activities. The river mouth is permanently open to Algoa Bay.

Tippers Creek (TC) is adjacent to the houses of Amsterdamhoek, and is known to be a popular and safe area for bathing (Figure 3: Plate C). The Creek was formerly a tidal channel, but with the erection of the Wylde Bridge, its upper end was cut off by the Northern embankment (MacKay, 1994). As a result, this area is sometimes poorly flushed resulting in the accumulation of fine muddy sediments (Reddering and Esterhuysen, 1981). Two stormwater drains and one conservancy tank is located in this area and following heavy rains, results in greywater and domestic sewage flowing into the estuary. This has led to regular reportings of poor water quality and the erection of a public notice board that states “swimming is not recommended due to possible water pollution” (Figure 3: Plate D).

The residential area of Swartkops Village (SKV; Figure 3: Plate E) is situated downstream of the Swartkops Village bridge (Figure 3: Plate F), is heavily utilised by cars and trucks, and is close to the industrial areas of Port Elizabeth. Due to the close proximity of the bridge to the area, the water is sometimes visibly polluted with oil and coal dust and has drainage holes that empty directly into the estuary (Hilmer and Bate, 1987). Additionally, a wastewater pumping station is located within close proximity to this sampling site which malfunctions from time to time. This will be environmental concern if wastewater leaks from the station and enters the estuary.

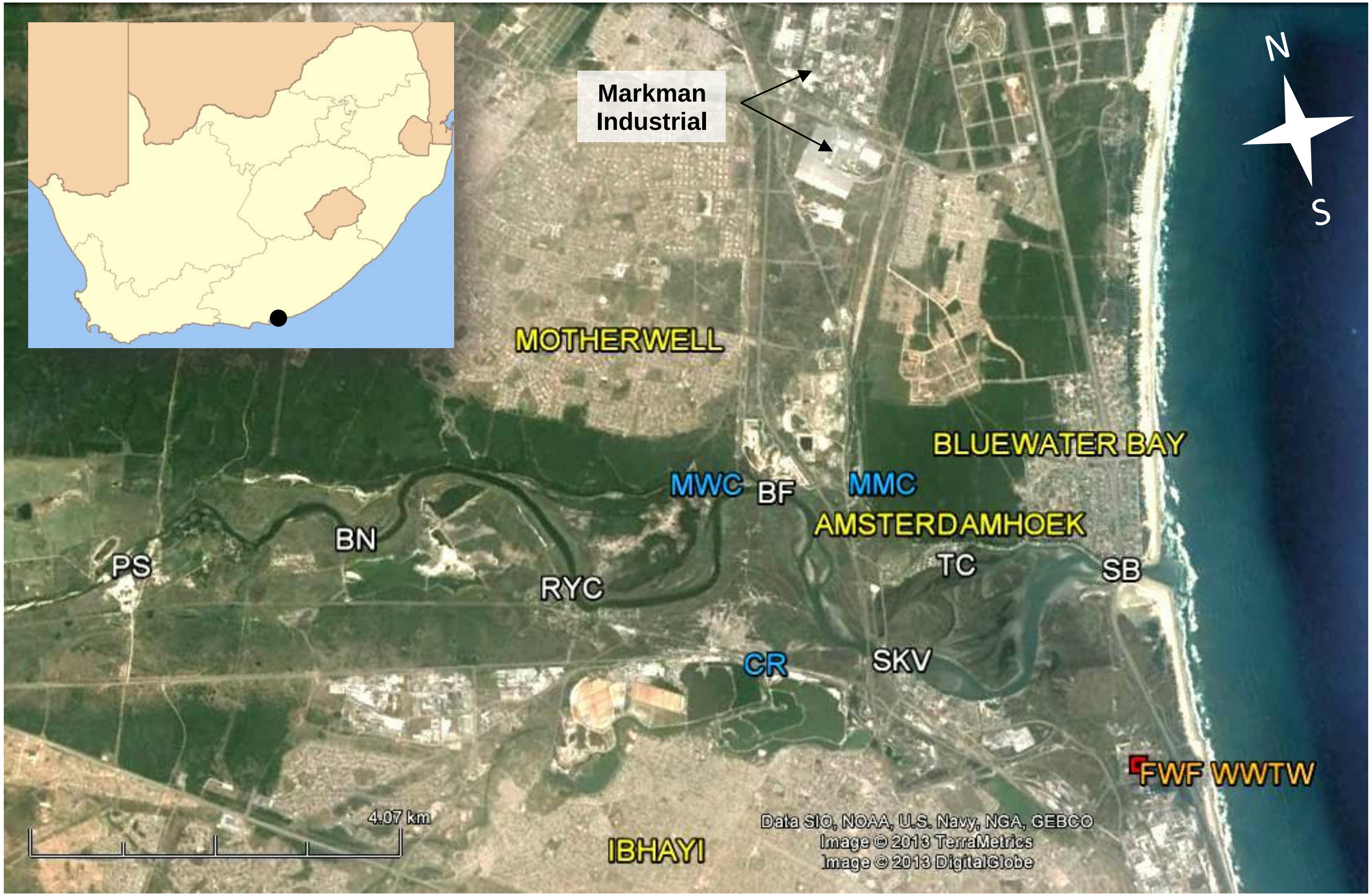


Figure 1: Locality map of water quality monitoring sites within the estuary. (The inset shows the relative position of the Swartkops Estuary (●) along the South African coastline).

Estuarine sites: Settlers Bridge (SB), Tippers Creek (TC), Swartkops Village (SKV), Brickfields (BF), Redhouse Yacht Club (RYC), Bar None (BN) and Perseverance (PS)

Points of entry into the estuary: Markman Canal (MMC), Chatty River (CR) and Motherwell Canal (MWC)

Wastewater Treatment Works (WWTW): Fishwater Flats (FWF)

Figure 2: Locality map of water quality monitoring sites in the freshwater river reaches showing wastewater treatment works (WWTW) and major residential and industrial areas. (D = Despatch, KN = KwaNobuhle, KJ = Kelvin Jones).

River sites: Perseverance Bridge (PSB), Van Schalkwyk Bridge (VSB), Frans Claasen Bridge (FCB), Nivens Bridge (NB), Elands River (ER) and Groendal Dam (GD)
Wastewater Treatment Works (WWTW): Despatch (D), Kelvin Jones (KJ) and KwaNobuhle (KN)

Table 14: Estuarine and freshwater monitoring sites.

	Sampling sites	DWA WMS code	Latitude (°S)	Longitude (°E)	Distance from mouth (~km)
SWARTKOPS MONITORING SITES					
Estuary	Settlers Bridge (SB)	M10 183968	33.8619	25.6271	0.4
	Tippers Creek (TC)	N/A (non DWA site)	33.8536	25.6155	2.2
	Swartkops Village (SKV)	M10 184019	33.8606	25.6005	4.0
	Brickfields (BF)	M10 102373	33.8400	25.5989	6.6
	Redhouse Yacht Club (RYC)	M10 183971	33.8366	25.5708	10.0
	Bar None (BN)	M10 183970	33.8214	25.5509	13.6
	Perseverance (PS)	M10 1000008483	33.8111	25.5312	16.4
Points of entry into the estuary	Chatty River (CR)	M10 102377	33.8536	25.5856	4.8
	Markman Canal (MMC)	N/A (non DWA site)	33.8433	25.6048	6.1
	Motherwell Canal (MWC)	M10 1000008475	33.8366	25.5950	7.0
River	Perseverance Bridge (PSB)	M10 1000002479	33.8121	25.5214	17.2
	Van Schalkwyk Bridge (VSB)	M10 191547	33.7964	25.4527	25.1
	Frans Claasen Bridge (FCB)	M10 102369	33.7892	25.4269	28.0
	Nivens Bridge (NB)	M10 102370	33.7711	25.3867	32.8
	Elands River (ER)	M10 102368	33.7675	25.3295	40.9
	Groendal Dam (GD)	M10 102378	33.6900	25.2667	53.7
WWTW	Despatch WWTW	M10 1000002484	33.8006	25.4969	19.8
	Kelvin Jones WWTW	M10 1000002489	33.7831	25.4261	27.5
	KwaNobuhle WWTW	M10 1000002494	33.8058	25.3994	29.7

Note: WWTW = Wastewater treatment works

The Chatty River (CR) is the fourth longest river within the Swrtkops catchment with its entire length (ca. 22 km in length) (Baird *et al*., 1986) contained within the boundaries of the Nelson Mandela Bay Municipality. The river enters the Swartkops Estuary just upstream of the Wylde Bridge and flows through the highly populated residential areas of Zwide, Veeplaas, New Brighton, Bethelsdorp and Missionvale from where the river receives polluted stormwater runoff. Originally the lower reaches of the Chatty River were poorly-defined and used to connect with the estuary through a series of shallow channels that meandered into the floodplain (Figure 3: Plate G and H). The Chatty River (now restricted to a narrow channel as a result of the establishment of saltpans at Veeplaas) is the largest tributary flowing directly into the estuary upstream of the Swartkops Village and is considered to be one of the main point sources of pollution entering the estuary (Enviro-Fish Africa, 2009). Furthermore, the Chatty River is one of three main point sources of pollution into the Swartkops Estuary (the other two being the Markman and Motherwell canals) (Baird *et al*., 1986; MacKay, 1994; Scharler *et al*., 1997) and is considered to be ecologically damaged, possibly beyond any significant rehabilitation (Klages *et al*., 2011).

Figure 3: Sampling site within the middle reaches of the Swartkops Estuary.

The major effect of the informal settlements derives from the lack of modern sanitation facilities and inadequate stormwater drainage systems that results in the discharge of raw sewage and litter into the river. Generally, where adequate sanitation services are not provided, night-soil buckets are occasionally emptied into the Swartkops system (MacKay, 1994; see Appendix A: Articles 4, 6 and 7), however this activity is on the decline with the recent installation of flush toilets (Klages *et al*., 2011). In addition to this, nitrogen may also originate from the saltpans, due to faecal matter produced by the large number of birds it occupies. The river is thus considered to be a major health risk by all local residents (Klages *et al*., 2011).

The Chatty River is not usually a perennial stream; in the absence of rainfall there is no natural flow. However, during periods of high rainfall, freshwater inflow from the Chatty River helps to lower salinity in the surface layers at the confluence with the Swartkops Estuary (MacKay, 1993). It is speculated that the water quality in the upper reaches of the river is more polluted than waters of the middle and lower reaches due to extensive salt marshes that act as a sink further downstream. According to Scharler *et al*. (1997) the Chatty River is an important source of nitrate. Concentrations are generally lower than in the Motherwell Canal (Berry and Robertson, 1996), yet tend to increase during high river flow (MacKay, 1993). When compared to the Swartkops River at Perseverance, MacKay (1993) further confirmed that the Chatty River (despite only covering one tenth of the area of the total catchment) is a major source of nitrogen in dry weather conditions, contributing between three and ten times as much as the Swartkops River. MacKay (1993) estimated that approximately 90% of the nitrogen was in the form of ammonium and is thus readily available to phytoplankton.

The Markman Canal (MC) enters the Swartkops Estuary approximately 6.1 km from the estuary mouth. An idustrial area is located on its northern bank where numerous stormwater drains discharge effluent into the canal (Figure 4: Plate I). The canal passes through a small peri-urban village (Figure 4: Plate J) before entering the Swartkops Estuary to the right of a road bridge (Figure 4: Plate K). The quality of the water in the canal is impacted by improper functioning of two sewege pump stations, namely the Aloes Pump Station (Figure 4: Plate L) and the Studebaker Pump Station (Figure 4: Plate M and see Appendix A: Articles 3, 5 and 7). Occasionally, these pump stations malfunction and divert domestic sewage and industrial wastewater into the Markman Canal (pers. comm.; also see Appendix A: Articles 1, 3 and 7). Despite excessive pollution loading, vegetation (mainly *Phragmites australis*) in the canal bed and the long travel time to the estuary, are considered to be relatively effective in removing pollutants (i.e. trace metals) (MacKay, 1994). This is in contrast with the Motherwell Canal (see below). However, it is suspected that this may have changed in recent years due to expansion of the industrial area.

Brickfields (BF) is located 6.6 km from the Swartkops River mouth and positioned between the Markman and Motherwell canals. Algoa Brick (Pty) Ltd is located adjacent to this site (Figure 4: Plate N) and may pose as a site of land-based contamination. Recent water quality tests reveal that *E. coli* counts regularly exceed 10 000 per 100 ml at Motherwell Canal and Brickfields (Enviro-Fish Africa, 2009). This area of the estuary is usually well mixed due to a sandbar that inhibits the downstream movement of a stratified water column (MacKay, 1993). During periods of high rainfall, freshwater inflows from the Motherwell and Markman canals in the vicinity of Brickfields helps to lower salinity in the surface layers at their confluence with the Swartkops Estuary (MacKay, 1993). Both increases (Scharler *et al*., 1997; Marais, 1984) and decreases (McLachlan and Grindley, 1974; Daniel, 1994) in turbidity towards the upper reaches have been recorded, with the highest turbidity often recorded in the middle reaches. According to Scharler *et al.* (1997) polluted water from the Motherwell and/or Markman Canal has an impact on the quality of the water in the middle reaches of the estuary.

The Motherwell Township (Figure 4: Plates O and P) is located along the length of the ~4.2 km long Motherwell Canal (MWC) that is serviced by a network of approximately 14 stormwater drains that deposit litter, debris and sewage into the canal. During moderate and high flows, litter and human waste is carried to the estuary via the canal. Several pollutant traps (Figure 7: Plate Q) have been put in place; however these are often blocked with litter and require regular maintenance, which is a rare occurrence. In dry weather, steady flow of highly polluted water is observed in the canal. Water quality tests reveal that *E. coli* counts regularly exceed 10 000 counts 100 ml^{-1} (Lord and Thompson, 1988; MacKay, 1993). MacKay (1993) noted that faecal coliform bacteria recorded in the estuary adjacent to the canal regularly exceeds 10 000 counts 100 ml^{-1}; sometimes reaching 10^8 counts 100 ml^{-1} in the canal itself. In addition to this, the canal is also a source of nutrients, especially nitrogen (MacKay, 1993; Snow, 2008). It is known that domestic effluent enters the stormwater canal following blockages or leaks of the sewerage system (see Appendix A: Article 1), which results in elevated nitrogen levels and faecal pollution detected in the canal.

The 19th of February 2010 marked the opening of what is known to be the first artificial wetland system of its kind in South Africa (Figure 5 and Figure 6) (SRK Consulting (Pty) Ltd, 2010). The wetland was designed to divert and filter 20% of polluted urban runoff from the stormwater canal through a series of ponds prior to diversion back into the estuary. To assess the efficacy of the artificial wetland system, water quality monitoring commenced in February 2010 and continued for a period of 11 months. Following weekly water quality analyses, the study showed a marked reduction in faecal bacteria counts, including *E. coli*, faecal coliforms and total coliforms, being discharged from the wetland system in relation to counts detected in the intake water.

Figure 4: Sampling sites within the middle reaches of the Swartkops Estuary.

Figure 5: An aerial photograph of the Motherwell artificial wetland taken after completion in 2009. The wetland includes a primary reinforced concrete containment cell (right) and two secondary reedbed cells for treatment (SRK Consulting (Pty) Ltd, 2010).

Figure 6: The Motherwell artificial wetland in 2013.

Although the artificial wetland has resulted in a marked reduction in faecal bacteria following the passage of stormwater through the wetland, faecal bacteria levels at the junction of the Motherwell Canal and the Swartkops Estuary still exceed acceptable levels for stormwater and recreation respectively. Since this is an area of the estuary where subsistence fishing occurs and where Xhosa men can be found practicing their traditional rituals and their entry into manhood (Figure 7: Plate S), good water quality of the canal and the adjacent estuary is required.

The Redhouse Yacht Club (RYC; Figure 7: Plate T) is situated in the village of Redhouse where numerous recreational activities take place, such as fishing, sailing, rowing and swimming (Enviro-Fish Africa, 2009; Plate U). The Redhouse River Mile swim event was held at the Redhouse Yacht Club until 2010, however due to water quality concerns, the event relocated to the Sundays River (see Appendix A: Article 4). Poor water quality emanates from partially treated sewage flowing into the upper reaches of the estuary and from the Motherwell Canal.

Bar None (BN; Figure 7: Plate V) is often used for recreational purposes such as bathing, fishing and canoeing, as well as grazing by cattle. Salt concentration pans located along the western bank of the estuary have been linked to elevated salinity levels in the vicinity. When the southern pan reaches maximum capacity, saline water is routed to the estuary via a narrow channel (MacKay, 1994).

Perseverance (PS; Figure 7: Plate W) is located at the tidal limit of the estuary and is characterised by shallow waters. Further upstream of this site on the river bank was once a wool pullery that used to discharge wastewater to aeration ponds nearby. According to Rump (pers. comm.), Deranco Blocks has now taken residency at this site, following the closure of the wool pullery. The Perseverance Abattoir is also located here and another abattoir, Karoor Osche located further upstream near the river bank. Furthermore, a gravel quarry is located within close proximity of the site and infestation by water hyacinth (*Eichhornia crassispes*) is frequently observed due to nutrient enrichment of the water as a result of wastewater discharge from wastewater treatment works located upstream (Haigh, 2002).

Figure 7: Sampling sites within the middle and upper reaches of the Swartkops Estuary and the Swartkops River.

Wastewater treatment works (WWTW)

The reuse of treated sewage within the catchment is approximately 4.1 Ml day^{-1} from Fishwater Flats WWTW and 0.8 Ml day^{-1} from Kelvin Jones WWTW, which thereafter is used for industrial purposes and for irrigation of public open spaces, communal gardens and sports fields (DWAF, 1999). Apart from Kelvin Jones WWTW in Uitenhage, two other WWTW are located in the lower catchment area of the Swartkops system, namely, KwaNobuhle and Despatch. Approximately 56% of the treated sewage from the Kelvin Jones, KwaNobuhle and Despatch WWTW is discharged into the Swartkops River System, of which 65% is from Kelvin Jones WWTW, 27% is from KwaNobuhle WWTW and 8% from Despatch WWTW. The treated wastewater discharged from Kelvin Jones, KwaNobuhle and Despatch WWTW contributes approximately 50% towards the flow in the Swartkops River estuary (DWAF, 1999).

As with most developing counties, WWTW within the Swartkops catchment are a major source of faecal bacteria and nutrients. Under eutrophic conditions high nutrient levels have in the past promoted the prolific growth of water hyacinth (*Eichhornia crassipes*) in the Swartkops River between Uitenhage and Perseverance (DWAF, 1999). Phosphate concentrations increase considerably below the Nic Claasen Bridge in Uitenhage and remain at high concentrations, even in the estuary. These raised levels are attributed mainly to the sewage discharges at KwaNobuhle, Uitenhage and Despatch. The Uitenhage and Despatch WWTW release approximately 6 822 500 m^3 and 835 000 m^3 per annum of treated wastewater into the Swartkops River respectively. The Brak River receives treated effluent from the KwaNobuhle WWTW of about 2 847 000 m^3 per annum and discharges it into the Swartkops River below Uitenhage (DWAF, 1999). The results of wastewater compliance assessments of the three WWTW are shown in Table 15 to Table 17. Overall, these wastewater compliance assessments indicated that the three WWTW performing poorly in terms of microbiological and chemical compliance criteria; where the highest risk area for all WWTW was stated as "poor effluent compliance" (DWA, 2012). Most concerning is that the flow amount exceeding the capacity of the respective WWTW ranges from 75 to 96 Ml d^{-1}, while the design capacities amoungst the three WWTW only range from 4.6 to 22 Ml d^{-1} (Table 16).

The compliance assessment of 2009 noted that 67% (84 out of 125) WWTW in the Eastern Cape do not measure and record their daily inflow, have no flow meters in place, are broken, are not repaired, have no instrumentation technician and/or operating engineers and don't understand the importance of measuring the inflow or have the ability to take the necessary readings (DWA, 2009). However, it was noted in the Green Drop assessment report of 2012 that the Nelson Mandela Bay Municipality was (at the time of the report) installing influent and effluent flow meters and chlorine disinfection.

Table 15: Compliance ratings of wastewater treatment works located within the Swartkops catchment.

Name of WWTW	Bacteriological Quality (health)		Physical Quality (aesthetic)			Chemical Quality		
	Escherichia coli	**Faecal coliform**	**pH**	**EC**	**SS**	**TOxN**	**NH_3**	**SRP**
Kelvin Jones	NC	NI	C	C	NC	NI	C	NI
Despatch	NC	NM	C	NC	C~ (LT)	NM	C~ (LT)	NM
KwaNobuhle	NC (LT)	NI	C	C	C	NI	NC (LT)	NI

Note: WWTW = Wastewater Treatment Works, C = Compliance, NC = Non-Compliance, NI = No Information, NM = No monitoring done, (LT) = Most Recent Trend. 'NI' and 'NM' equals a situation of Non-Compliance for purposes of this assessment. This is based on the rationale that monitoring and access to effluent quality is a legal (licensed) requirement. Until such information has been obtained and verified, the WWTW cannot be taken to be compliant.

Source: DWA (2009)

Table 16: Wastewater compliance requirements according to Occupational Health and Safety Act 85 of 1998. Assessment of 2009.

	First order risk			Second order risk	Third order risk						
	Design capacity of plant (Ml d^{-1})	**Actual flow amount (Ml d^{-1})**	**Flow amount exceeding/on and below capacity (Ml d^{-1})**	**Number of non-compliance trends for the various parameters (A)**	**Compliance (C)/non-compliance (NC)/to technical skills**	**Flow amount exceeding capacity (Ml d^{-1})**	**Design capacity rating (A)**	**Capacity exceedance rating (B)**	**Effluent failure rating (C)**	**Technical skills Rating (D)**	**Cumulative Risk Rating (AxB+C+D)**
Kelvin Jones	22	16.6	5.4	5	2	75	3	3	5	2	16
Despatch	4.6	3.6	1.1	5	2	77	1	3	5	2	10
KwaNobuhle	7.8	7.5	0.3	5	2	96	2	3	5	2	13

Source: DWA (2009)

Table 17: Green Drop wastewater treatment site inspection results of 2012.

	Kelvin Jones	**Despatch**	**KwaNobuhle**
Technology type	Activated sludge and sludge lagoons	Activated sludge and BNR and sludge drying beds	Activated sludge and BNR and sludge lagoons
Treatment capacity (Ml d^{-1})	24 (large plant)	8.5 (medium plant)	9 (medium plant)
Operational % i.t.o. capacity	70.8%	35.3%	72.2%
Microbiological compliance	16.0%	83.0%	16.0%
Chemical compliance	47.8%	72.8%	58.2%
Annual average effluent quality compliance	32.3%	74.1%	54.3%
Highest risk area	Poor effluent compliance		

Note: 'BNR' = Biological Nitrate Removal
Source: DWA (2012)

A fourth wastewater treatment plant, Fishwater Flats, enters the Papenkuils River approximately 1.5 km from its river mouth at a distance of about 6 km downshore of the Swartkops River mouth. It has previously been noted that nutrient inputs from the Fishwater Flats outfall near the mouth of the Papenkuils River are a possible cause for concern (Watling and Emmerson, 1981). Zinc concentration has been reported as 500 times greater than the highest concentration of 8 $\mu g\ l^{-1}$ found nearby the Swartkops River, whereas lead is more than 50 times higher (Watling and Emmerson, 1981).In another study, Emmerson *et al*. (1983) concluded that concentrations of nutrients (ammonia, nitrate, nitrite and phosphate), metals (aluminium, arsenic, cadmium, chromium, copper, cyanide, fluoride, iron, lead, mercury, selenium and zinc) and *E. coli* were elevated in the vicinity of the Fishwater Flats sewage outfall into the Algoa Bay. However, it was noted that these inputs had minimal overall effects on the immediate coastal environment and rather resulted in biological enrichment. Further investigations are required to determine whether these effects have increased over the years and whether there is a dilution effect on nutrients, trace metals and faecal bacteria upstream of the Papenkuils River mouth and the Fishwater Flats outfall as this may be a source of pollutants which could enter the Swartkops River mouth.

4.2.2. Collation of historical water quality data

Table 18 summarises available data on water quality that has been published by several authors, including water quality data recorded by the Department of Water Affairs and data obtained from a baseline study conducted by SRK Consulting (Pty) Ltd (2011) for the Nelson Mandela Bay Municipality. Authors not referred to in the table include major reviews and reports by McLachlan (1972), Melville-Smith (1978), Hanekom (1980), Hilmer (1984), Baird *et al*. (1986) and Baird *et al*. (1988). These reviews and reports were referred to for historical data on physico-chemical variables, phytoplankton biomass, phytoplankton dominant taxa and spatial and temporal frequencies in the occurrence of macrophytes. Water quality data collection by the Department of Water Affairs has been largely inconsistent with sampling frequencies varying among estuarine and freshwater sites, from almost weekly to annually. These inconsistencies (data gaps) are illustrated in Table 29 to Table 31 in Appendix B. For a complete summary of water quality data, the tables also include data collected through academic studies and this study.

Table 18: Collation of historical data on bacteria, trace metals, nutrients and phytoplankton biomass obtained from several studies.

	Watling and Watling (1982)			Emmerson (1985)			Lord and Thompson (1988)			Scharler *et al* (1997)				DWAF (1995 – 2012)			NMBM: SRK Consulting (Pty) Ltd (2011)		
Parameters	**B**	**M**	**N**	**B**	**M**	**N**	**B**	**M**	**N**	**B**	**M**	**N**	**P**	**B**	**M**	**N**	**B**	**M**	**N**
Settlers Bridge				X		X						X	X	X		X	X	X	X
Tippers Creek		X																	
Swartkops Village		X		X		X	X							X		X	X	X	X
Chatty River														X			X	X	X
Markman Canal		X																	
Brickfields				X		X	X					X	X	X		X	X	X	X
Motherwell Canal		X												X		X	X	X	X
Redhouse Yacht Club		X		X		X	X							X		X	X	X	X
Bar None		X										X	X			X			
Perseverance		X		X		X	X					X	X	X		X	X	X	X
Perseverance Bridge																X			
Van Schalkwyk Bridge																X			
Frans Claasen Bridge																X	X	X	X
Niven Bridge																X	X	X	X
Elands River																X			
Groendal Dam																X			

Note: 'B' = bacteria, 'M' = trace metals, 'N' = nutrients, 'P' = phytoplankton biomass

4.2.3. Sampling analysis during 2012 and 2013

Water samples were analysed for physico-chemical, chemical and biological characteristics. Physico-chemical characteristics included salinity, turbidity, temperature, pH, dissolved oxygen and TSS whereas chemical characteristics included nutrients (ammonium, total oxidised nitrogen, dissolved inorganic nitrogen and dissolved inorganic phosphorus), and trace metals (aluminium, arsenic, cadmium, chromium, copper, cyanide, fluoride, iron, lead, mercury, selenium and zinc). The biological composition of the water included studies on *E. coli*, enterococci, phytoplankton biomass (chlorophyll-*a*) and phytoplankton community composition. Rainfall and river flow data were obtained from the South African Weather Service and the Department of Water Affairs respectively. For sites where the depth exceeded 0.5 m, surface and bottom water samples were collected and for sites with depths less than 0.5 m, only surface water samples were collected. Replicate water samples were collected at all sites and sent to external laboratories (Talbot and Talbot Laboratories and Pathcare) for analyses. It is worth noting that time in the field was restricted due to sample delivering times requested by the external laboratories. For this reason, a preliminary sampling session was undertaken on 18 September 2012 to establish the time required to collect and filter water samples within the study area (Table 19).

Due to times constraints imposed during sampling, water samples for nutrient and phytoplankton studies were filtered upon return to the laboratory. Throughout the sampling period and during transportation to the relevant laboratories, all water samples were kept cool. The study made use of trace metal and faecal bacteria data reported by Talbot and Talbot Laboratories and Pathcare (Table 19).

4.2.4. Hydrological conditions

The Department of Water Affairs measured the Swartkops River flow data as daily mean cumecs ($m^3 s^{-1}$) and the South African Weather Services in Port Elizabeth provided daily rainfall totals. Daily flow rate readings were taken at Nivens Bridge, which is located in the upper reaches of the estuary and upstream of the furthest sampling site (Perseverance) and were auto-corrected for flow at Perseverance, as this is the only gauge in operational use. Only daily rainfall totals recorded at a rain gauge in Uitenhage (33.7140° S, 25.4350° E) were used. Rainfall measurements that were previously taken at the Swartkops Power Station (33.8660° S, 25.6000° E) were halted prior to this study. For this reason, rainfall measurements taken at the Port Elizabeth Airport were used for the years 2012 and 2013, including measurements recorded at Bluewater Bay during the same period for comparison.

4.2.5. Physico-chemical parameters

In situ measurements for pH, temperature (°C), salinity (ppt), and dissolved oxygen (DO, mg l^{-1}) were measured at each site at 0.5 m depth intervals until the bottom was reached, using a 650 MDS YSI multiprobe. Typically, only sub-surface readings were taken at Tippers Creek, Chatty River, Markman Canal, Motherwell Canal and Perseverance as the water depth as these sites were seldom deeper than 0.5 m. Total suspended solids were analysed by collecting 250 ml sub-surface and near-bottom water samples (depending on the depth of the water column) using a weighted pop-bottle. The water samples were filtered through Whatman GF/C filter paper, dried at 75 °C to constant mass and weighed. Upon filtration, the filters were returned to the oven to dry at 75 °C for 24 hours and the total suspended sediment expressed as mg l^{-1}.

Table 19: Sampling and analysis regimes.

	Surface water only	Surface and near bottom water	Analysed by
September 2012 (Preliminary)			
Physico-chemical		0.5 m depth intervals until the bottom	NMMU Botany Department
Nutrients		X	NMMU Botany Department
Faecal bacteria	X		Talbot and Talbot Laboratories (Port Elizabeth and Pietermaritzburg)
Phytoplankton biomass (chlorophyll-*a*)		X	NMMU Botany Department
Phytoplankton taxa		X	NMMU Botany Department
November 2012, February 2013, May 2013, August 2013			
Physico-chemical		0.5 m depth intervals until the bottom	NMMU Botany Department
Total suspended solids		X	NMMU Botany Department
Nutrients		X	NMMU Botany Department
Trace metals	X		Talbot and Talbot Laboratories (Pietermaritzburg), Pathcare (Cape Town)
Faecal bacteria	X		Talbot and Talbot Laboratories (Port Elizabeth), Pathcare (Port Elizabeth)
Phytoplankton biomass (chlorophyll-*a*)		X	NMMU Botany Department
Phytoplankton taxa		X	NMMU Botany Department

Surface sites only: Tippers Creek (TC), Chatty River (CR), Markman Canal (MMC), Motherwell Canal (MWC), Perseverance (PS)
Surface and near bottom sites: Settlers Bridge (SB), Swartkops Village (SKV), Brickfields (BF), Redhouse Yacht Club (RYC), Bar None (BN)

4.2.1 Nutrients

Water samples for nutrient analyses were collected at sub-surface and near-bottom depths (depending on the depth of the water column) using a weighted pop-bottle. All sampling bottles, glassware and apparatus were acid-stripped with 1% HCl for 24 hours and then rinsed thoroughly with deionized water prior to use. Water samples were filtered through Millipore syringe filters (0.45 µm pore size), stored in 150 ml pharmaceutical bottles and frozen until analyses could commence. All results were expressed as mg l^{-1}.

Ammonium (NH_4^+), total oxidised nitrogen (TOxN) and soluble reactive phosphorus (expressed as dissolved inorganic phosphorus) were analysed using the standard spectrophotometric methods of Parsons *et al*. (1984), Bate and Heelas (1975) and Parsons *et al*. (1984), respectively and dissolved inorganic nitrogen concentrations were obtained through calculation of the sum of NH_4^+ and TOxN.

4.2.2. Trace metals

Water samples for trace metal analyses were collected at sub-surface and near-bottom depths (depending on the depth of the water column) using a weighted pop-bottle. All water samples were kept on ice throughout field sampling and during transportation to the laboratory and analysed for aluminium, arsenic, cadmium, chromium, copper, cyanide, fluoride, iron, lead, mercury, selenium and zinc based. All metals were analysed using accredited techniques (Table 20). Trace metals were expressed as mg l^{-1} or µg l^{-1}.

4.2.3. Phytoplankton biomass

Phytoplankton biomass (chlorophyll-a)

Phytoplankton biomass in the estuary was measured as chlorophyll-*a*. Water samples (~500 ml) were gravity-filtered through Whatman (GF/C) glass-fibre filters. Chlorophyll-*a* was extracted overnight in a cold room by placing the frozen filters into glass vials with 10 ml of 95% ethanol (Merck 4111). Chlorophyll-*a* was spectrophotometrically determined according to Hilmer (1990) as revised from Nusch (1980). Absorbance of the supernatant was determined at 665 nm before and after acidification with 0.1N HCl, using a GBC UV-VIS spectrophotometer. Chlorophyll-*a* was expressed as µg l^{-1} and calculated as follows:

$$Chl\text{-}a\ (\mu g.l^{-1}) = (E_{b665} - E_{a665}) \times 29.6 \times (v/(V \times l))$$

Where:

Eb665 = absorbance at 665 nm before acidification

Ea665 = absorbance at 665 nm after acidification

29.6 = constant calculated from the maximum acid ratio (1.7) and the specific absorption coefficient of chlorophyll-a in ethanol (82 g l^{-1} cm^{-1})

v = volume of solvent used for the extraction (ml)

V = volume of sample filtered (l)

l = path length of spectrophotometer cuvette (cm)

4.2.4. Phytoplankton community composition

Water samples (200 ml) were collected from the sub-surface and near-bottom waters, preserved with glutaraldehyde and stored at 4 °C until further analyses in the laboratory of the Botany Department (NMMU) according to Coulon and Alexander (1972). After settling, the Zeiss IM 35 inverted microscope was used to count and identify the microalgal groups at a maximum magnification of 630x, where either 200 frames were counted per sample or 200 cells was counted. The area of each frame was approximately 3.142 mm^2. The cells were classified according to different microalgae groups, that is, diatoms, flagellates, dinoflagellates, chlorophytes, cyanobacteria, and *Euglena*. The counts for the different phytoplankton groups were expressed as cells ml^{-1} and calculated as follows (Snow, 2000):

$$Cells\ ml^{-1} = ((\pi r2)/A) \times C/V$$

Where:

A = area of each frame (mm^2)

C = number of cells in each frame

V = volume of sample in settling chamber (ml)

4.2.5. Faecal bacteria

Surface waters were collected at all sampling sites and analysed by external laboratories for *Escherichia coli* and enterococci enumeration (Table 21). For method descriptions refer to http://www.idexx.com/resource-library/water/enterolert-e-procedure-en.pdf for enterococci enumeration and http://www.oxoid.com/UK/blue/prod_detail/prod_detail.asp?pr=BR0071&org=71&c=UK&lang=EN for *E. coli* enumeration. Faecal bacteria counts were expressed as counts per 100 ml^{-1}.

4.2.6. Visual observations

Throughout the study period visual observations were noted for the presence of litter or debris and free-floating aquatic plants in the water.

4.2.7. Data presentation and statistical analyses

Short-term water quality survey (2012 to 2013)

Physico-chemical data (salinity, temperature, pH and DO) collected during each sampling session were presented as contour plots using Grapher 6.1.21 (Golden Software, Inc.). A two-way ANOVA (without replication) test was used to determine significant ($p < 0.05$) variability between sampling sites and sampling months, whereas a one-way ANOVA test determined significant ($p < 0.05$) differences between surface and bottom water measurements. Monthly nutrient, chlorophyll-*a* and TSS data were plotted in relation to salinity profiles and the tests for significant differences, normality and variance were completed using the MINITAB Version 15 (Minitab, Inc.) statistical package. Mean values were expressed as mean ± standard error of the mean and all analyses were done at $\alpha = 0.05$. Data were tested for normality using the Bartlett's test for normality.

Table 20: Analyses performed on water samples by Talbot and Talbot Laboratories.

Trace metals ($\mu g\ l^{-1}$)	**Laboratory preparation**	**Laboratory Method**
Fluoride as F (total)	No chemical treatment or filtration	SPADNS method by Spectrophotometry
Cyanide as CN (total)	Distillation	Spectrophotometer DR2700
Iron as Fe (total)	Digested using concentrated HNO_3 acid	Atomic Absorption
Aluminium as Al (total)	Digested using concentrated HNO_3 acid	Atomic Absorption
Cadmium (total)	Digested using concentrated HNO_3 acid	Atomic Absorption
Chromium as Cr (total)	Digested using concentrated HNO_3 acid	Atomic Absorption
Copper as Cu (total)	Digested using concentrated HNO_3 acid	Atomic Absorption
Mercury as Hg (dissolved)	Filtration using 0.45 µm and acidification (HNO_3 acid)	ICP-MS
Lead as Pb (total)	Digested using concentrated HNO_3 acid	Atomic Absorption
Zinc as Zn (total)	Digested using concentrated HNO_3 acid	Atomic Absorption
Arsenic (dissolved)	Filtration using 0.45 µm and acidification (HNO_3 acid)	ICP-MS
Selenium (dissolved)	Filtration using 0.45 µm and acidification (HNO_3 acid)	ICP-MS
Note: 'ICP-MS' = Inductively Coupled Plasma Mass Spectrometry		

Table 21: Analyses performed on water samples by Pathcare.

	Units	**Laboratory preparation**	**Laboratory Method**	**Detection limit**
Escherichia coli	Counts 100 ml^{-1}	No chemical treatment or filtration	Membrane filtration and fluoresence method by plating on MUG agar	< 1 count 100 ml^{-1}
Enterococci	Counts 100 ml^{-1}	1:10 dilution made	Enterolert-E	< 1 count 100 ml^{-1}

If the test result was negative, then a Johnson Transformation was performed to normalise the data. Levene's test for homogeneity of variance, with a confidence interval of 95%, was used to test the equality of variance across variables.

Following a positive test, a general linear model analysis of variance was conducted on the transformed data to test the effect of the different predictors (sampling site, sampling time and depth) on the response variables at $\alpha = 0.05$. The analysis of variance (ANOVA – Tukey's test) was used to determine whether significant spatial and temporal differences existed for nutrient, chlorophyll-*a* and TSS during the study period and at each sampling site and depth. However, when individual sampling sessions were analysed, the Kruskal-Wallis test was used. This test determined significant ($p < 0.05$) spatial and temporal variability in the absence and presence of the three sites which enter the estuary, namely, the Settlers Bridge (representative of the marine environment), Chatty River, Markman Canal, Motherwell Canal and Perseverance (representative of the freshwater environment).

Statistical analyses were not performed on trace metal measurements due to single sample values and some appearing below detection limits. *Escherichia coli* and enterococci counts were spatially and temporally presented. Following log transformation of the bacteria data, a two-way ANOVA (without replication) test was used to determine significant ($p < 0.05$) variability in faecal bacteria counts. Thereafter, differences between spatial and temporal variances were tested for statistical significance using the ANOVA Tukey's test. The short-term study did not generate sufficient data required to statistically confirm whether *E.coli* levels in the estuary pose an on-going risk to human health as a continuous dataset obtained over a period of five years was required. However, for enterococci counts, a 'single sample target value' was available and used for compliance analyses (RSA DEA, 2012; see section 2.8.2). As there was no 'single sample target value' for *E. coli* levels in marine waters, compliance assessment on the estuarine water studied during 2012 and 2013 could not be determined. However, the long-term assessment criteria were used when the historical data were intergrated with the current data. Spearman rank correlation was used to test the strength of association between all physico-chemical, nutrients, chlorophyll-*a*, phytoplankton species biomass (cells ml^{-1}; log transformed) and faecal bacteria (log transformed). Additionally, phytoplankton groups with a relative abundance greater than 10% were considered to be dominant. Trace metals were not included here due to single value measurements (i.e. a small population size) obtained during the study period. To determine the overall health status of the estuary and to indicate the suitability of water at each monitoring location for a particular water use, water quality data were compared with water quality guidelines.

Historical data

Water quality data were statistically analysed for spatial and temporal trends. Where sufficient data were not available, historical data were descriptively addressed. This was the case for phytoplankton dominant taxa and free-floating plants (macrophytes; presence and absence observations only) recorded at estuarine monitoring sites. Dissolved inorganic nitrogen (total oxidised nitrogen and ammonium) and soluble reactive phosphorus were expressed as mg l^{-1} [DIN] and mg l^{-1} [dissolved inorganic phosphorus, DIP], respectively. Bacteria levels (i.e. *E. coli*) were expressed as counts 100 ml^{-1} (or log count 100 ml^{-1} for graphical representation) and trace metals expressed as μg l^{-1}.

For all parameters and where possible, the single factor ANOVA test was used to determine annual/seasonal variability, whereas Pearson's Product Moment Correlation was used to determine increases or decreases with time. Additionally, historical data for DIN and DIP of the freshwater sites of the Swartkops River were presented as frequency distributions of the trophic levels for DIN and DIP, and also presented as percentages exceeding acceptable levels of DIN (< 0.5 mg l^{-1}) and DIP (< 0.005 mg l^{-1}) for freshwater systems (DWAF, 1996a). Since historical data on nutrient levels within the Swartkops Estuary were insufficient for statistical analyses, historical data on nutrient levels in the river reaches were analysed as a means of determining the impact that river inflow has had on the water quality of the Swartkops Estuary over the years.

In addition, all water quality parameters (including the short-term water quality survey results; see below) were spatially and temporally presented as the range and/or mean values (dependent on which forms were obtainable from the literature). DIN and DIP were presented as the range, median, mean, annual/seasonal variability ($p < 0.05$), annual correlation ("r", linearity with time), and the significance thereof ($p < 0.05$). For trace metal analyses, historical data from the varied sources were graphically presented as single mean values and assessed for temporal and spatial trends. Historical data for bacteria countrs were presented as frequency distributions.

For compliance analyses, all trace metal concentrations ($\mu g\ l^{-1}$) except iron and aluminium were compared with the SANS 241:2011 drinking water quality guidelines (as a means of accessing the suitability of the water for recreational water use; see section 2.7.2) and with the DWAF (1995) guidelines for suitable levels of trace metals in coastal aquatic ecosystems. In addition to this, all historical data, including the short-term water quality survey results (see below) were graphically presented as the range and/or mean values (dependent on which forms were available). For compliance analyses, all *E. coli* data were compared with the guidelines for recreational water use of coastal waters as described in RSA DEA (2012).

To illustrate the relationship between freshwater inflow and inorganic nutrient, historical nutrient data recorded at Nivens Bridge between 1995 and 2012 were related to river inflow data also recorded at Nivens Bridge. Additionally, nutrient loads from WWTW were assessed according to daily nutrient loads ($kg\ d^{-1}$) discharged from each sewage plant, including the minimum and maximum ranges recorded from 2009 to 2013 and thereafter related to minimum and maximum inorganic nutrients recorded at estuarine and freshwater monitoring sites.

4.3. **Results**

4.3.1. Physico-chemical parameters

Table 22 shows freshwater flow readings, rainfall measurements and the tidal regime on the days that sampling occurred. "Wet-weather" conditions (i.e. 5 mm or more rainfall recorded in the 24 hours prior to sampling; MacKay, 1994) were not recorded in the Uitenhage area and in

Bluewater Bay. Higher flow rates were recorded on 20 November 2012 as a result of high rainfall recorded in October 2012 (< 233 mm) (Figure 42; see Appendix C), prior to sampling on 20 November 2012, higher flow rates were recorded on the day of sampling. Sampling occurred during both high flow (1.37 – 2.14 $m^3 s^{-1}$) and low flow periods (0.22 – 0.50 $m^3 s^{-1}$), as reflected by rainfall in the weeks prior to sampling (Figure 42; see Appendix C). Figure 8 shows annual rainfall measurements recorded between 1995 and 2013. The data showed that annual rainfall has been variable for the period 1995 to 2013 and that the total annual was higher in the first year (678 mm) of the study compared to the second year (338 mm). The average rainfall for the catchment area was 472 $\pm$ 23.3 mm (283 – 701 mm).

The effect of rainfall on river flow rate is depicted in Figure 9, and shows a positive correlation between the two variables and a regular occurrence of flow events in the catchment area. During the past 18 years, a median flow of 0.10 $m^3 s^{-1}$ and a mean flow of 1.92 $\pm$ 0.23 $m^3 s^{-1}$ (0 – 665 $m^3 s^{-1}$) has been recorded at Nivens Bridge with 94% of these flow readings lower than the mean, indicating that the mean flow rate is weighted by sporadic episodes of high flow. However, during the past 5 years (2009 to 2013), the average flow has increased to 2.92 $m^3 s^{-1}$ (0 – 665 $m^3 s^{-1}$) with 94% of the flow readings lower than the mean.

Figure 10 and Figure 11 show monthly profiles of rainfall and freshwater inflow (at Perseverance: autocorrected from flow at Nivens Bridge) respectively. Monthly rainfall patterns recorded during the study and historical data showed subtle differences. Flow data showed that in the months in which sampling occurred freshwater flow to the estuary was lower in February, May and August 2013 compared to historical flow measurements and higher in September and November 2012 compared to the historical data.

Table 22: Sampling dates and hydrodynamic conditions at the time of sampling.

Sampling date	**Mean daily flow rate [monthly mean] ($m^3 s^{-1}$)**	**Rainfall: 24 hours prior to sampling [< 10 days] (mm)**		**Tide[1]**
		Uitenhage	**Bluewater Bay**	
18/09/2012*	1.37 [1.36]	1.2 [6.8]	2.2 [5.8]	High: 4:41 AM (1.94 m) Low: 10:42 AM (0.15 m)
20/11/2012 (neap tide)	2.14 [8.86]	0 [3.4]	0 [12.8]	High: 8:33 AM (1:56 m) Low: 2:49 PM (0.80 m)
12/02/2013 (spring tide)	0.50 [0.33]	0 [31.8]	0 [29.8]	High: 4:56 AM (2.08 m) Low: 11:01 AM (0.29 m)
21/05/2013 (flood tide)	0.31 [0.30]	0	0.2 [1.2]	Low; 6:51 AM (0.58 m) High: 12:53 PM (1.51 m)
14/08/2013 (neap tide)	0.22 [0.21]	0 [1.4]	0.2 [4.4]	Low: 1:54 AM (0.61 m) High: 8:11 AM (1.41 m)

Note: Spring tide on 17/09/2012
Source: [1]www.kwathabeng.co.za/tides/

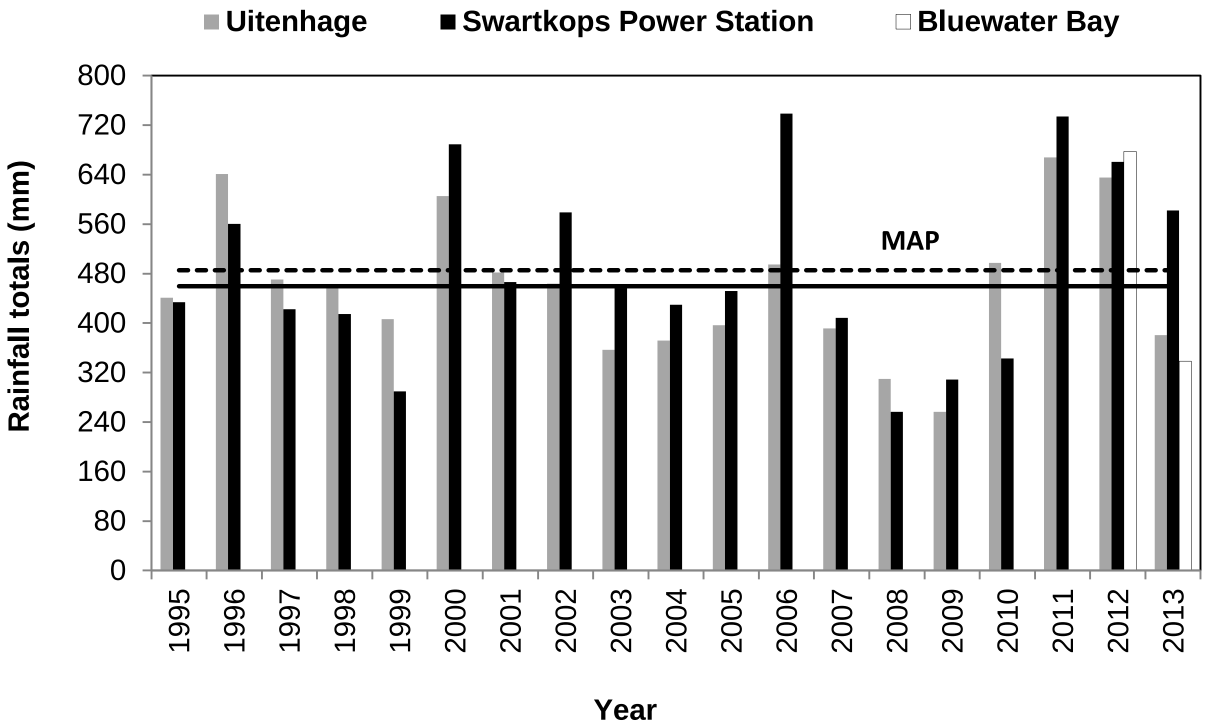


Figure 8: Historical annual rainfall totals recorded at three gauging stations. (The broken line represents the mean annual precipitation (MAP) in Uitenhage and the solid line represents the MAP calculated for the rainfall measured at the Swartkops Power Station).

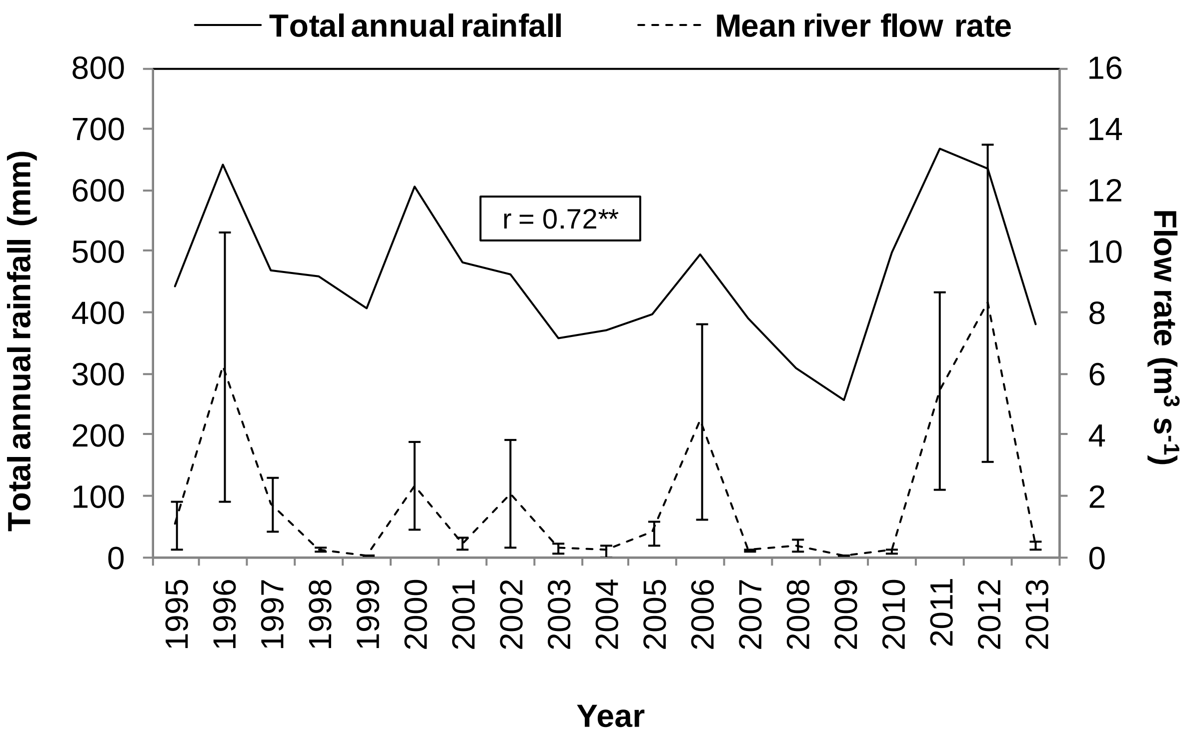


Figure 9: The effect of rainfall on river flow rate at Nivens Bridge showing annual variability. ("r" = correlation between rainfall and flow, ** indicates $p < 0.05$).

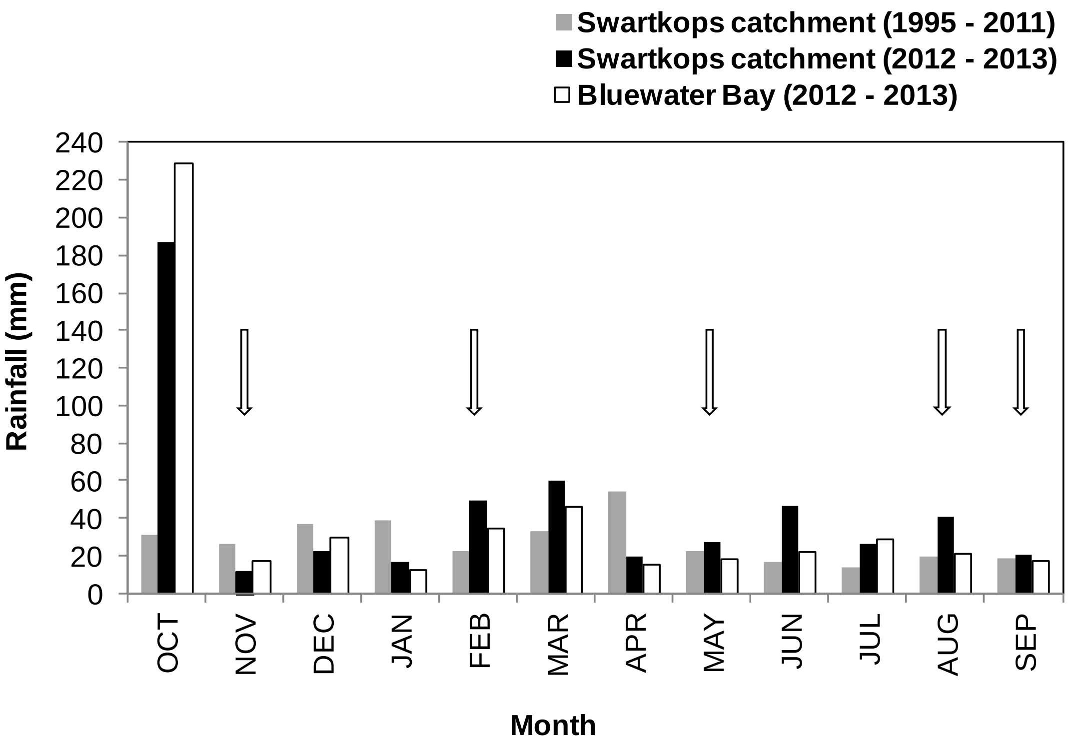


Figure 10: Historical monthly rainfall (1995 – 2011) and recent monthly rainfall recorded in September 2012, November 2012, February 2013, May 2013 and August 2013. (Arrows denote sampling months).

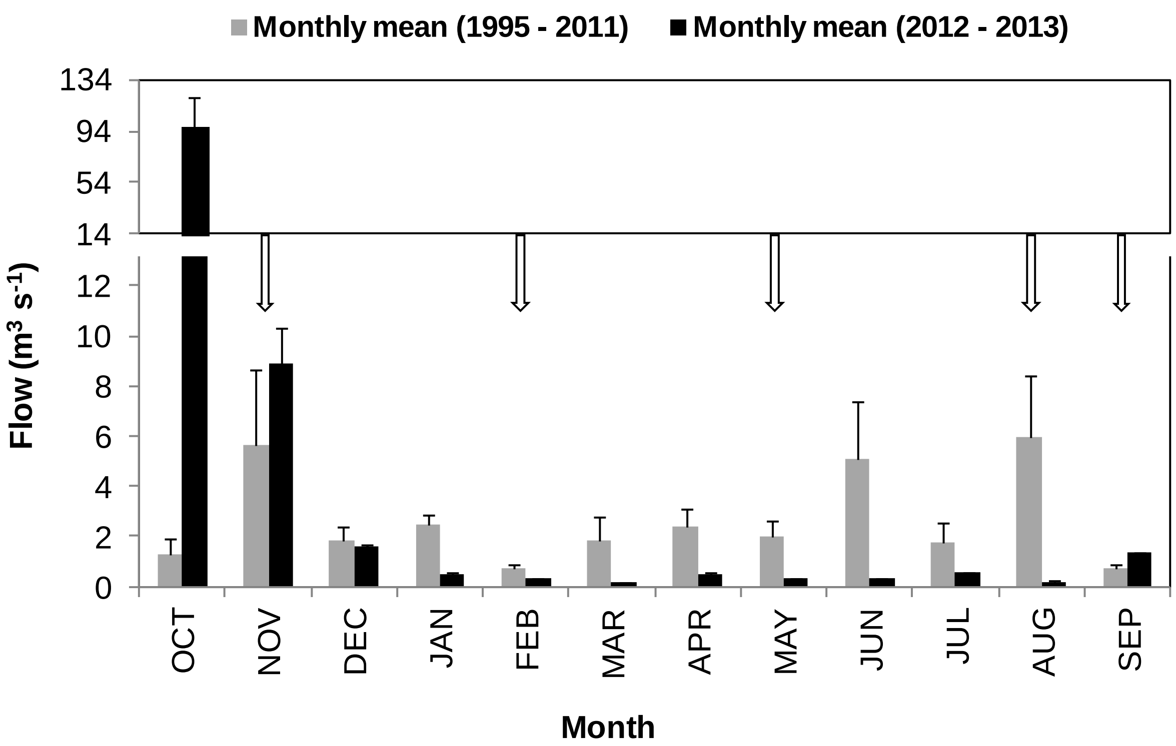


Figure 11: Historical monthly rainfall (1995 – 2011) and recent monthly rainfall recorded in September 2012, November 2012, February 2013, May 2013 and August 2013. (Arrows denote sampling months).

4.3.2. Physico-chemical parameters

Salinity

a) Short-term spatial and temporal study (2012 – 2013)

Salinity showed a typical marine-dominated profile; characterised by a permanently open mouth and a consistent base flow of river water. Measurements decreased significantly ($r = -0.93$; $p < 0.05$; $n = 60$) towards the tidal limit (Perseverance, 16.4 km) with values ranging from 1.2 to 35.5 ppt and a mean of 21.2 ± 1.5 ppt ($n = 60$) (Figure 12). As a result, salinity readings between estuarine sites were significantly ($F = 36.1$; $df = 6$; $p < 0.05$; $n = 35$) different. The degree of stratification (i.e. the difference in salinity between surface and bottom water) (Figure 13) varied between sampling sessions, though never exceeded 10 ppt at any site. A maximum vertical difference of 7.6 ppt was recorded at Swartkops Village (4.0 km from the mouth) in November 2012 and a difference of 7.9 ppt was recorded at Redhouse Yacht Club (10.0 km from the mouth) in May 2013. Figure 14 illustrates significant ($F = 3.4$; $df = 4$; $p < 0.05$; $n = 35$) temporal changes, where the highest mean salinity was recorded in August 2013 (25.8 ± 3.03 ppt) and the lowest in November 2012 (18.2 ± 1.85 ppt). At the tidal limit of the estuary, salinity was lowest in February (~1.2 ppt; spring ebb tide) and highest in August 2013 (~2.4 ppt; neap flood tide). Salinity at the head of the estuary remained relatively constant but the extent of marine intrusion was most apparent at Bar None, 13.6 km from the mouth; September 2012 (8.4 ppt), May 2013 (14.9 ppt), and August 2013 (17.2 ppt). Mean salinity recorded at the five points of entry to the estuary showed significant differences between sampling sites ($F = 67.2$; $df = 4$; $p < 0.05$; $n = 25$). Salinity levels were generally higher in the Chatty River (7.1 ± 1.07 ppt), followed by Markman Canal (3.4 ± 0.60 ppt) and Motherwell Canal (3.2 ± 0.23 ppt).

b) Comparison with past data

The salinity regime of the estuary has been noted as varying from year to year depending on the amount of rain received (Baird *et al*., 1986). According to MacKay and Schumann (1990) there is a tendency to stratification at neap tides or during periods of increased freshwater inflow and conversely, a tendency to vertical mixing at spring tide; trends which were also observed in the present study. Table 33 (see Appendix C) compares salinity measurements recorded since 1979 to salinity recorded during 2012 and 2013. At a distance of approximately 13.6 km from the mouth (Bar None), the estuary was fresher in this study compared to past data. Surface and bottom salinity differences were also greater than past data. Vertical differences ranged from 0 to 7.9 ppt, whereas in the past differences were smaller (1 to 2 ppt; MacKay, 1994). The observed changes could be related to increased flow from upstream WWTW.

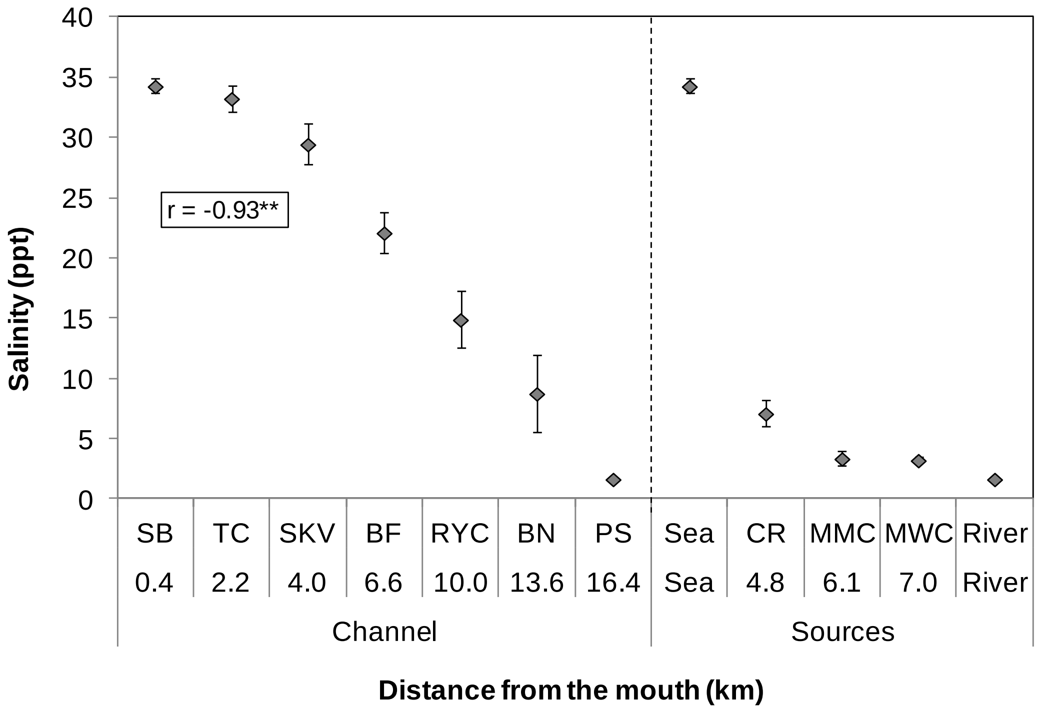


Figure 12: Salinity (ppt) regime of the estuary channel and at points of entry into the estuary ("sources"). (Mean ± SE, "r" = correlation with distance from the mouth, ** indicates $p < 0.05$).

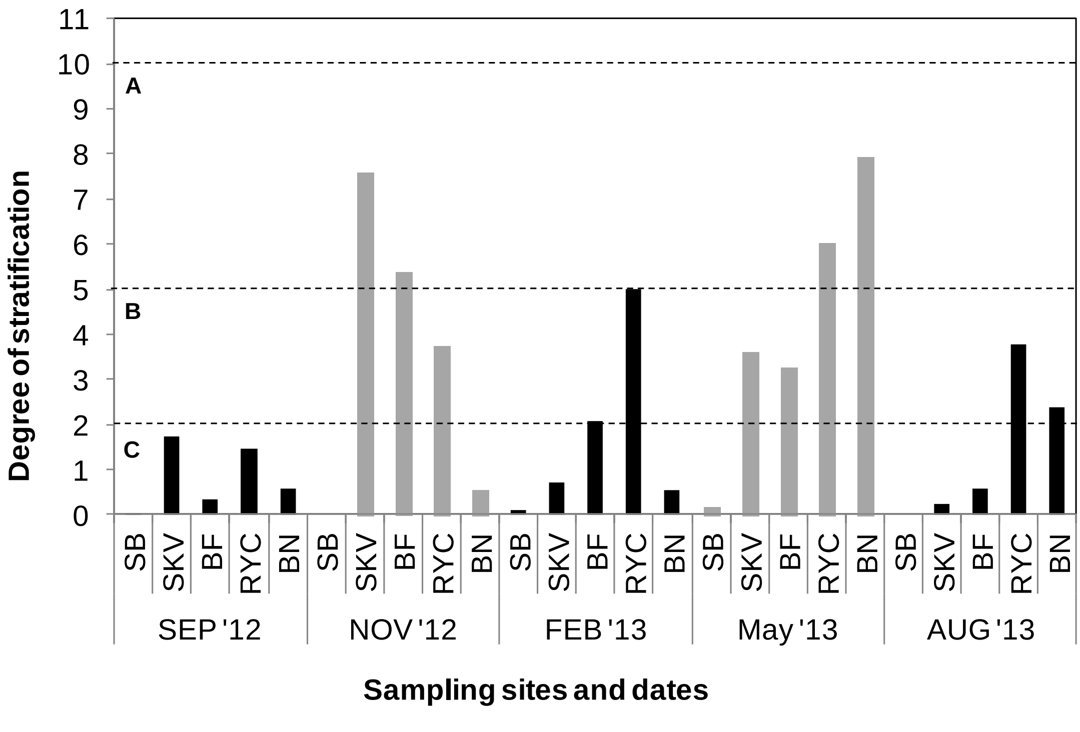
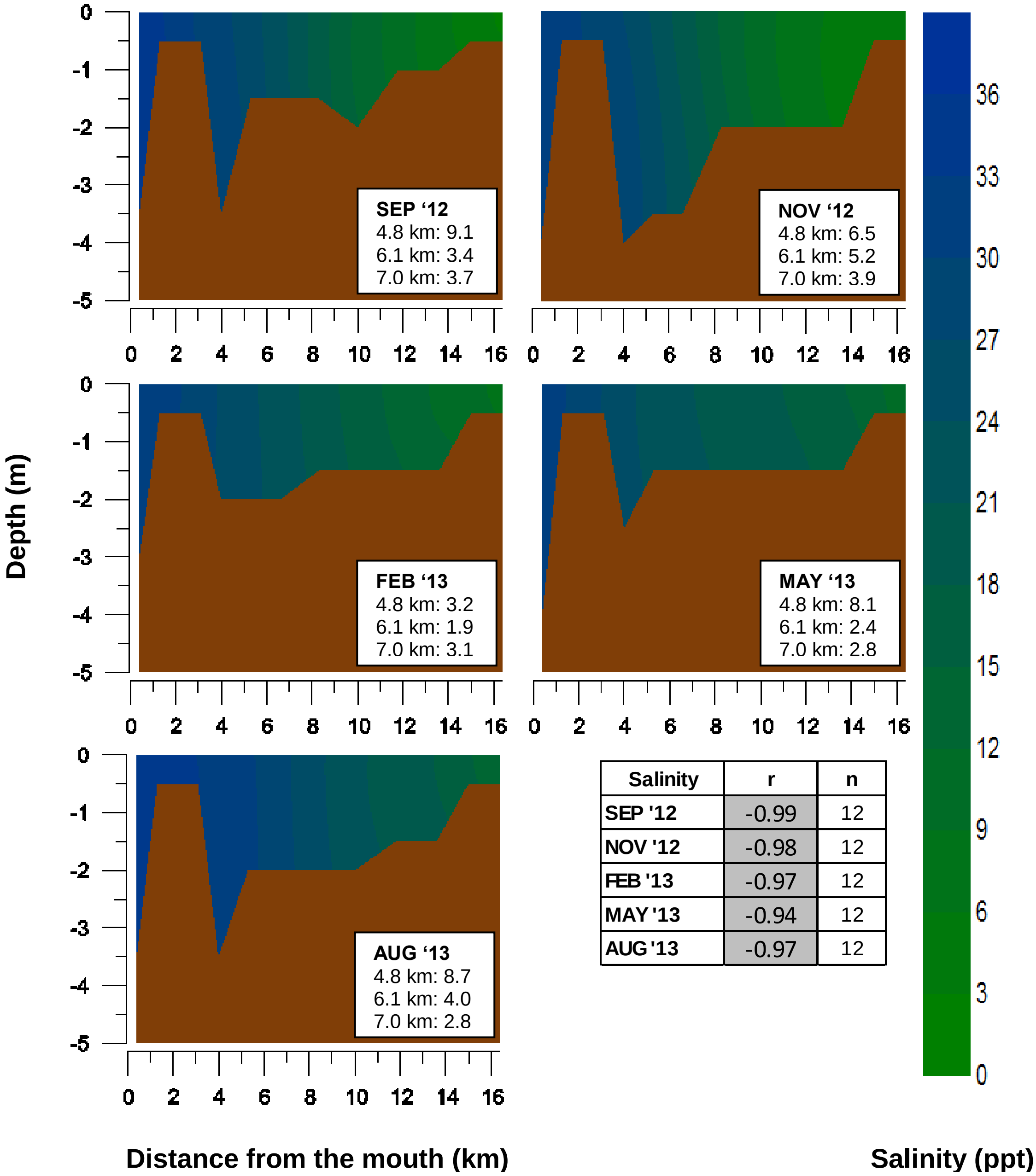


Figure 13: Salinity stratification based on differences between surface and bottom salinity measurements. ("A" = partially stratified, strong; "B" = partially stratified, weak; "C" = vertically homogenous).



Salinity	r	n
SEP '12	-0.99	12
NOV '12	-0.98	12
FEB '13	-0.97	12
MAY '13	-0.94	12
AUG '13	-0.97	12

Figure 14: Monthly salinity (ppt) profiles of the estuary from the mouth (Settlers Bridge, 0.4 km) to the tidal limit (Perseverance, 16.4 km). (The inserts refer to salinity measured at the three points of entry into the estuary (4.8 km: Chatty River, 6.1 km: Markman Canal, 7.0 km: Motherwell Canal) and the table shows correlation ("r") with distance from the mouth. Values shaded in grey are significant ($p < 0.05$)).

Temperature

a) Short-term spatial and temporal study (2012 – 2013)

Water column temperatures were significantly different between sampling sites ($F = 3.2$; $df = 6$; $p < 0.05$; $n = 35$) and months ($F = 115.0$; $df = 4$; $p < 0.05$; $n = 35$) and ranged from 13.4 to 27.1 °C (19.6 ± 0.52 °C). Temperatures represented a seasonal pattern, with the lowest values recorded in August 2013 (14.4 ± 0.20 °C) and the highest in February 2013 (25.2 ± 0.54 °C) (Figure 15). Water column temperatures increased with distance from the mouth.

b) Comparison with past data

Table 34 (see Appendix C) compares temperatures recorded since 1979 to temperatures recorded during 2012 and 2013. The data shows that no changes have occurred with time.

pH

a) Short-term spatial and temporal study (2012 – 2013)

The pH of the water column (Figure 16) ranged from 7.1 to 9.1 (8.1 ± 0.05; $n = 60$). Spatial and temporal data showed no significant ($p > 0.05$) differences and tidal change had no observable effect on the longitudinal distribution of pH in the estuary. There was also no significant ($p > 0.05$) difference in pH when comparing water within the estuary to that in the Motherwell and Markman canals, and in the Chatty River.

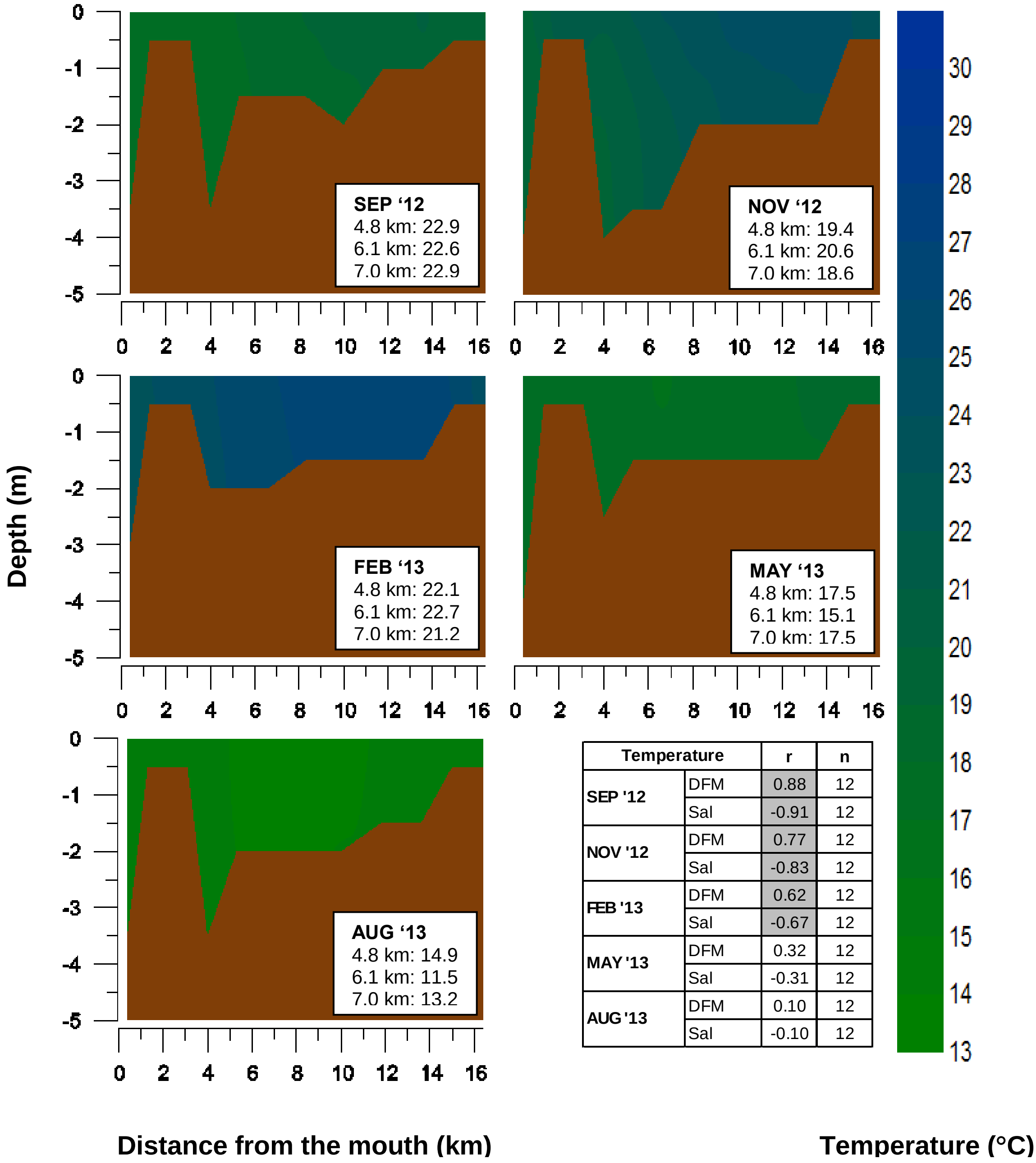
b) Comparison with past data

The results of the present study were consistent with reviewed historical data in that neither longitdunal nor vertical pH gradiets are evident in the estuary (McLachlan, 1972; Emmerson, 1985; Scharler *et al*., 1997). An analysis of past data indicated that water column temperatures have not changed with time (see Appendix C: Table 35).

Dissolved oxygen

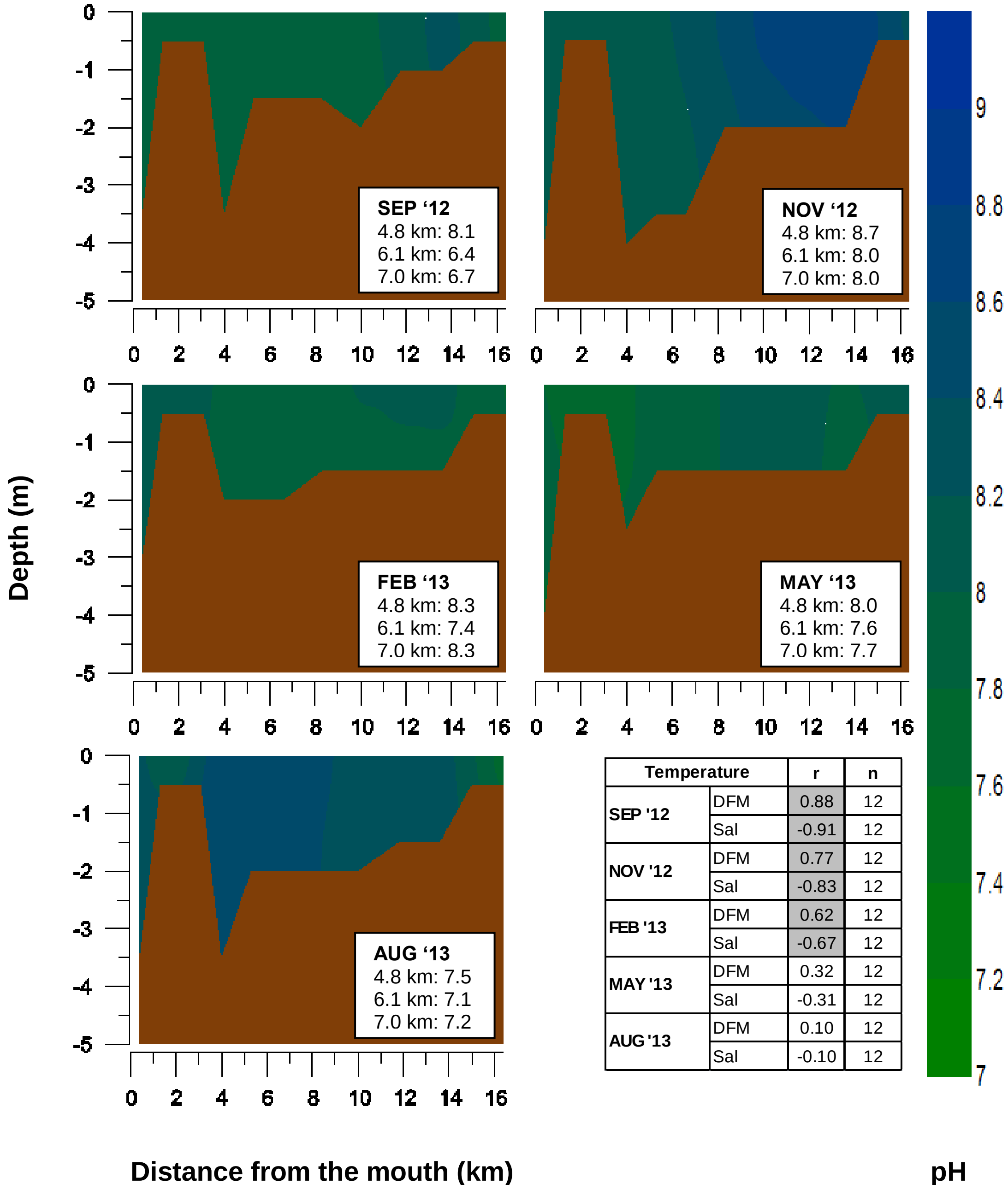
a) Short-term spatial and temporal study (2012 – 2013)

Dissolved oxygen (Figure 17) ranged from 1.3 to 18.2 mg l^{-1} (7.2 ± 0.43 mg l^{-1}; $n = 60$) showing significant differences between surface and bottom waters ($F = 9.3$; $df = 1$; $p < 0.05$; $n = 50$). Surface-to-bottom DO differences of greater than 6 mg l^{-1} were recorded at Redhouse Yacht Club and Bar None in November 2012 and February 2013. Hypoxic conditions (DO: 2 – 3 mg l^{-1}) were recorded in February 2013 at Brickfields (2.7 mg l^{-1}) and Bar None (1.3 mg l^{-1}) and in May 2013 at Bar None (1.9 mg l^{-1}).



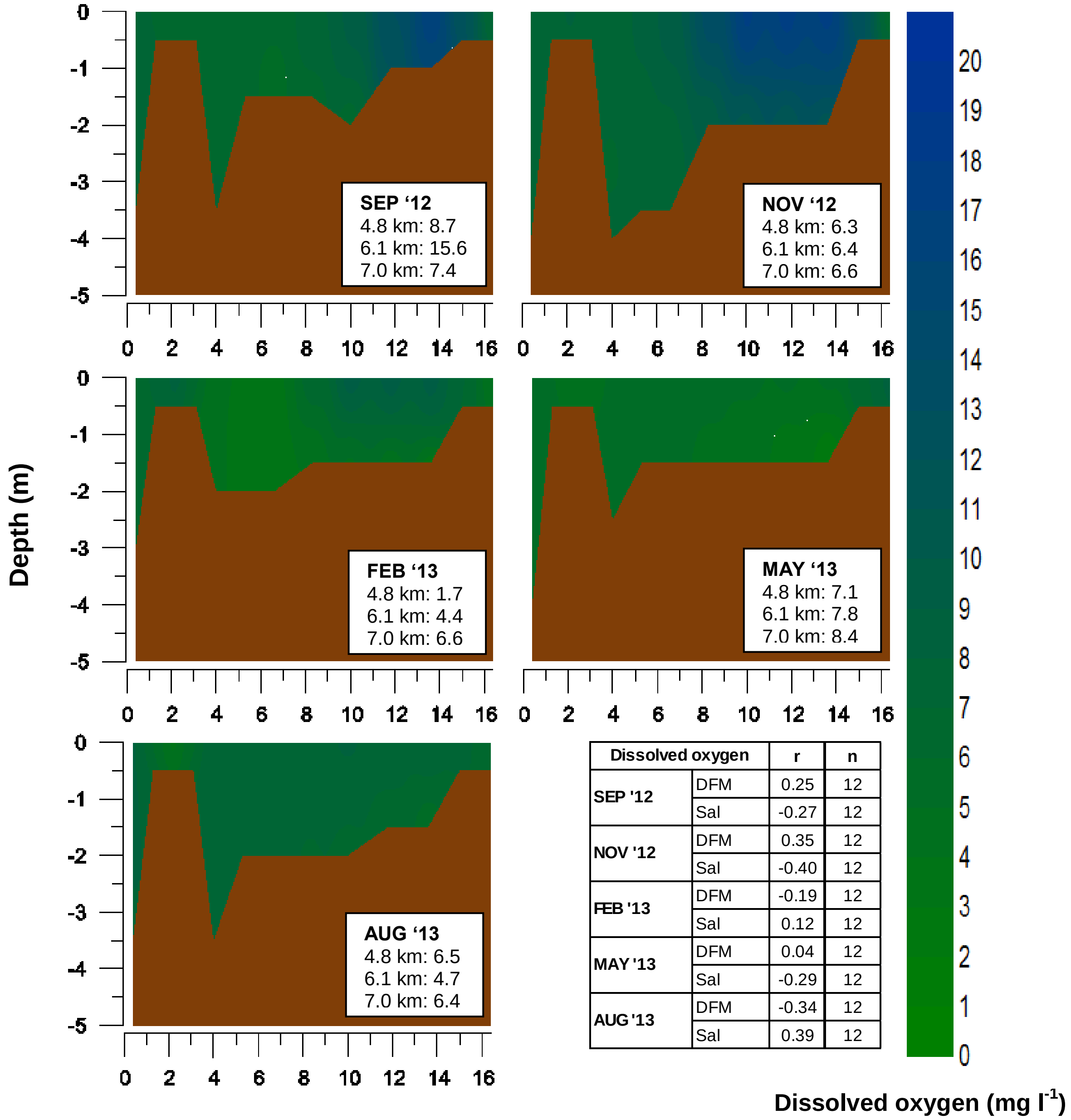
Temperature		r	n
SEP '12	DFM	0.88	12
	Sal	-0.91	12
NOV '12	DFM	0.77	12
	Sal	-0.83	12
FEB '13	DFM	0.62	12
	Sal	-0.67	12
MAY '13	DFM	0.32	12
	Sal	-0.31	12
AUG '13	DFM	0.10	12
	Sal	-0.10	12

Figure 15: Monthly temperature (°C) profiles of the estuary from the mouth (Settlers Bridge, 0.4 km) to the tidal limit (Perseverance, 16.4 km). (The inserts refer to temperatures measured at the three points of entry into the estuary (4.8 km: Chatty River, 6.1 km: Markman Canal, 7.0 km: Motherwell Canal) and the table shows correlation ("r") between temperature and distance from the mouth (DFM) and salinity (Sal). Values shaded in grey are significant ($p < 0.05$)).



Temperature		r	n
SEP '12	DFM	0.88	12
	Sal	-0.91	12
NOV '12	DFM	0.77	12
	Sal	-0.83	12
FEB '13	DFM	0.62	12
	Sal	-0.67	12
MAY '13	DFM	0.32	12
	Sal	-0.31	12
AUG '13	DFM	0.10	12
	Sal	-0.10	12

Figure 16: Monthly pH profiles of the estuary from the mouth (Settlers Bridge, 0.4 km) to the tidal limit (Perseverance, 16.4 km). (The inserts refer to pH measurements recorded at the three points of entry into the estuary (4.8 km: Chatty River, 6.1 km: Markman Canal, 7.0 km: Motherwell Canal) and the table shows correlation ("r") between pH and distance from the mouth (DFM) and salinity (Sal). Values shaded in grey are significant ($p < 0.05$)).



Dissolved oxygen		r	n
SEP '12	DFM	0.25	12
	Sal	-0.27	12
NOV '12	DFM	0.35	12
	Sal	-0.40	12
FEB '13	DFM	-0.19	12
	Sal	0.12	12
MAY '13	DFM	0.04	12
	Sal	-0.29	12
AUG '13	DFM	-0.34	12
	Sal	0.39	12

Figure 17: Monthly dissolved oxygen (mg l^{-1}) profiles of the estuary from the mouth (Settlers Bridge, 0.4 km) to the tidal limit (Perseverance, 16.4 km). (The inserts refer to dissolved oxygen measured at the three points of entry into the estuary (4.8 km: Chatty River, 6.1 km: Markman Canal, 7.0 km: Motherwell Canal) and the table shows correlation ("r") between dissolved oxygen and distance from the mouth (DFM) and salinity (Sal). Values shaded in grey are significant ($p < 0.05$)).

Significant temporal changes were noted ($F = 3.4$; $df = 4$; $p < 0.05$; $n = 35$) with the highest mean concentration recorded in November 2012 (9.4 ± 1.26 mg l^{-1}) and lowest in May 2013 (5.4 ± 0.57 mg l^{-1}). The highest DO concentrations were measured between the Redhouse Yacht Club and Perseverance, and the lowest at Tippers Creek. It is important to note that DO was super-saturated, reaching 18.2 mg l^{-1}, in the surface water between the Redhouse Yacht Club and Bar None as a result of a dense phytoplankton bloom. The study found no significant differences between the channel sites and the five sites that enter the estuary.

b) Comparison with past data

Historically the water in the Swartkops Estuary has been well oxygenated (McLachlan, 1972; Emmerson, 1985; Scharler *et al*., 1997) with few reported instances of hypoxic conditions. McLachlan (1972) obtained a relatively constant concentration of approximately 4.5 mg l^{-1} from the mouth to the tidal limit, whereas Emmerson (1985) recorded a constant yet higher mean DO concentration of 7.2 ± 0.12 mg l^{-1} and a range of 1.8 to 11.0 mg l^{-1}. A review of historical data (see Appendix C: Table 36) showed that while DO levels have remained relatively constant in the lower and middle reaches of the estuary, DO in the upper reaches and at the tidal limit have oscillated with time. Consistent with the findings by McLachlan (1972), Emmerson (1985), and Scharler *et al*. (1997) were the absence of longitudinal DO gradients; although not consistent with these studies were significant ($p < 0.05$) differences in surface-to-bottom DO concentrations. In the present study (2012 – 2013), DO concentrations were lower during winter, a trend not observed by Emmerson (1985) and Scharler *et al*. (1997). There was no correlation between DO, temperature and salinity.

Water clarity (Secchi depth)

a) Short-term spatial and temporal study (2012 – 2013)

Water clarity (Figure 18) decreased significantly ($r = -0.61$, $p < 0.05$, $n = 25$) from the mouth to the upper reaches of the estuary with measurements ranging from 372 to 47 cm, and an overall mean of 132 ± 13 cm ($n = 25$) for the estuary. Figure 19 illustrates monthly water column clarity in relation to maximum water column depths. Secchi depth measurements in the lower reaches of the estuary (up to Swartkops Village) were variable during each sampling session, and appeared to be influenced by tidal stage and not by the rate of freshwater inflow. The effect of increased freshwater flow on water clarity was evident from the middle to the upper reaches of the estuary, where in November 2012 (~57 cm, neap flood tide, 2.14 $m^3 s^{-1}$) water clarity was most apparent and in August 2013 (~153 cm, neap flood tide, 0.22 $m^3 s^{-1}$) was least evident. Data indicated a sharp decline in water clarity in November 2012 from the mouth (~372 cm) to the upper reaches of the estuary at Bar None (~47 cm); where a dense phytoplankton bloom (> 10 000 cells ml^{-1}) was present.

b) Comparison with past data

During 1993 and 1994, Scharler *et al*. (1997) observed a decrease in water clarity from the head to the mouth (102 ± 65 cm to 86 ± 31 cm) of the estuary, i.e. the water column became less transparent towards the mouth; however, increases from the head to the mouth have also been noted (McLachlan and Grindley, 1974; Daniel, 1994). In these studies, the highest turbidity was often recorded in the middle reaches of the estuary near Brickfields due to polluted stormwater from the Motherwell and/or Markman canals. MacKay (1994) also noted an increase in water clarity from the head to the mouth and further concluded that the water is clearer on the flood tide than on the ebb tide and that water clarity is in most part affected by runoff under moderate and high flow conditions. The longitudinal trend in water clarity observed in this study was consistent with those of McLachlan and Grindley (1974) and Daniel (1994), in that water clarity decreased from the head to the mouth. On average, water clarity of the upper riverine reaches has decreased by approximately 50% since the study by Scharler *et al*. (1997).

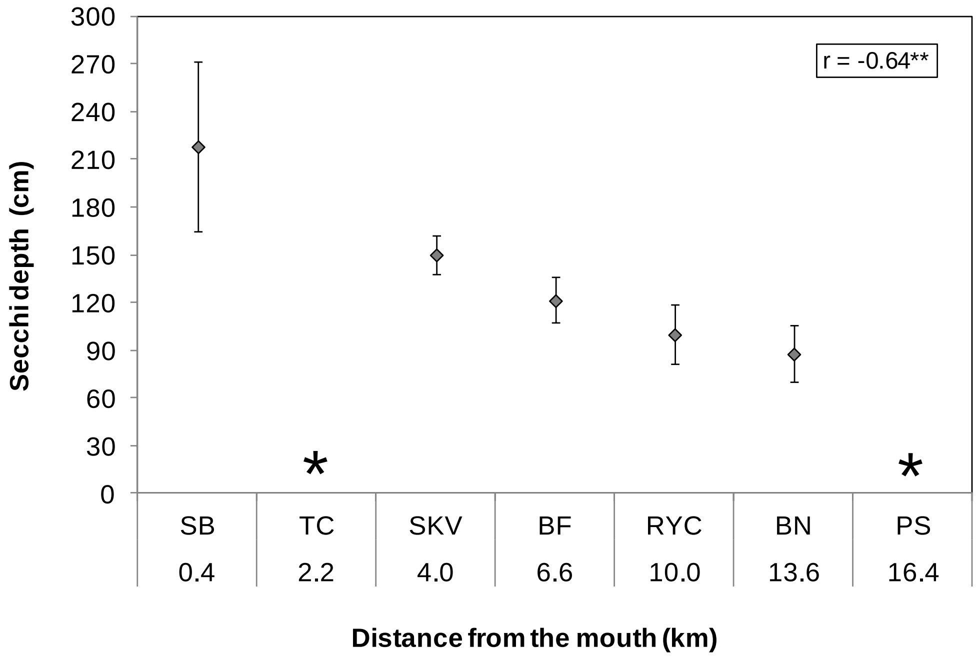


Figure 18: Spatial variation in Secchi depth (cm) recorded in the estuary channel. (Mean ± SE, “r” = correlation with salinity, ** indicates $p < 0.05$, * denotes sites were where water column depths were not recorded).

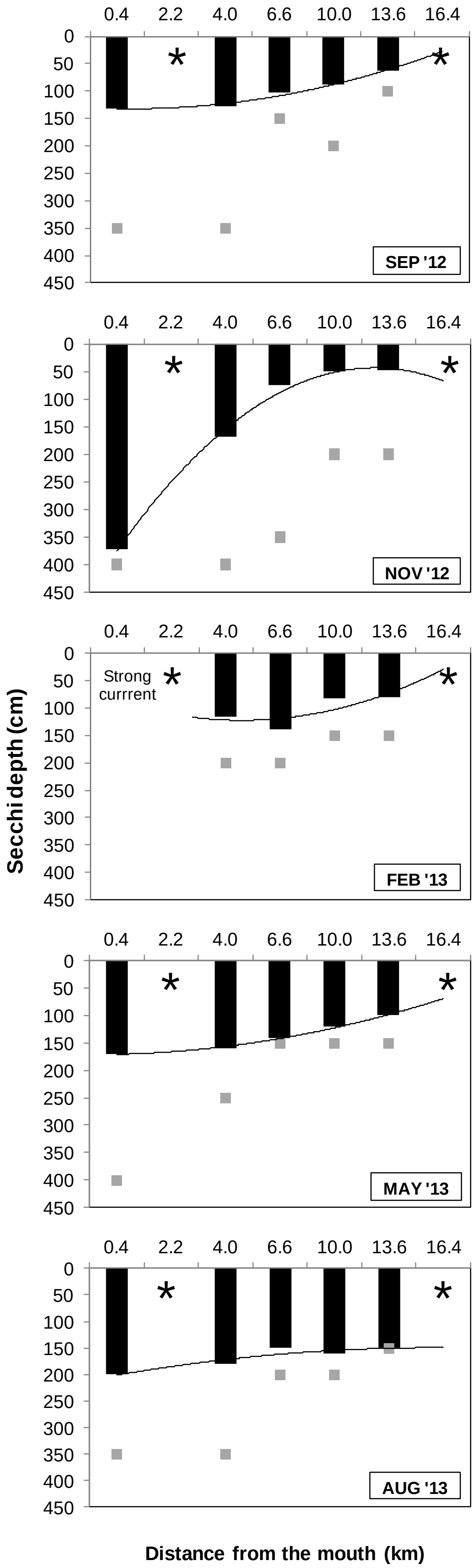


Figure 19: Monthly profiles of Secchi depth (cm) in relation to maximum water column depths (grey squares). (* denotes sites where water column depths were not recorded).

Total suspended solids

a) Short-term spatial and temporal study (2012 – 2013)

Total suspended solids ranged from 5.2 to 240 mg l^{-1} with an overall mean of 36.5 ± 5.05 mg l^{-1} (n = 48) for the estuary (Figure 20). The maximum TSS, 240.4 mg l^{-1}, is an anomaly and was measured at a shallow site in Tippers Creek where fine sediments are easily suspended by boat activity. If this value is excluded, then the overall mean for the estuary is 31.5 ± 2.16 mg l^{-1}. Differences between months (F = 8.3; df = 3; $p < 0.05$; n = 28) (Figure 21) and between surface and bottom waters (F = 8.2; df = 1; $p < 0.05$; n = 40) were significant. Total suspended solids were generally higher in the Markman Canal (~100.1 mg l^{-1}) and in the Chatty River (~70.0 mg l^{-1}). The Swartkops River and the Motherwell Canal were not major sources of TSS.

b) Comparison with past data

In the baseline study conducted by SRK Consulting (Pty) Ltd (2011) in November 2012, a TSS range of 92 to 668 mg l^{-1} was recorded on a spring-ebb tide, with a general decrease from the mouth to the tidal limit (see Appendix C: Table 37). These results and those of the present study were recorded under similar conditions of flow and tidal stage. The data showed that TSS were lower in the present study, and unlike past data decreased with distance from the mouth.

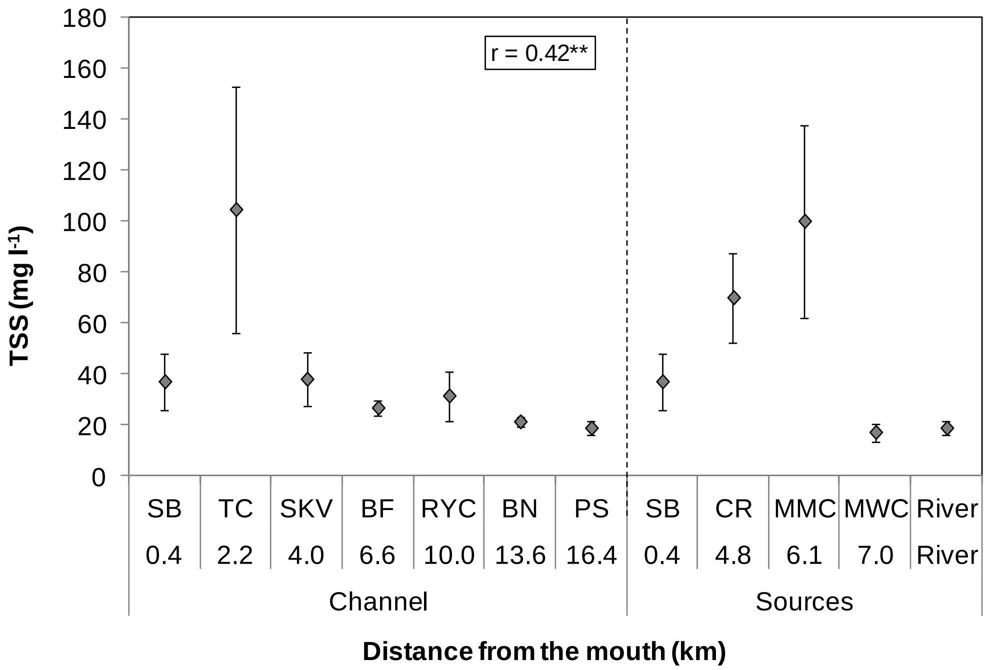


Figure 20: Spatial variation in total suspended solids (TSS; mg l^{-}) recorded in the estuary channel and at points of entry into the estuary ("sources"). (Mean ± SE, "r" = correlation with salinity, ** indicates $p < 0.05$).

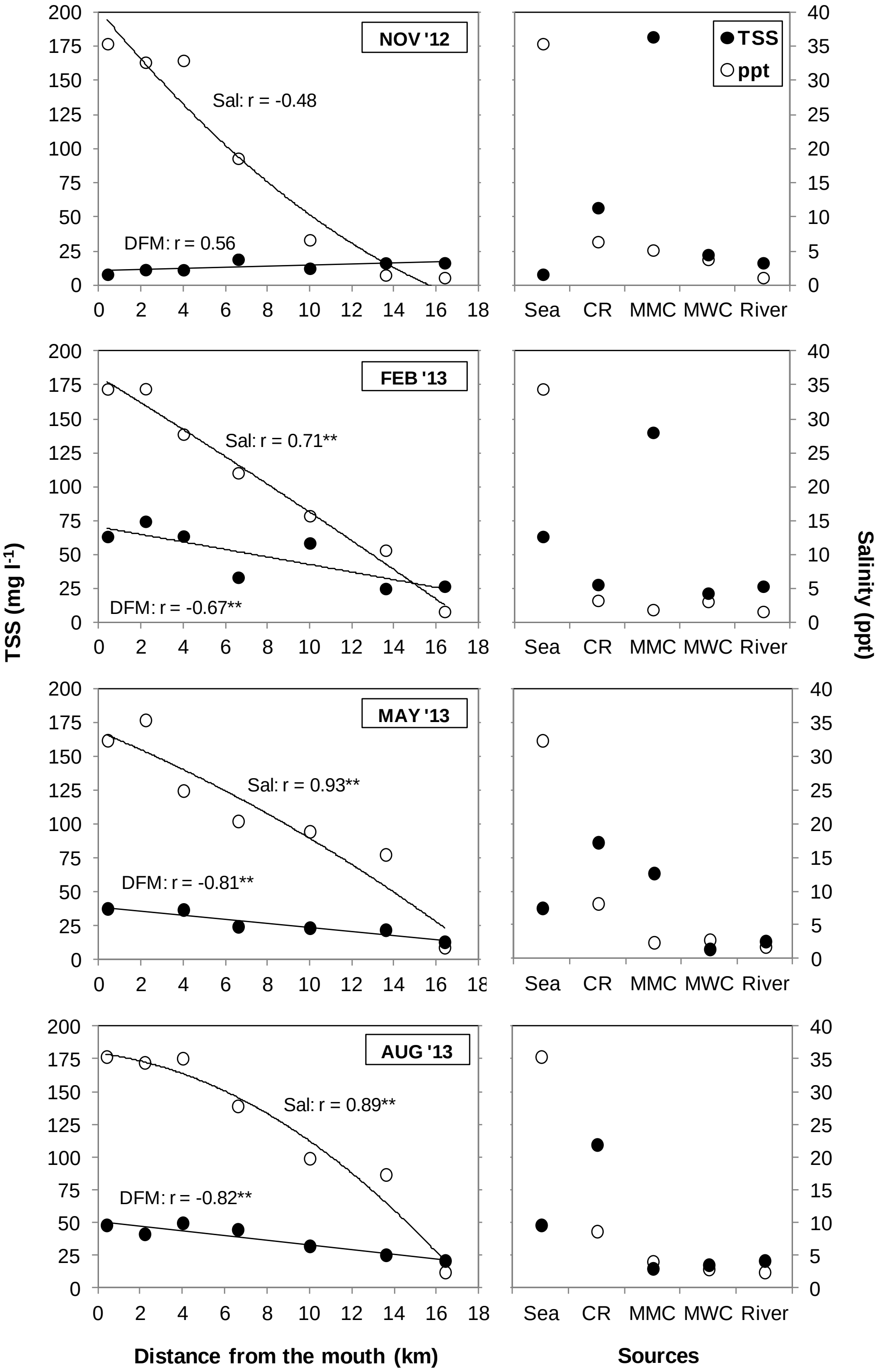


Figure 21: Vertically averaged total suspended solids (TSS; mg l^{-1}) recorded in the estuary channel and at points of entry into estuary ("source"). (Mean ± SE, "Sal" = salinity, "DFM" = distance from the mouth, "r" = correlation with Sal or DFM, ** indicates $p < 0.05$).

4.3.3. Nutrients

Ammonium

a) *Short-term spatial and temporal study (2012 – 2013)*

Ammonium (NH_4^+) increased significantly ($p < 0.05$) with distance from the mouth. Overall, depth averaged NH_4^+ concentrations (Figure 22) showed significant spatial ($F = 4.4$; $df = 6$; $p < 0.05$; $n = 35$) and temporal ($F = 3.7$; $df = 4$; $p < 0.05$; $n = 35$) differences with concentrations ranging from 0 to 1.93 mg l^{-1} (0.20 ± 0.04 mg l^{-1}). Figure 23 shows that the highest mean water column concentration was recorded in February 2012 (0.53 ± 0.26 mg l^{-1}) on a spring ebb tide and the lowest in May 2013 (0.12 ± 0.02 mg l^{-1}) on low tide with an incoming tide. Ammonium concentrations were significantly higher at points entering the estuary between the middle and upper reaches resulting in spatial variability within the estuary channel. The Motherwell Canal (1.36 ± 0.54 mg l^{-1}) was the greatest source of NH_4^+ to the estuary, followed by the Chatty River (1.06 ± 0.40 mg l^{-1}), the Markman Canal (1.00 ± 0.36 mg l^{-1}) and Perseverance (0.76 ± 0.31 mg l^{-1}). Ammonium concentrations recorded at these sites were generally higher in February 2013, and lower in August 2013. Stormwater emanating form the Motherwell Canal had an effect on NH_4^+ concentrations downstream of the canal.

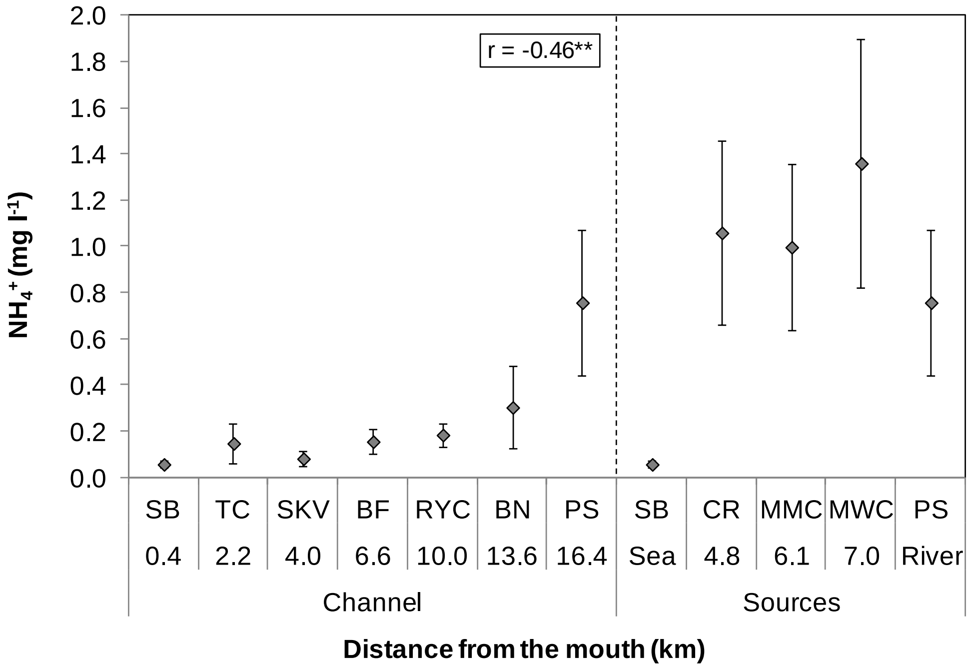


Figure 22: Spatial variation in ammonium concentration (NH_4^+; mg l^{-1}) recorded in the estuary channel and at points of entry into the estuary ("sources"). (Mean ± SE, "r" = correlation with salinity, ** indicates $p < 0.05$).

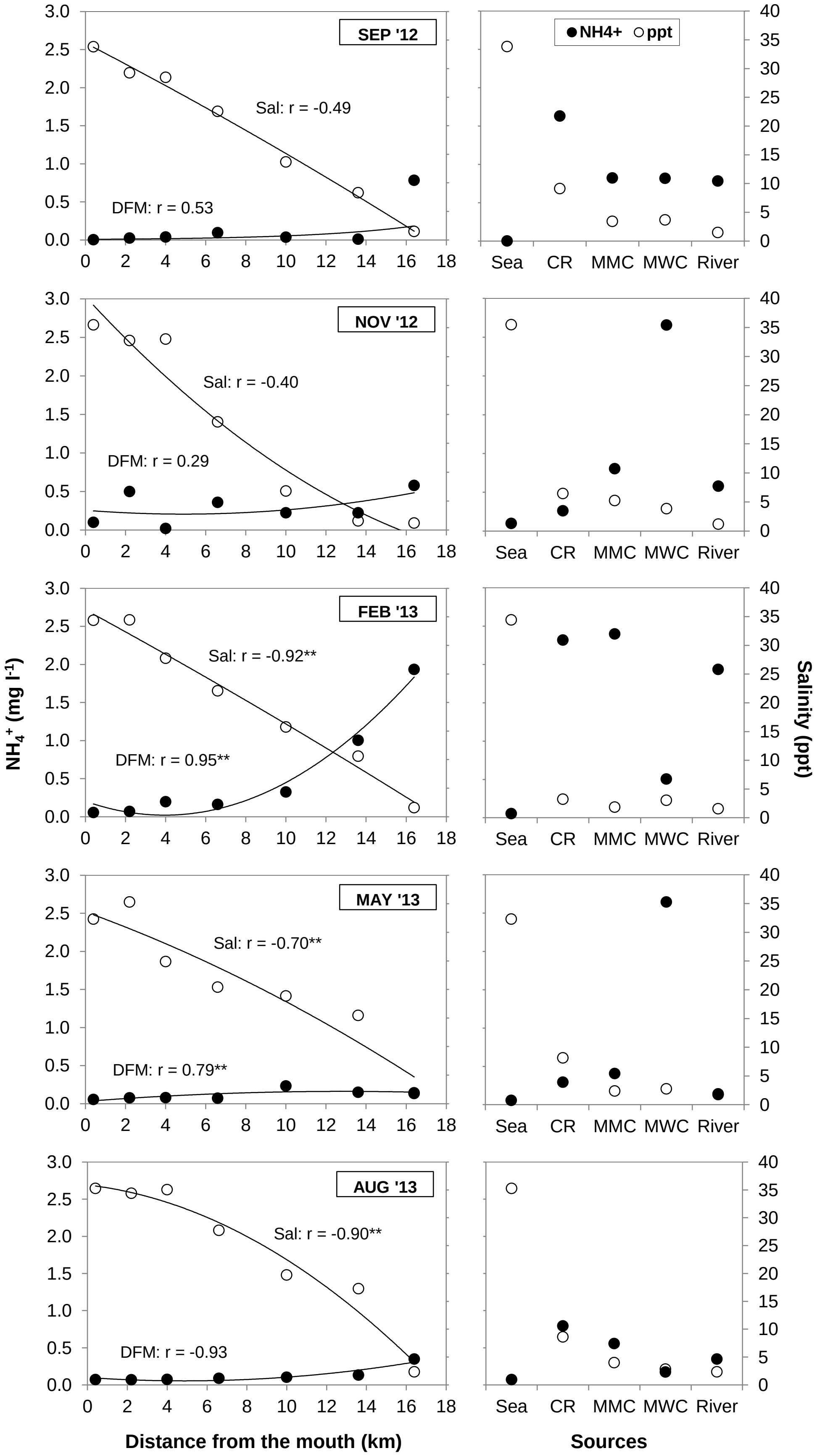


Figure 23: Vertically averaged ammonium concentrations (NH_4^+; mg l^{-1}) measured in the estuary and at points of entry into estuary ("sources"). (Mean ± SE, "Sal" = salinity, "DFM" = distance from the mouth, "r" = correlation with Sal or DFM, ** indicates p < 0.05).

b) Comparison with past data

Historical data shows that average NH_4^+ concentrations have ranged from 0.04 to 0.59 mg l^{-1} (see Appendix C: Table 38). Previous studies recorded lower NH_4^+ concentrations in the middle reaches of the estuary between Brickfields and Bar None to that recorded in the present study, indicating a general increase with time. Studies by Watling (1982), Hilmer (1984), Emmerson (1985), MacKay (1993) and Scharler *et al*. (1997) showed that freshwater flow had no effect on NH_4^+ concentrations in the estuary due to nutrient supplements from the Motherwell Canal and the Chatty River.

Total oxidised nitrogen

a) Short-term spatial and temporal study (2012 – 2013)

Total oxidised nitrogen (TOxN) concentrations (Figure 24) ranged from 0 – 5.09 mg l^{-1} (0.70 ± 0.12 mg l^{-1}) in the present study. Significant differences in TOxN were found between months ($F = 20.7$; df = 4; $p < 0.05$; n = 35) and sites ($F = 49.0$; df = 6; $p < 0.05$; n = 35), and a strong negative correlation between TOXN and salinity ($r = -0.77$; $p < 0.05$; n = 59) was apparent. The highest mean water column concentration was recorded in May 2013 (1.54 ± 0.65 mg l^{-1}), and the lowest in February 2013 (0.35 ± 0.16 mg l^{-1}) (Figure 25). Significant spatial ($F = 11.2$; df = 4; $p < 0.05$; n = 25) differences in TOxN were noted between the sites that enter the estuary – TOxN was highest in the Motherwell Canal (6.73 ± 1.07 mg l^{-1}), followed by the Markman Canal (5.13 ± 1.74 mg l^{-1}), Perseverance (2.60 ± 0.68 mg l^{-1}) and the Chatty River (1.4 ± 0.60 mg l^{-1}). Polluted stormwater from the Motherwell Canal was the principal source of TOxN to the estuary. TOxN recorded at the tidal limit (Perseverance) and in the Markman Canal were statistically similar ($p > 0.05$) and noted as secondary sources. Moreover, the Swartkops River was noted as being as much of a source of TOxN to the estuary as the Chatty River – an observation also noted by MacKay (1994). In November 2012 high freshwater flow (2.14 m^3 s^{-1}) and a receding tide was associated with elevated TOxN at Brickfields (0.71 ± 0.17 mg l^{-1}). However, overall, freshwater inflow and tidal stage showed no apparent effect on TOxN in the estuary.

b) *Comparison with past data*

A review of past and present data reveals that TOxN has always increased with distance from the mouth. Total oxidised nitrogen was lower in the study by Scharler *et al*. (1997) than measurements recorded between 1979 and 1981 by Emmerson (1985) (see Appendix C: Table 39). In the subsequent time series, levels remained relatively constant; however, TOxN was elevated at Brickfields (0.22 mg l^{-1}, 0.09 – 4.69 mg l^{-1}) and in this study, TOxN levels between Bar None and Perseverance were even higher, showing a general increase with time.

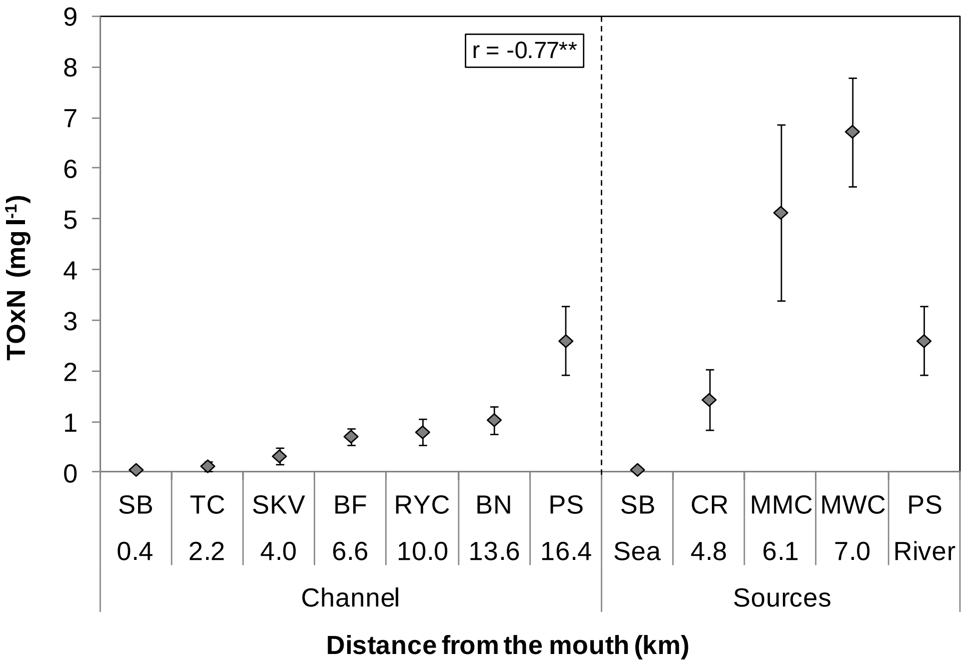


Figure 24: Spatial variation in total oxidised nitrogen concentration (TOxN; mg l^{-1}) recorded within the estuary channel and at points of entry into the estuary (“sources"). (Mean ± SE, “r” = correlation with salinity, ** indicates $p < 0.05$).

Dissolved inorganic nitrogen

a) Short-term spatial and temporal study (2012 – 2013)

Dissolved inorganic nitrogen (DIN) concentrations ranged from 0.01 to 5.23 mg l^{-1} (0.90 ± 0.13 mg l^{-1}) and increased significantly with distance from the mouth ($r = 0.81$; $p < 0.05$; $n = 59$) (Figure 26). Additionally, DIN concentrations recorded across the different months ($F = 5.0$; df = 4; $p < 0.05$; $n = 35$) and sites ($F = 25.5$; df = 6; $p < 0.05$; $n = 35$) were significantly different. The highest mean concentration was recorded in May 2013 (1.65 ± 0.66 mg l^{-1}), and the lowest in August 2013 (0.70 ± 0.31 mg l^{-1}) (Figure 27). There were significant spatial ($F = 66.7$; df = 4; $p < 0.05$; $n = 25$) and temporal ($F = 4.0$; df = 4; $p < 0.05$; $n = 25$) differences between the five points of entry into the estuary. The Motherwell Canal (8.09 ± 1.09 mg l^{-1}) was the greatest source of DIN to the estuary, with levels statistically similar yet lower than levels recorded in the Markman Canal (6.13 ± 1.67 mg l^{-1}) and at the tidal limit (3.35 ± 0.53 mg l^{-1}). Thus, the Swartkops River was considered to be as much of a source of DIN to the estuary as the tributaries located in the middle reaches of the estuary (Scharler and Baird 2003; Potgieter, 2008). Freshwater inflow in November 2012 led to elevated DIN concentrations at Brickfields (1.05 mg l^{-1}) due to nutrient enrichment from the Motherwell Canal (9.50 mg l^{-1}). Mean DIN concentrations recorded between Brickfields and Perserence generally exceeded the 0.50 mg l^{-1} water quality guideline for acceptable levels of DIN in coastal marine ecosystems.

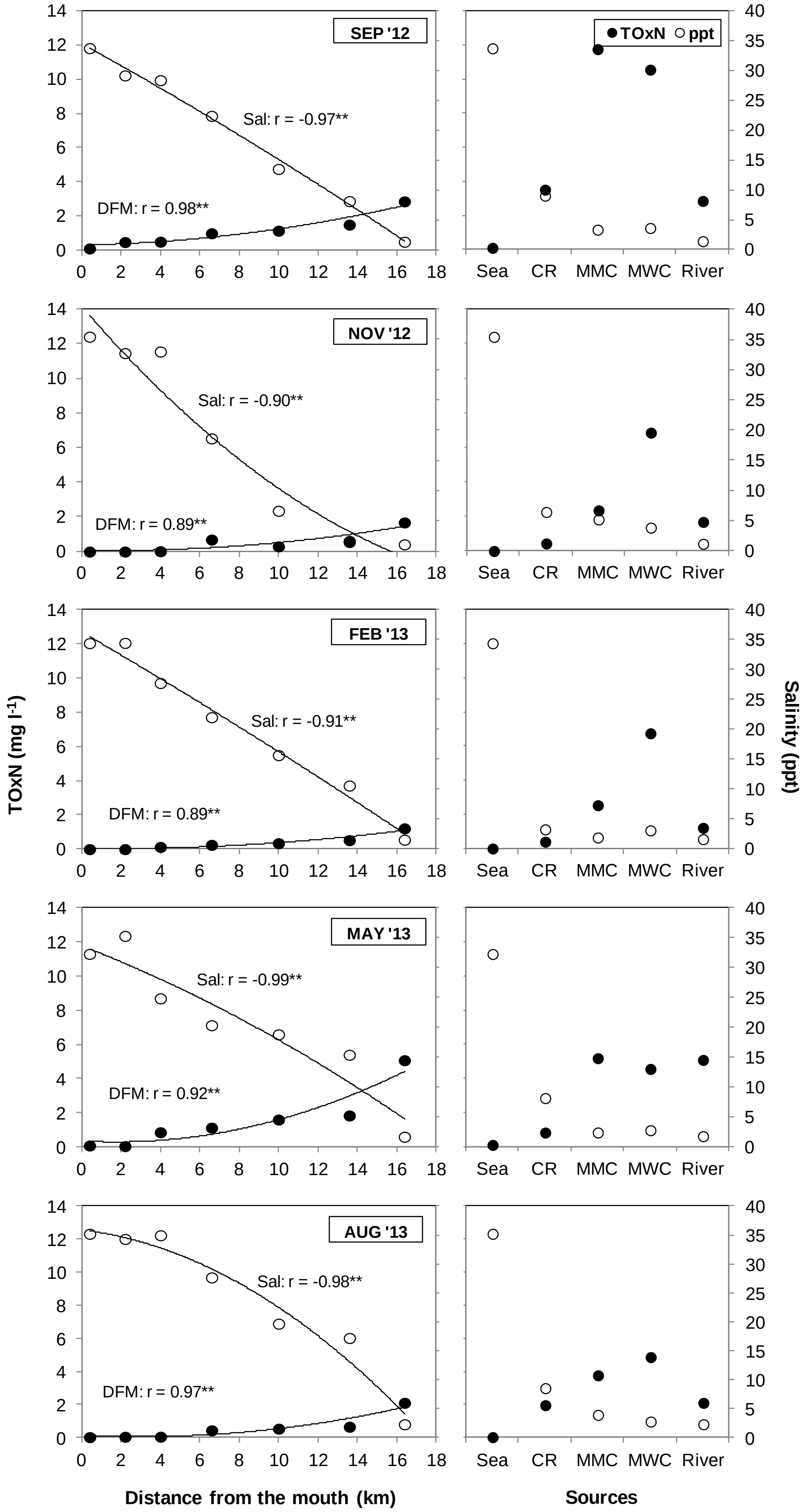


Figure 25: Vertically averaged total oxidised nitrogen concentrations (TOxN; mg l^{-1}) recorded in the estuary channel and at points of entry into estuary ("sources"). (Mean ± SE, "Sal" = salinity, "DFM" = distance from the mouth, "r" = correlation with Sal or DFM, ** indicates $p < 0.05$).

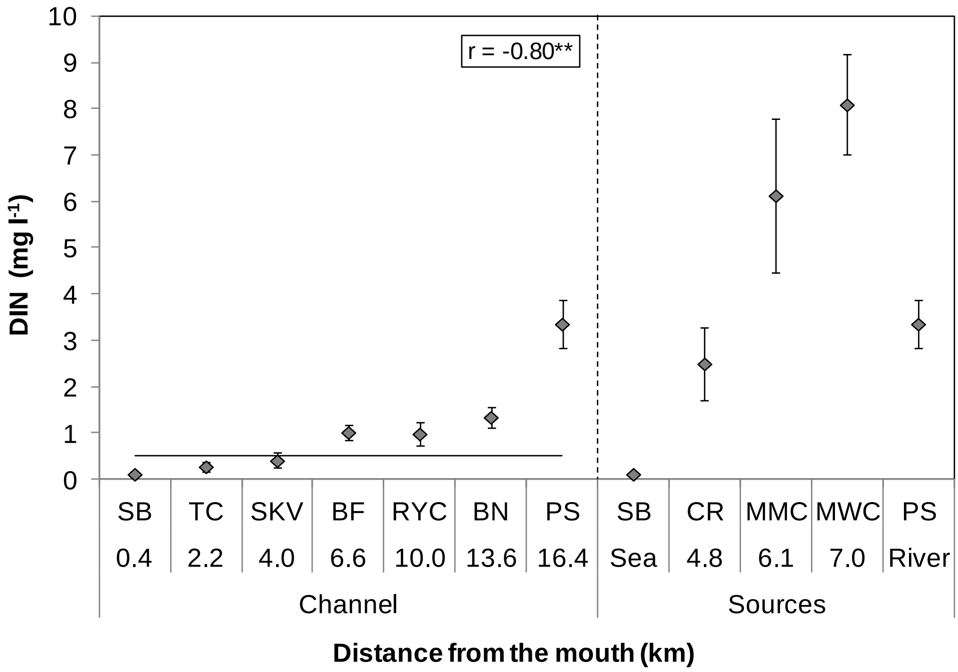


Figure 26: Spatial variation in dissolved inorganic nitrogen concentration (DIN; mg l^{-1}) recorded in the estuary channel and at points of entry into the estuary. (Mean ± SE, "r" = correlation with salinity, ** indicates $p < 0.05$. The horizontal line denotes the guideline threshold for DIN in estuarine systems).

b) *Comparison with past data*

Eutrophic levels of DIN (DIN > 0.5 mg l^{-1}) have prevailed in the Swartkops Estuary since the study by Scharler *et al*. (1997) (see Appendix C: Table 40), but at the time attained relatively low levels of phytoplankton biomass in comparison with the present study. Furthermore, concentrations recorded between Brickfields and Perseverance have exceeded water quality guidelines (0.5 mg l^{-1}; UNEP/Nairobi Convention Secretariat and CSIR, 2009) for marine coastal waters since the study by Scharler *et al*. (1997). Dissolved inorganic nitrogen concentrations at the tidal limit have increased more progressively than concentrations recorded in the middle reaches of the estuary. Dissolved inorganic nitrogen concentration recorded in the Motherwell Canal has increased from 4.94 mg l^{-1} (1.18 – 12.31 mg l^{-1}) to 8.98 mg l^{-1} (2.91 – 13.74 mg l^{-1}) in this study. Potgieter (2008) noted that the Swartkops Estuary acted as a sink for DIN under high and low flow conditions, which suggested a poor relationship between hydrodynamics and inorganic nitrogen in the Swartkops Estuary. The results of the present study corroborate this observation.

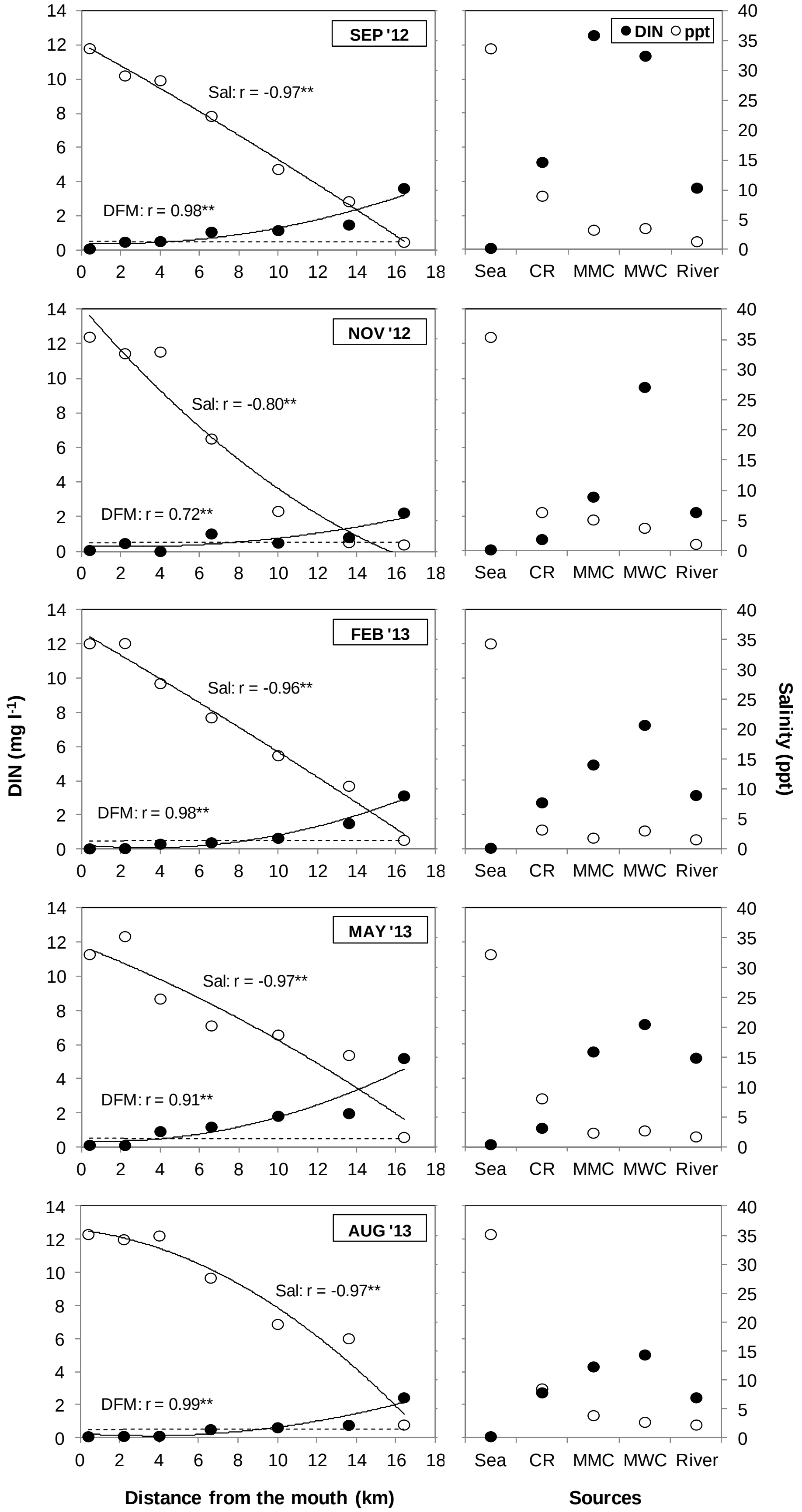


Figure 27: Vertically averaged total oxidised nitrogen concentration (DIN; mg l^{-1}) recorded in the estuary channel and at points of entry into estuary ("sources"). (Mean ± SE, "Sal" = salinity, "DFM" = distance from the mouth, "r" = correlation with Sal or DFM, ** indicates $p < 0.05$).

Dissolved inorganic phosphorus

a) Short-term spatial and temporal study (2012 – 2013)

Dissolved inorganic phosphorus (DIP) ranged from 0.02 to 3.58 mg l^{-1} (0.54 ± 0.08 mg l^{-1}) and increased significantly ($r = 0.89$; $p < 0.05$) with distance from the mouth (Figure 28). Temporal differences in DIP concentrations were significant ($F = 7.8$; df = 4; $p < 0.05$; n = 35), with the highest concentration recorded in August 2013 (0.95 ± 0.48 mg l^{-1}) and the lowest in November 2013 (0.26 ± 0.12 mg l^{-1}) (Figure 29). Significant differences in DIP were found between estuarine sites ($F = 38.1$; df = 6; $p < 0.05$; n = 35), with the Swartkops River being the primary source of phosphate to the estuary. Furthermore, DIP concentrations were significantly ($p < 0.05$) higher at Perseverance (1.78 ± 0.47 mg l^{-1}) than concentrations recorded at Chatty River (0.40 ± 0.08 mg l^{-1}), Motherwell Canal (0.19 ± 0.05 mg l^{-1}), and Markman Canal (0.14 ± 0.02 mg l^{-1}). Throughout the estuary, mean DIP concentrations exceeded the 0.05 mg l^{-1} water quality guideline stipulated for acceptable levels of DIP in coastal marine ecosystems of the West Indian Ocean region.

b) *Comparison with past data*

Present and past studies (Emmerson, 1985; Scharler *et al*., 1997 and MacKay, 1994) have shown that DIP increases in concentration from the mouth to the tidal limit (see Appendix C: Table 41). Emmerson (1985) also found that during flood conditions and high river discharge, DIP levels tended to decline throughout the estuary. Potgieter (2008) concluded that DIP in the river decreases with increasing flow and ascribed this to a dilution effect on DIP at high flow rates; an observation previously reported by Hilmer (1984) and one which has been observed in the present study.

Collated data provided in Table 28 shows that DIP levels recorded in the Swartkops Estuary have exceeded acceptable levels (0.05 mg l^{-1}; UNEP/Nairobi Convention Secretariat and CSIR, 2009) for coastal marine ecosystems as early as the 1970s.

Riverine nutrient levels

Data recorded by DWA from 1995 to 2013 showed DIN levels at Nivens Bridge ranged from 0.04 to 14.40 mg l^{-1} (1.20 ± 0.09 mg $^{l-1}$) whereas DIP ranged from 0.01 to 3.08 mg l^{-1} (0.16 ± 0.02 mg $^{l-1}$) (Table 23). Futhermore, DIN levels are increasing in the Swartkops River (r = 28, $p > 0.05$, n = 343), though not as rapidly as DIP ($r = 0.51$, $p < 0.05$, n = 357) (Table 23). Eight-two percent (292 out of 357) of the DIP concentrations recorded at Nivens Bridge were lower than the mean; which suggested that the mean was weighted by sporadic pulses of DIP into the river.

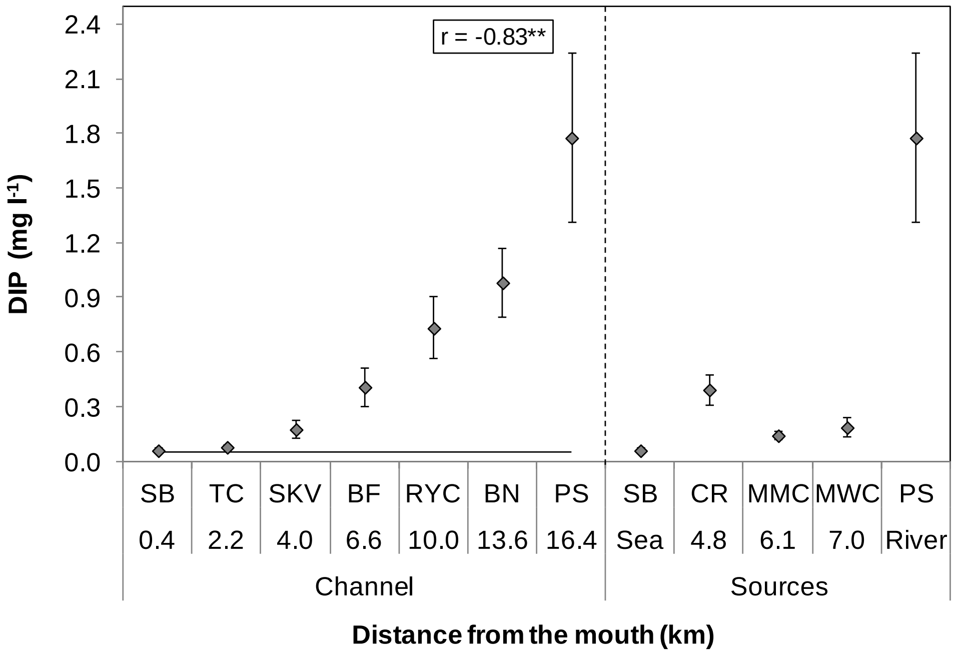


Figure 28: Spatial variation in dissolved inorganic phosphorus concentration (DIP, mg l^{-1}) recorded within the estuary channel and at points of entry into the estuary. (Mean ± SE, “r” = correlation with salinity, ** indicate $p < 0.05$. The horizontal line denotes the guideline threshold for DIP concentrations in estuaries).

Eutrophic (> 2.5 mg l^{-1}) concentrations of DIN have been occurring infrequently in the Swartkops River (Nivens Bridge) (15%; n = 343), whereas mesotrophic (0.5 – 2.5 mg l^{-1}; 38%) and oligotrophic (< 0.5 mg l^{-1}; 48%) concentrations have occured frequently (Figure 30). Futher analyses found that eutrophic (>0.025 mg l^{-1}; 68%) levels of DIP predominate in the Swartkops River and therefore, the river is a greater source of phosphate to the estuary than nitrogen. A compliance assessment (Figure 30) indicated that 52% of DIN concentrations recorded at Nivens Bridge exceeded the guideline value of 0.50 mg l^{-1}, whereas all DIP concentrations recorded over the last 18 years exceeded the guideline of 0.005 mg l^{-1} (DWAF, 1996a) for acceptable levels of DIP in freshwater systems of South Africa.

<u>Influence of effluent discharges on estuarine nutrient levels</u>

The ecological health of an estuary is influenced by the quality and quantity of river water. Unfortunately, the nearest flow gauge to the Swartkops Estuary is located 16 km from Perseverence in an area referred to as Nivens Bridge. Both the quality and quantity of the riverine water within this stretch is impacted by flow contributions from several river tributaries, and stormwater canals, as well as three wastewater treatment works (KwaNobuhle, Kelvin Jones and Despatch). Consequently, correlations between flow measured at the gauging station and water quality variables measured in the estuary could not be determined.

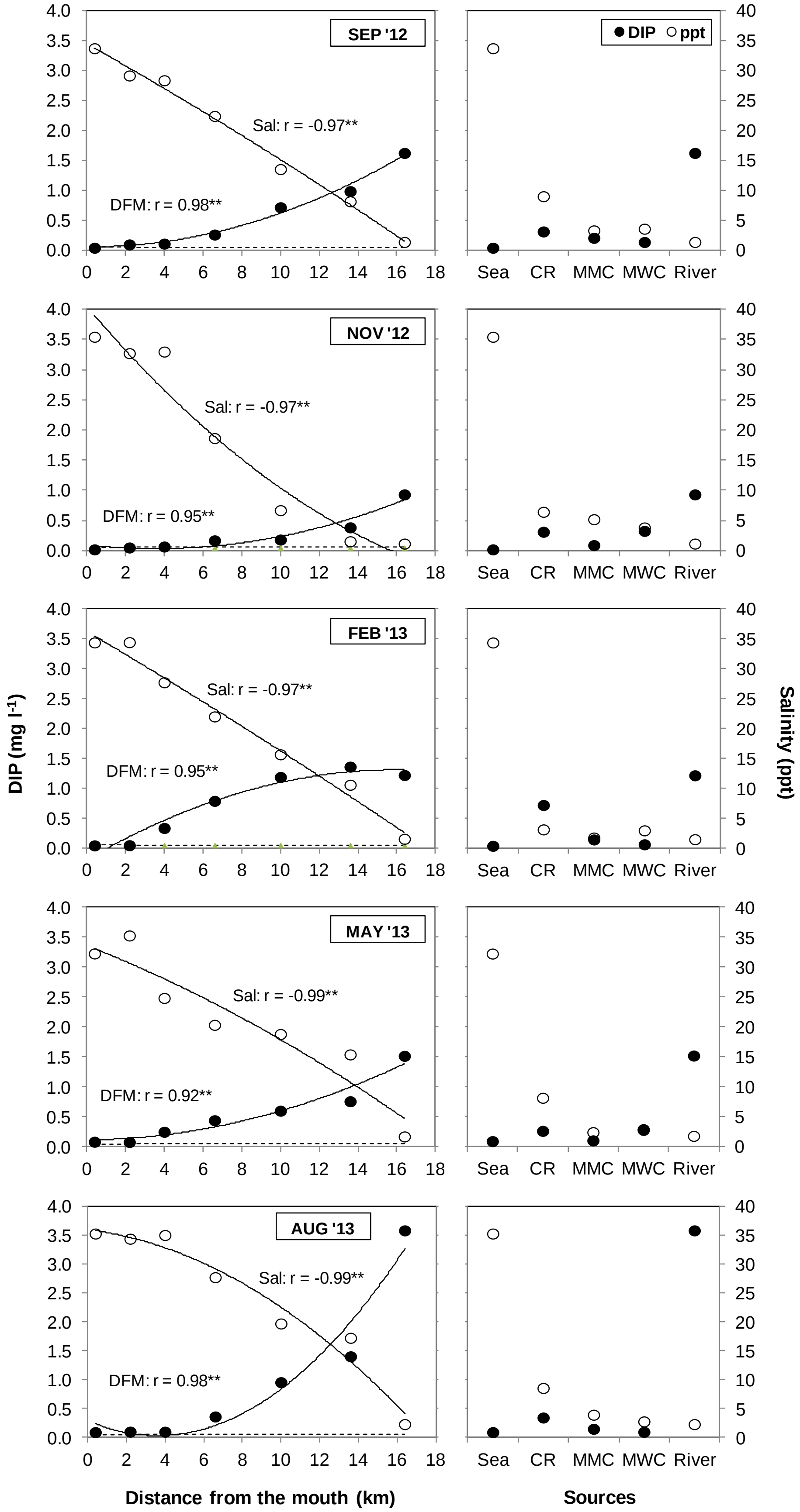


Figure 29: Vertically averaged dissolved inorganic phosphorus concentration (DIP; mg l^{-1}) recorded in the estuary channel and at points of entry into estuary. (Mean ± SE, "Sal" = salinity, "DFM" = distance from the mouth, "r" = correlation with Sal or DFM, ** indicates $p < 0.05$).

Table 23: Dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) concentrations recorded at Nivens Bridge between 1995 and 2013.

	DIN (mg l^{-1})	DIP (mg l^{-1})
n (data points)	343	357
Mean	1.20	0.16
SE	0.09	0.02
Median	0.57	0.04
Minimum	0.04	0.01
Maximum	14.40	3.08
Annual variability	S***	S***
Monthly variability	S***	NS
Trend (r)	0.28	0.51

Note: *S*, $p < 0.05$; S**, $p < 0.01$; S***, $p < 0.001$; $\alpha = 0.05$. "r" is calculated based on annual median values and refers to linearity with time. Shaded values indicate $p < 0.05$).*

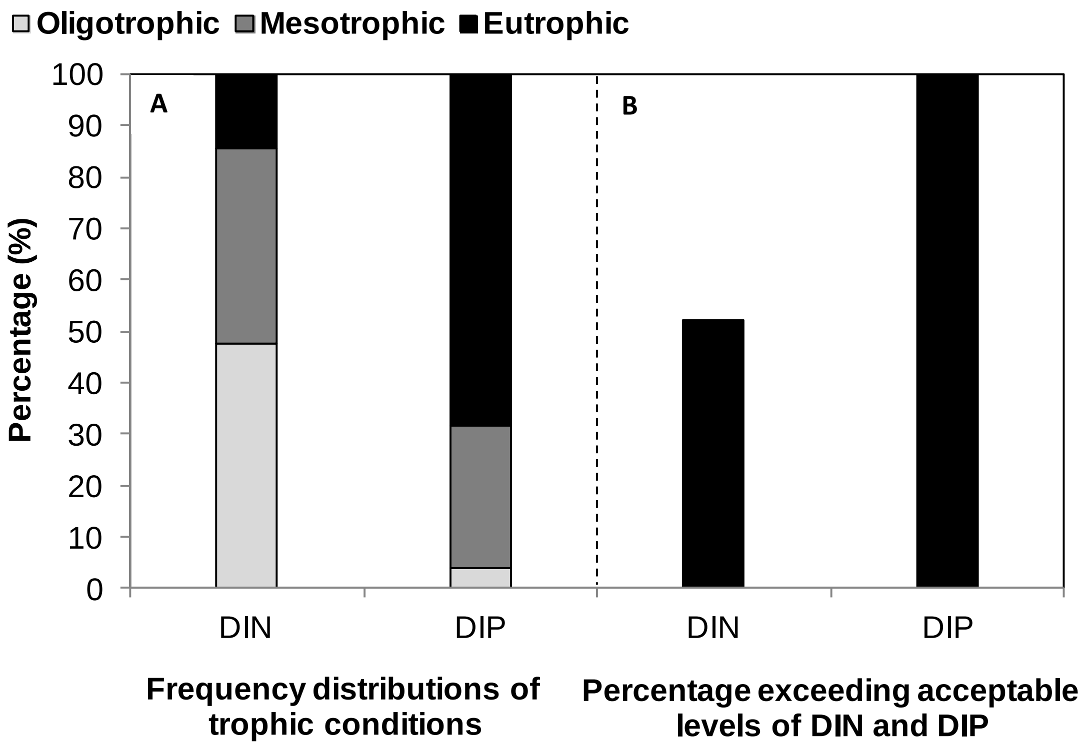


Figure 30: Frequency distributions of dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) trophic states in the Swartkops River (a) and the percentage of DIN and DIP measurements exceeding acceptable levels in aquatic ecosystems.

To identify the source of inorganic nutrients in the Swartkops Estuary, DIN (Figure 31) and DIP (Figure 32) loads discharged from WWTW between 2009 and 2013 were examined and related to concentrations measured in the river and estuary. DIN concentrations increased steadily from the mouth of the estuary to the tidal limit. DIN ranges at the freshwater sites were higher than those recorded within the estuary. The results indicated that DIN was elevated at sites located within close proximity to WWTW. Considering all monitoring sites within the river reaches, the highest DIN concentrations were measured at Van Schalkwyk Bridge (VSB; 6.6 mg l^{-1}, 0.6 – 18.9 mg l^{-1}), which is located downstream of the Kelvin Jones WWTW discharge site. From the median DIN concentrations measured from 2009 to 2013, it was estimated that the daily load determined for each of the WWTW ranged from 35.5 to 170.0 kg DIN d^{-1}. The highest median daily load was discharged from the KwaNobuhle WWTW (170.0 kg DIN d^{-1}, 16.0 – 622.4 kg DIN d^{-1}), followed by Kelvin Jones (152.6 kg DIN d^{-1}, 28.5 – 1 167.9 kg DIN d^{-1}) and then Despatch (35.5 kg DIN d^{-1}, 7.4 – 81.1 kg DIN d^{-1}).

Dissolved inorganic phosphorus, like DIN, increased steadily from the mouth (0 – 0.1 mg l^{-1}, 0.1 mg l^{-1}) to the head of the Swartkops Estuary (Perseverance: 0.2 – 8.2 mg l^{-1}, 1.6 mg l^{-1}) (Figure 9). At the freshwater monitoring sites DIP ranges were highest at the Van Schalkwyk Bridge (0.1 – 10.8 mg l^{-1}, 4.1 mg l^{-1}), followed by Perseverance Bridge (0.1 – 9.1 mg l^{-1}, 3.4 mg l^{-1}), and Elands River (0 – 3.6 mg l^{-1}; > 0.1 mg l^{-1}). Furthermore, it was found that the daily DIP load determined for each of the WWTW ranged from 13.4 to 128.6 kg DIP d^{-1} and that the highest median daily load was discharged from the Kelvin Jones WWTW (128.6 kg DIP d^{-1}, 0.9 – 826.1 kg DIP d^{-1}), followed by KwaNobuhle (46.4 DIP d^{-1}, 6.6 – 130.5 kg DIP d^{-1}), and then Despatch (13.4 kg DIP d^{-1}, 0.2 – 53.2 kg DIP d^{-1}).

4.3.4. Trace metals

Conclusions regarding the spatial and temporal variability in trace metals require consistent monitoring (under similar hydrological conditions) to detect trends. Several aspects of historical data on trace metals restricted temporal analyses, these included: limited data and single sample measurements, poor sensitivity of laboratory techniques resulting in metals that went undetected (i.e. laboratory detection limits for some metals were higher than water quality guidelines) and in some instances, both dissolved (SRK Consulting (Pty) Ltd, 2011) and total (this study; November 2012) metals were measured for the same metal. This resulted in data sets that were not comparable to historical data and to compliance criteria, i.e. water quality guidelines for trace metals in coastal marine waters (DWAF, 1995) which refer to total metals only and not to dissolved metal concentrations.

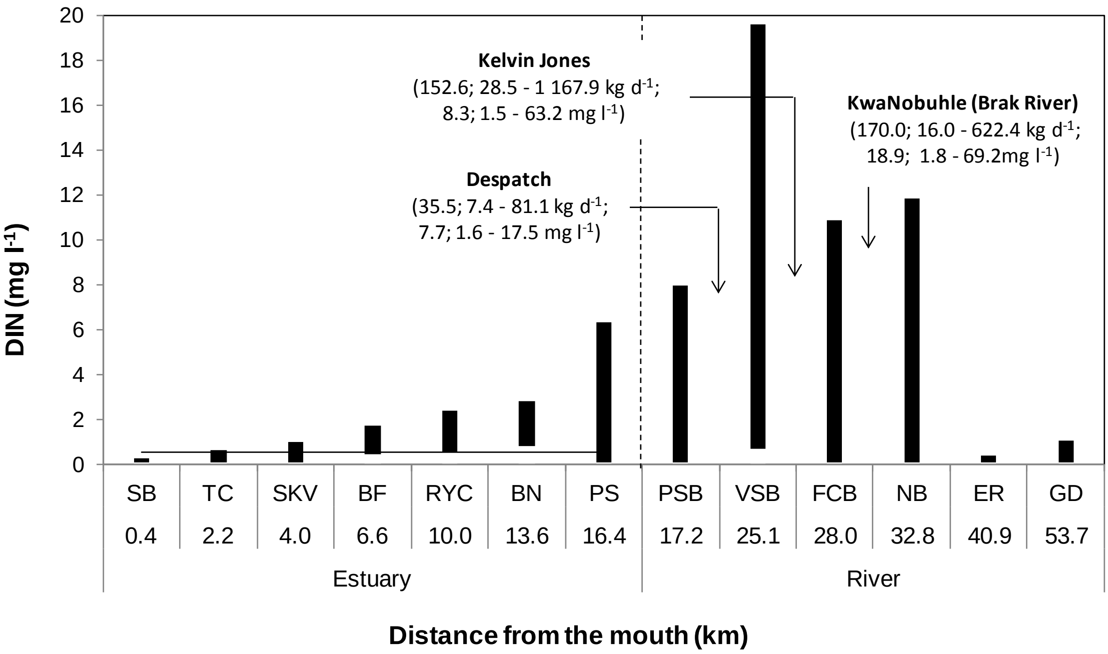


Figure 31: Minimum and maximum concentrations of dissolved inorganic nitrogen (DIN) measured in the Swartkops Estuary and river in relation to concentrations measured at the wastewater treatment works from 2009 to 2013. DIN discharged is expressed as median, minimum and maximum values. (The horizontal line denotes the guideline threshold for DIN in estuarine systems).

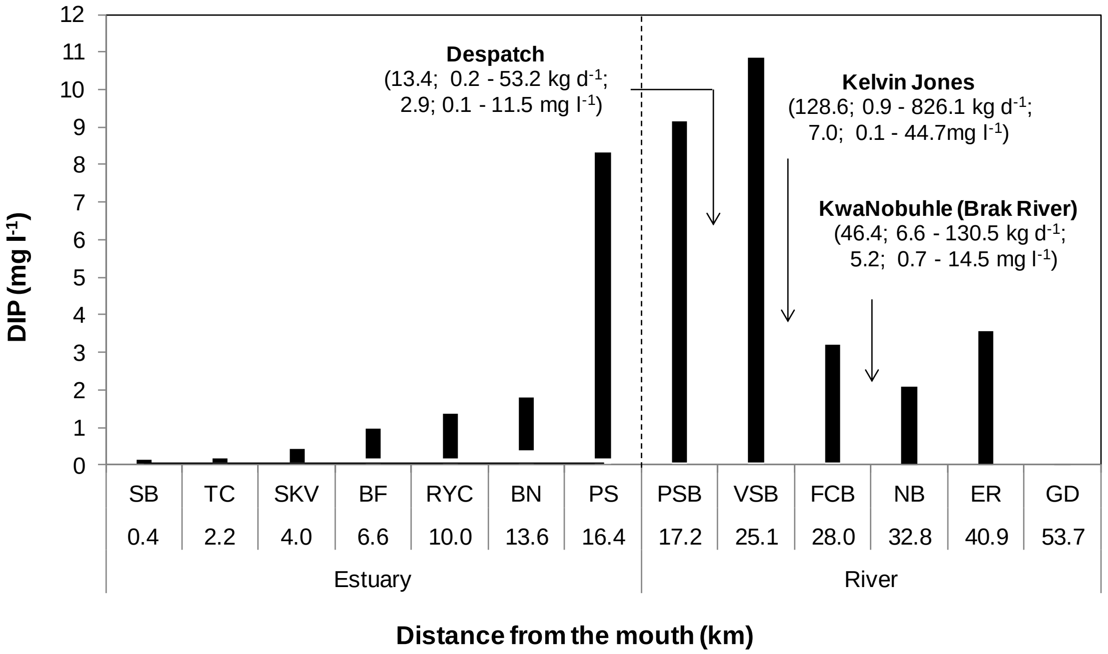


Figure 32: Minimum and maximum concentrations of dissolved inorganic phosphorus (DIP) measured in the Swartkops Estuary and river in relation to concentrations measured at the wastewater treatment works from 2009 to 2013. DIP discharged is expressed as median, minimum and maximum values. (The horizontal line denotes the guideline threshold for DIP in estuarine systems).

Past and present trace metal concentrations are provided in Figure 33, and all other metals for which historical measurements were not available for comparison are provided in Appendix D Table 42. Only copper, lead, zinc, iron and cadmium concentrations recorded in the present study could be compared to those measured by Watling and Watling (1982). Since the data constituted only two monitoring studies (i.e. Watling and Watling, 1982 and this study), the degree of confidence was not strong. Nevertheless, the study by Watling and Watling (1982) measured low levels of copper (3.2 ± 1.03 μg l^{-1}, 0.1 – 5.0 μg l^{-1}), lead (1.2 ± 0.35 μg l^{-1}, 0.4 – 2.5 μg l^{-1}), zinc (3.2 ± 1.03 μg l^{-1}, 1.6 – 7.2 μg l^{-1}), iron (142.0 ± 34.30 μg l^{-1}, 43 – 256 μg l^{-1}) and cadmium (0.2 ± 0.04 μg l^{-1}, 0.1 – 0.3 μg l^{-1}) in the estuary and elevated levels of iron in the Markman (314 μg l^{-1}) and Motherwell (305 μg l^{-1}) canals. Concentrations of iron and cadmium were not considered safe for recreational use of the estuary (i.e. intake of water should not exceed “200 ml per day, that is, 100 ml per recreational session with two sessions per day”), whereas levels of copper, zinc, and cadmium exceeded acceptable levels of trace metals in coastal aquatic ecosystems. Total chromium and dissolved selenium were highlighted as regarded as of concern, requiring further investigation to confirm their levels of toxicity in the estuary. Total chromium (130 μg l^{-1}) exceeded guidelines at Tippers Creek, rendering the water not suitable for recreation and for coastal aquatic life. Dissolved selenium concentrations exceeded guideline levels from the mouth to Swartkops Village. Elevated concentrations of total chromium were measured in the Markman Canal (140 μg l^{-1}) and Motherwell Canal (130 μg l^{-1}). Overall, this study found that all trace metal concentrations recorded in the estuary at the tidal limit at Perseverance and in the Motherwell Canal increased by approximately 93% from 1977 to 2012. Mercury levels in the estuary and at the sites of entry into the estuary have remained undetected since the study by Watling and Watling (1982). In the present study lead was undetectable at all sites.

4.3.5. Phytoplankton biomass (chlorophyll-*a*)

a) Short-term spatial and temporal study (2012 – 2013)

Chlorophyll-*a* showed an inverse relationship with salinity ($r = -0.83$; $p < 0.05$; $n = 60$) (see Appendix D: Table 43), therefore, the upper reaches of the estuary (between Redhouse Yacht Club and Perseverance) was the most productive region. Chlorophyll-*a* concentrations were significantly different between sites ($F = 13.7$; df = 6; $p < 0.05$; $n = 35$), with concentrations ranging from 0 to 248 μg l^{-1} (31.8 ± 6.56 μg l^{-1}). Significant temporal differences ($F = 5.3$; df = 4; $p < 0.05$; $n = 35$) were also noted, with the highest concentration recorded in November 2012 (58.3 ± 25.23 μg l^{-1}) on a neap flood tide, and lowest in August 2013 (9.2 ± 4.10 μg l^{-1}), also on a neap flow tide.

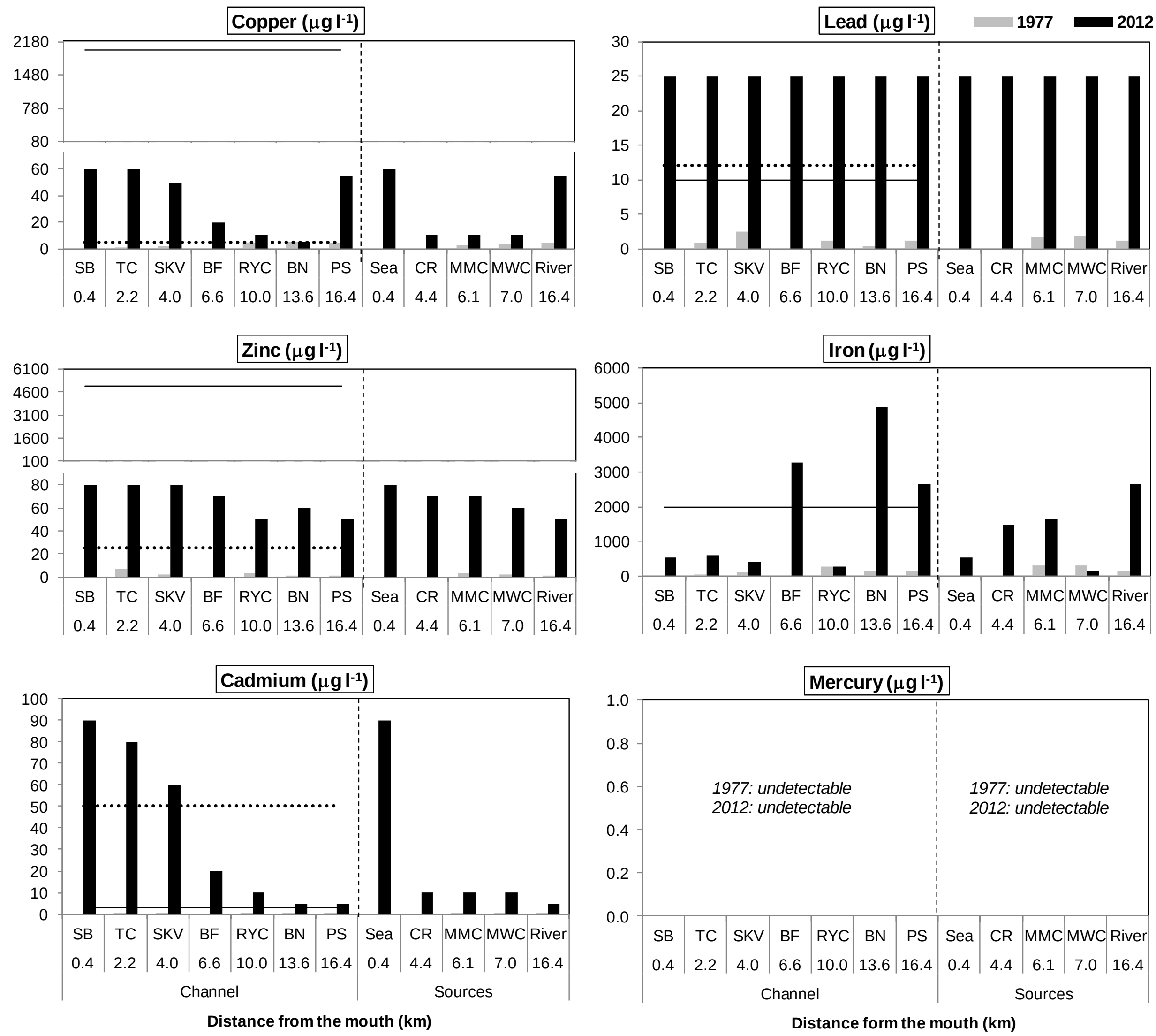


Figure 33: Heavy metal concentrations (µg l^{-1}) in the water column of the sampling sites within the estuary and at points of entry into the estuary. (Solid horizontal lines denote guideline values for acceptable levels of trace metals for recreation and broken horizontal lines denote guidelines coastal marine ecosystems).

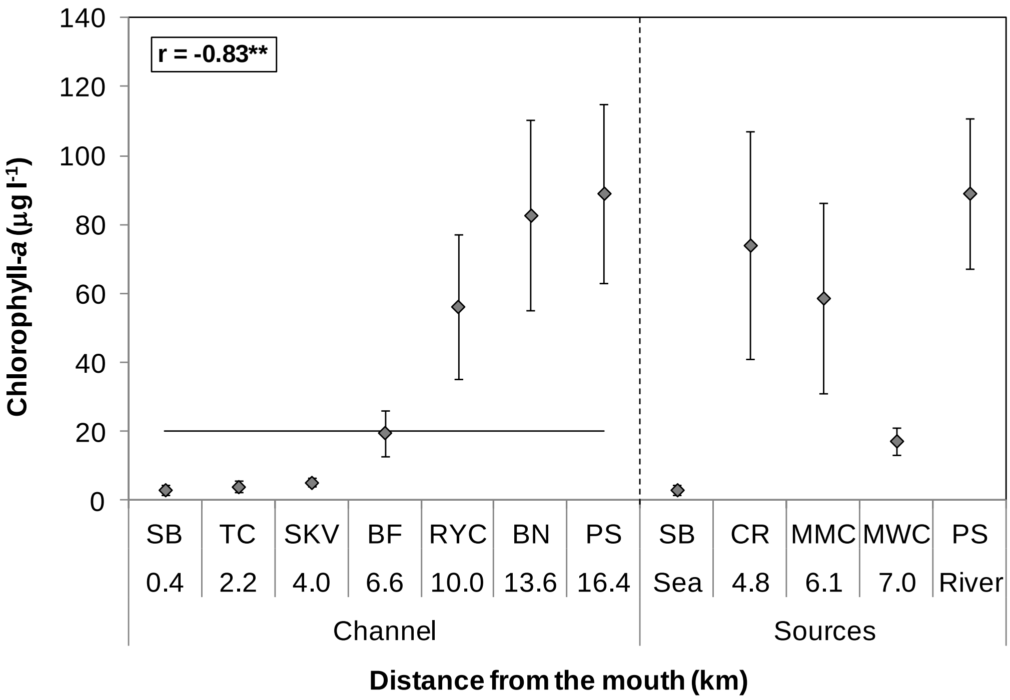


Figure 34: Spatial variation in chlorophyll-*a* concentrations (µg l^{-1}) measured within the estuary channel and at points of entry into the estuary. (Mean ± SE, "r" refers to linear correlation with salinity, ** indicates $p < 0.05$. The horizontal line refers to the chlorophyll-*a* concentration at which phytoplankton blooms occur).

There were no apparent effects of tidal stage and flow regime on chlorophyll-*a* in the estuary, since the highest concentrations were recorded on a neap flood tide in November 2012 (58.3 ± 25.23 µg l^{-1}) and also on a spring ebb tide in February 2013 (52.7 ± 20.19 µg l^{-1}), with flow conditions of 2.14 and 0.50 m^3 s^{-1} respectively. Chlorophyll-*a* was positively correlated with NH_4^+ ($r = 0.51$; $p < 0.05$), TOxN ($r = 0.51$; $p < 0.05$), DIN ($r = 0.57$; $p < 0.05$) and DIP ($r = 0.67$; $p < 0.05$), and negatively correlated with salinity ($r = -0.83$; $p < 0.05$). Furthermore, a significant relationship between chlorophyll-*a* concentrations and temperature ($r = 0.54$; $p < 0.05$) was apparent and a significant inverse relationship between surface chlorophyll-a with water clarity ($r = -0.69$; $p < 0.05$; $n = 25$). The latter indicated that water clarity was poorest in waters where phytoplankton biomass was the highest. The present study has shown that the high nutrient concentrations are consistently supporting a phytoplankton biomass of bloom concentrations (> 20 µg l^{-1}) from the middle to upper reaches of the estuary, which indicates a deterioration in water quality over time.

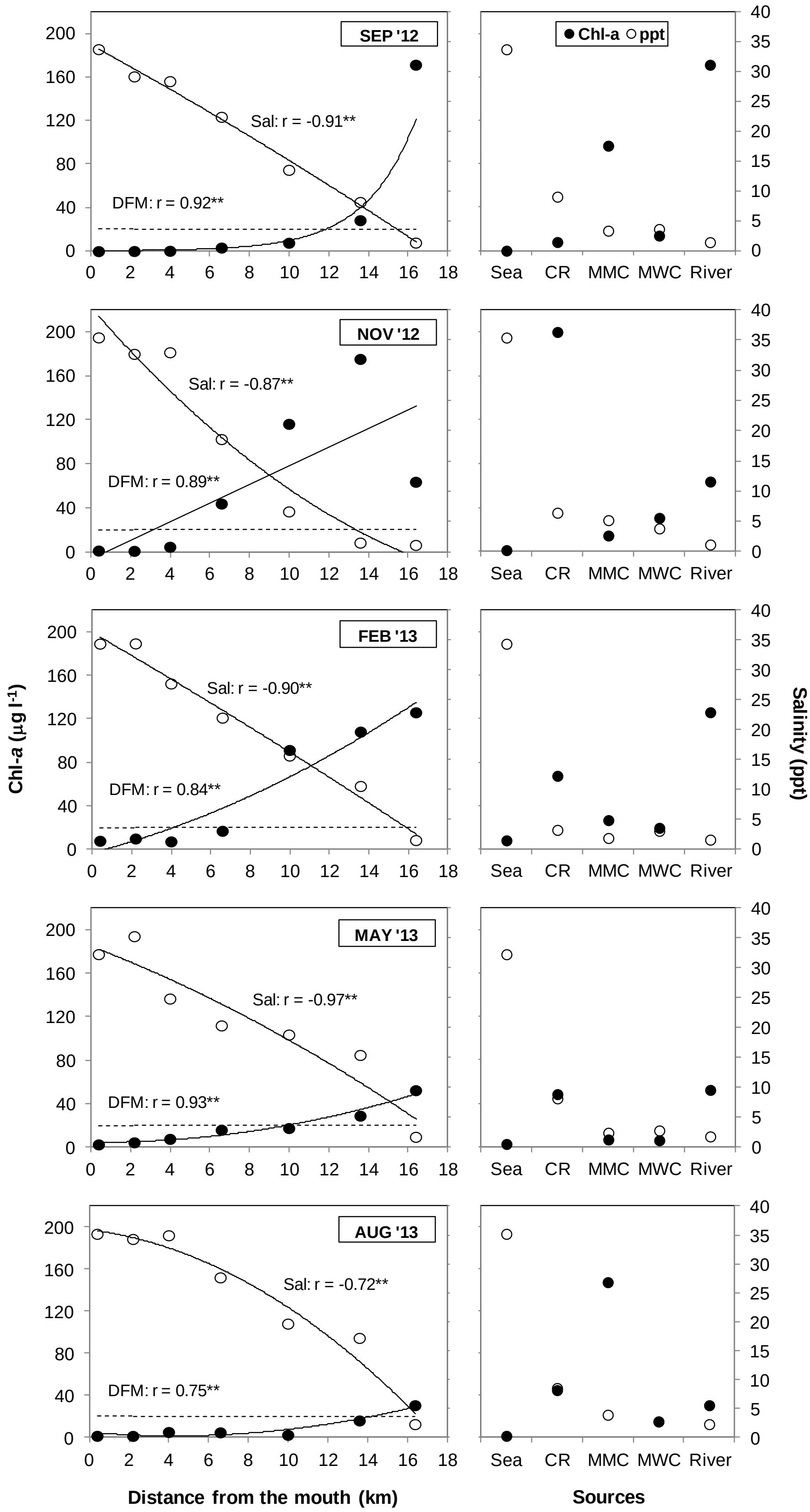


Figure 35: Vertically averaged chlorophyll-*a* concentrations (µg l^{-1}) measured along the length of the estuary and at points of entry into the estuary. (Mean ± SE; “Sal” = salinity, “DMF” = distance from the mouth, “r” = correlation with Sal or DFM, ** indicates $p < 0.05$. The horizontal line refers to the chlorophyll-*a* concentration at which phytoplankton blooms occur).

b) *Comparison with past data*

Historical data showed that there has been an increase in phytoplankton biomass, particularly in the middle to upper reaches of the estuary (Table 24). Hilmer (1984), Scharler *et al*. (1997) and Adams and Bate (1994) also concluded that the most productive region of the Swartkops Estuary is the upper reaches, with chlorophyll-*a* concentrations decreasing from the tidal limit to the mouth. The poor relationship between chlorophyll-*a* and freshwater flow that was observed in the present study were consistent with findings of Binning (1999) and Scharler *et al*. (2003). Hilmer (1984) found that of all the nutrients, fluctuations in phytoplankton were most responsive to changes in ammonium; whereas in the present study, this was true for DIP. Moreover, Scharler *et al*. (1997) found no relationship between salinity and chlorophyll-*a*, whereas the present study showed positive relationships between the two parameters. With turbidity levels having increased since the study by Scharler *et al*. (1997), with particularly high turbidity levels recorded in November 2012 (Figure 19), it is suggested that turbidity played a major role in primary production during this month of the study. In support of this, Scharler *et al*. (1997) found that an increase in turbidity was attributed to an increase in chlorophyll-*a* levels of approximately 53%, in addition to higher DIP and lower NH_4^+ concentrations.

4.3.6. Phytoplankton community composition

Spatial and temporal variability between phytoplankton groups are presented in Figure 36 and phytoplankton cell density in Table 25. Species that could not be identified were labelled “unknown” and cell density greater than 10 000 cells ml^{-1} indicated phytoplankton blooms. In the present study, phytoplankton blooms occurred from Swartkops Village to the tidal limit of the estuary. Blooms consisted of flagellates (September 2012, February 2013, May 2013 and August 2013), diatoms (September and November 2012) and dinoflagellates (only at one site in September 2012).

During a spring ebb tide in September 2012, flagellates (379 – 4 759 cells ml^{-1}) were the dominant group from the lower to the middle reaches of the estuary in the vicinity of Brickfields, whereas diatoms (257 – 27 126 cells ml^{-1}), dinoflagellates (0 – 15 885 cells ml^{-1}) and flagellates (557 – 14 864 cells ml^{-1}) were recorded in the upper reaches between Redhouse Yacht Club and Perseverance. In November 2012, elevated freshwater inflow introduced diatom cells (1 393 – 47 378 cells ml^{-1}) into the upper reaches of the estuary between Perseverance and Swartkops Village, whereas further downstream flagellates (929 – 1 217 cells ml^{-1}) were dominant between Settlers Bridge and Tippers Creek. During a spring ebb tide in February 2012, several phytoplankton groups occurred; diatoms and flagellates in the lower reaches between Settlers Bridge and Swartkops Village; diatoms, flagellates and chlorophytes between Brickfields to Redhouse Yacht Club; and dinoflagellates at Tippers Creek.

Table 24: Mean chlorophyll-*a* concentrations ($\mu g\ l^{-1}$) recorded in the past and in the present.

Site name and distance from the mouth [km]	Scharler *et al.* (1997)	Binning (1999)	This study (depth averaged)
Settlers Bridge [0.4 km]	4.1	22.5	2.9
Swartkops Yacht Club [4.0 km]	-	18.2	5.2
Brickfields [6.6 km]	6.7	-	17.2
Redhouse Yacht Club [10 km]	-	24.8	47.3
Bar None [13.6 km]	8.6	20.6	71.6
Perseverance [16.4 km]	22.3	23.7	89.0

In May and August 2013, flagellates were the dominant group in the estuary. The diatoms were significantly higher in the surface water compared to the bottom ($p < 0.05$; $n = 25$), and most pronounced during phytoplankton blooms. Cell densities were almost consistenly lower at the tidal limit of the estuary, suggesting that the phytoplankton production occured *in situ* and were not necessarily introduced to the estuary from the riverine reaches. Worth noting was the occurrence of *Euglena* cells (4 394 cells ml^{-1}) at the Perseverance in May 2013 and in the Markman Canal (19 230 cells ml^{-1}) in August 2013. Their presence appeared to coincide with high concentrations of TOxN and DIN; Perseverance (TOxN: 5.1 mg l^{-1} and DIN: 5.2 mg l^{-1}) and Markman Canal (TOxN: 3.8 mg l^{-1} and DIN: 4.4 mg l^{-1}).

Correlation analyses (see Appendix D: Table 43) showed several significant relationships between water quality characteristics and phytoplankton group responses. For example, both diatoms ($r = 0.71$; $p < 0.05$; $n = 60$) and chlorophytes ($r = 0.56$; $p < 0.05$; $n = 60$) indicated a preference for warmer waters riverine water. Diatoms and dinoflagellates were positively associated with elevated DO levels ($r = 0.28$, $p < 0.05$, $n = 60$; $r = 0.36$, $p < 0.05$, $n = 60$; respectively) and negatively associated with TSS ($r = -0.36$, $p < 0.05$, $n = 60$; $r = -0.34$, $p < 0.05$, $n = 60$; respectively). Flagellates were the only group to correlate positively and significantly ($p < 0.05$) with all nutrients (NH_4^+, $r = 0.42$; TOxN, $r = 0.43$; DIN, $r = 0.44$; DIP, $r = 0.49$; $n = 60$), whereas chlorophyte cells only showed a significant positive relationship with NH_4^+ ($r = 0.38$; $p < 0.05$; $n = 60$). Dinoflagellates, a significant inverse relationship with NH_4^+ ($r = -0.41$; $p < 0.05$; $n = 60$). The data further indicated that elevated chlorophyll-*a* levels were ascribed to flagellates ($r = 0.44$; $p < 0.05$; $n = 60$), diatoms ($r = 0.59$; $p < 0.05$; $n = 60$) and chlorophytes ($r = 0.53$; $p < 0.05$; $n = 60$).

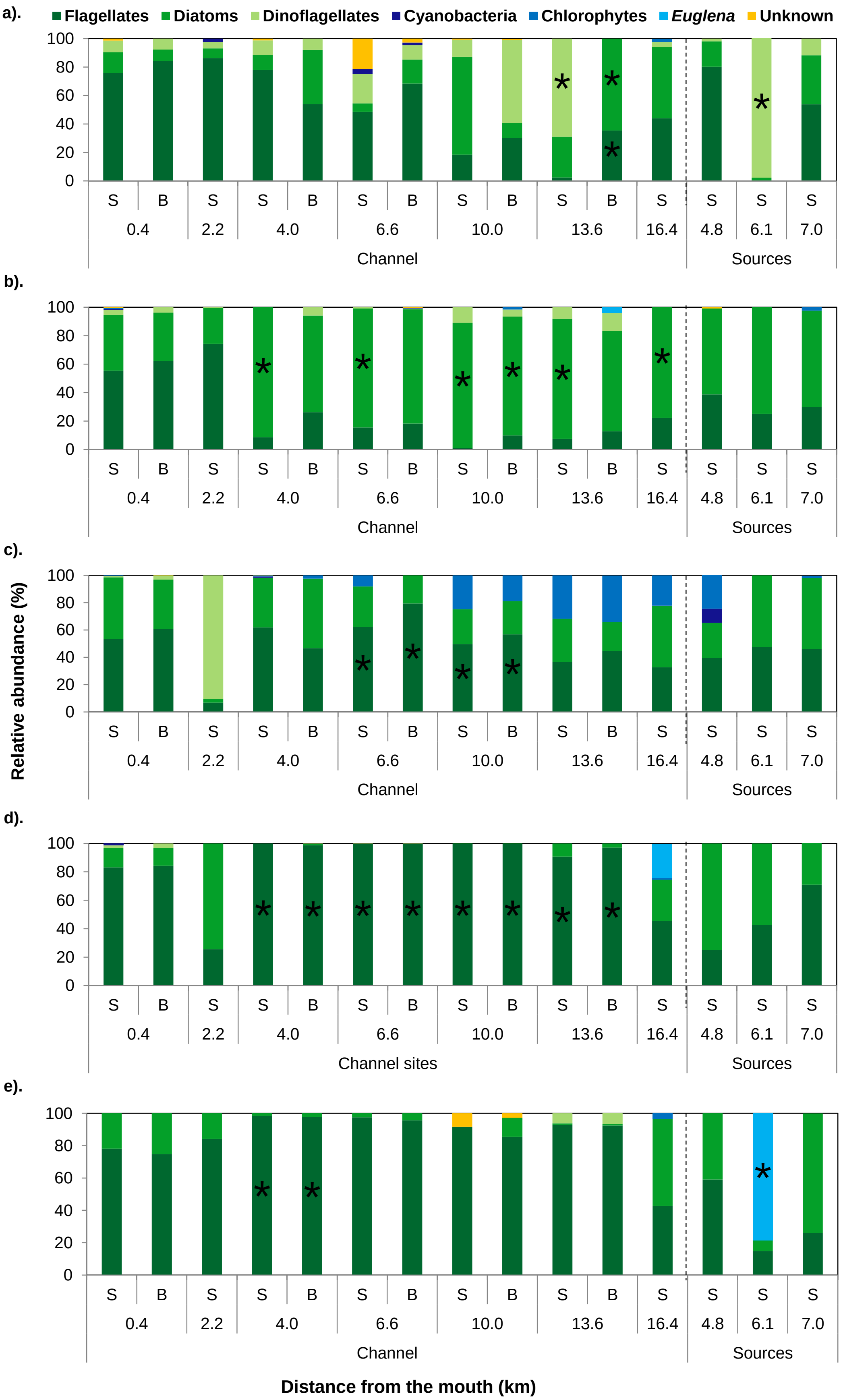


Figure 36: Phytoplankton community composition (relative abundance) recorded along the length of the estuary and at points of entry into the estuary in September 2012 ('a'), November 2012 ('b'), February 2013 ('c'), May 2013 ('d') and August 2013 ('e'). (The asterisks indicate bloom-forming species [chlorophyll-*a* > 20 $\mu g\ l^{-1}$]).

Table 25: Dominant phytoplankton groups recorded along the length of the estuary and at points of entry into the estuary. (Bold indicates blooms [chlorophyll-*a* > 20 µg l^{-1}]).

	SEP '12	NOV '12	FEB '13	MAY '13	AUG '13
Surface water					
Settlers Bridge [0.4 km]	1 483	1 203	734	1 610	2 628
Tippers Creek [2.2 km]	2 180	929	743	614	790
Swartkops Village [4.0 km]	708	**10 646**	1 505	**41 246**	**11 148**
Brickfields [6.6 km]	1 125	**18 115**	**13 935**	**42 027**	2 601
Redhouse Yacht Club [10.0 km]	2 516	**32 328**	**13 006**	**43 364**	2 369
Bar None [13.6 km]	**15 885**	**47 378**	6 812	**46 820**	2 637
Perseverance [16.6 km]	2 070	**11 259**	4 579	8 175	4 857
Bottom water					
Settlers Bridge [0.4 km]	1 525	1 217	665	2 898	1 779
Swartkops Village [4.0 km]	379	1 393	1 440	**25 640**	**11 148**
Brickfields [6.6 km]	1 240	8 361	**11 457**	**28 427**	1 889
Redhouse Yacht Club [10.0 km]	1 393	**25 640**	**10 750**	**46 263**	689
Bar None [13.6 km]	**14 864** **27 126**	6 903	5 827	**16 164**	2 588
Sources					
Chatty River [4.8 km]	1 634	2 162	6 967	9 290	4 459
Markman Canal [6.1 km]	**13 006**	33	232	470	**19 230**
Motherwell Canal [7.0 km]	4 756	6 848	7 190	9 197	7 757

Note: refer to the legend in Figure 40 for phytoplankton groups based according to the colour scheme. Values in bold refer to bloom conditions (> 10 000 cells ml^{-1}).

Key	Flagellates	Diatoms	Dinoflagellates	*Euglena*

Although bottom waters of anthropogenically impacted areas can become anoxic, rapid growth of phytoplankton in the overlying nutrient-rich waters can also generate substantial concentrations of oxygen (Kennish, 1997). In the present study, water quality conditions recorded in February and May 2013 corroborate this scenario; where as mentioned previously, hypoxic (< 3 mg l^{-}) conditions developed at Brickfields and Bar None in February and at Bar None in May (see Appendix D: Table 45). Hypoxic conditions (1.30 mg l^{-1}) recorded at Bar None were associated with a surface-to-bottom difference in DO of 8.63 mg l^{-1}. In the surface water, this was associated with eleveated NH_4^+ (1.46 mg l^{-1}), DIN (2.31 mg l^{-1}) and DIP (1.41 mg l^{-1}) and an increase phytoplankton biomass (158.1 µg l^{-1}). As neither nitrogen nor phosphorus were in short-supply, it was suggested that these conditions gave rise to surface waters which were populated by relatively similar cell densities of flagellatess (6 812 cells ml^{-1}), diatoms (5 822 cells ml^{-1}) and chlorophyte (5 883 cells ml^{-1}).

Not previously reported in literature, is the co-existance of several phytoplankton groups in the Swartkops Estuary. The present study found phytoplankton community richness increases during spring ebb tides, and that nutrient availability further defines the co-existance of different phytoplankton groups. It was observed that lower ammonium and phosphate concentrations and higher TOxN concentrations favoured the co-existance of flagellates, diatoms and dinoflagellates, whereas, higher ammonium and phosphate concentrations and lower TOxN concentrations favoured the co-existance of dinoflagellates, diatoms and chlorophytes.

Overall, the study found that a spring ebb tide allowed several phytoplankton groups to co-exist; whereas increased freshwater flow (2.14 m^3 s^{-1}), on a neap flood tide supported a bloom of diatom cells, and low flow (0.22 – 0.31 m^3 s^{-1}) conditions, also on a neap flood tide, generally supported flagellate cells. Chlorophyte cells were only recorded in February 2013 when elevated levels of NH_4^+ and DIP were found and when conditions within the water column were relatively stable (i.e. during the slack waters of the spring ebb tide) and when river flow was low.

4.3.7. Faecal bacteria

a) Short-term spatial and temporal study (2012 – 2013)

Escherichia coli and enterococci counts are provided in Figure 37 and Figure 38 respectively. Log transformed counts of *E. coli* ($r = 0.52$; $p < 0.05$; $n = 35$) and enterococci ($r = 0.32$; $p > 0.05$; $n = 35$) increased with distance from the mouth of the estuary. This trend can be attributed to stormwater discharge that was high in faecal matter from the Motherwell Canal (MWC) and from the Swartkops River, which receives sewage effluent from three wastewater treatment plants. In the estuary, *E. coli* counts ranged from 0 to 42 000 counts 100 ml^{-1} and were significantly variable between months ($F = 5.7$; $df = 4$; $p < 0.05$; $n = 25$) and between sites ($F = 5.6$; $df = 6$; $p < 0.05$; $n = 25$), with the highest concentration measured at Brickfields (8 589; 25 – 42 000 counts 100 ml^{-1}), and Perseverance (832; 51 – 2 800 counts 100 ml^{-1}) and the lowest at Settlers Bridge (18; 0 – 47 counts 100 ml^{-1}). Counts measured in the Chatty River (1 500; 118 – 3 800 counts.100 ml^{-1}) and Markman Canal (889; 12 – 3 999 counts 100 ml^{-1}) were found to be similar. *Escherichia coli* counts were generally highest in November 2012 (0 – 42 000 counts 100 ml^{-1}) and lowest in September 2012 (0 – 51 counts 100 ml^{-1}).

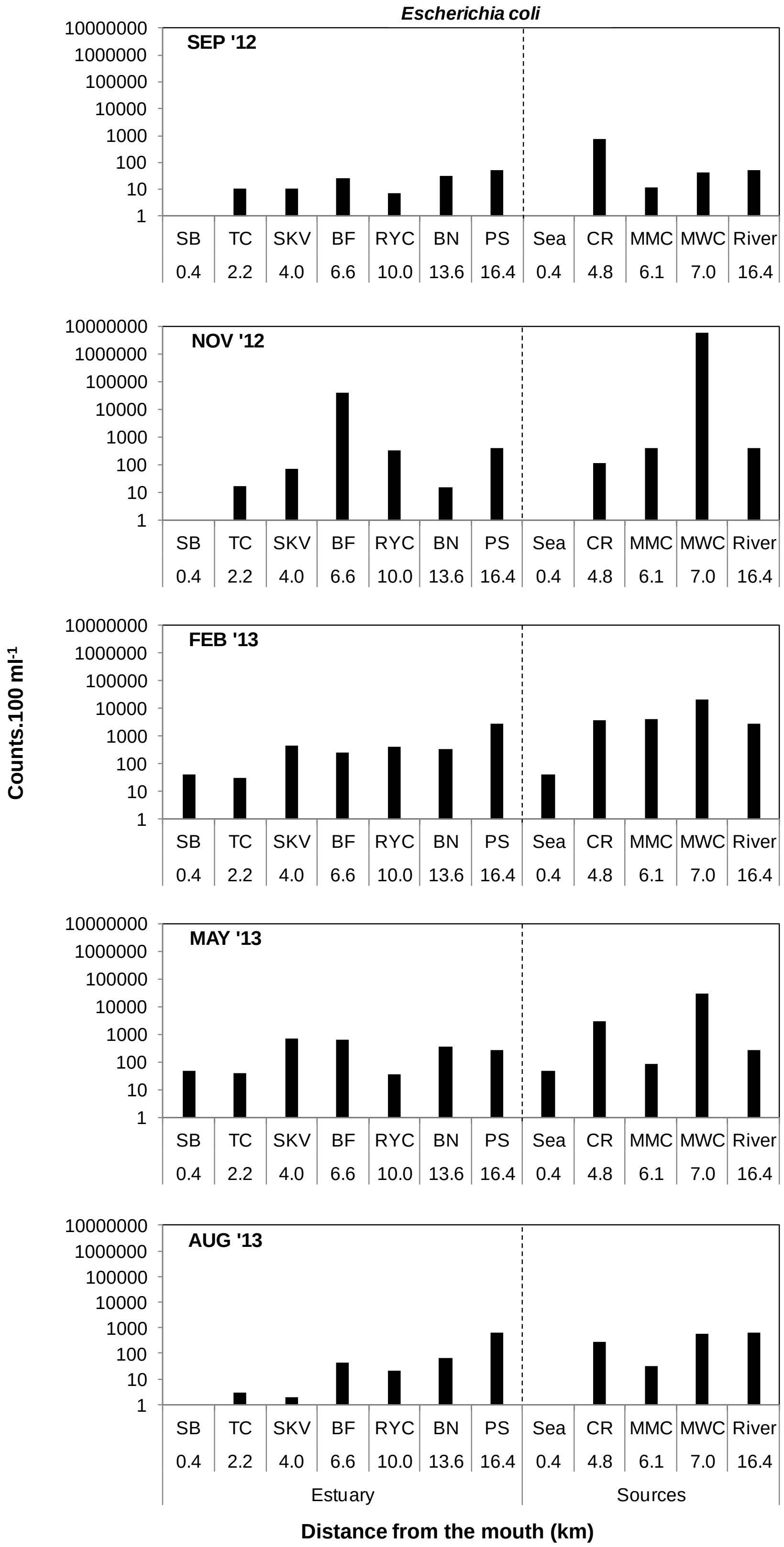


Figure 37: Spatial and temporal variability in *Escherichia coli* counts measured along the length of the estuary and at points of entry into the estuary.

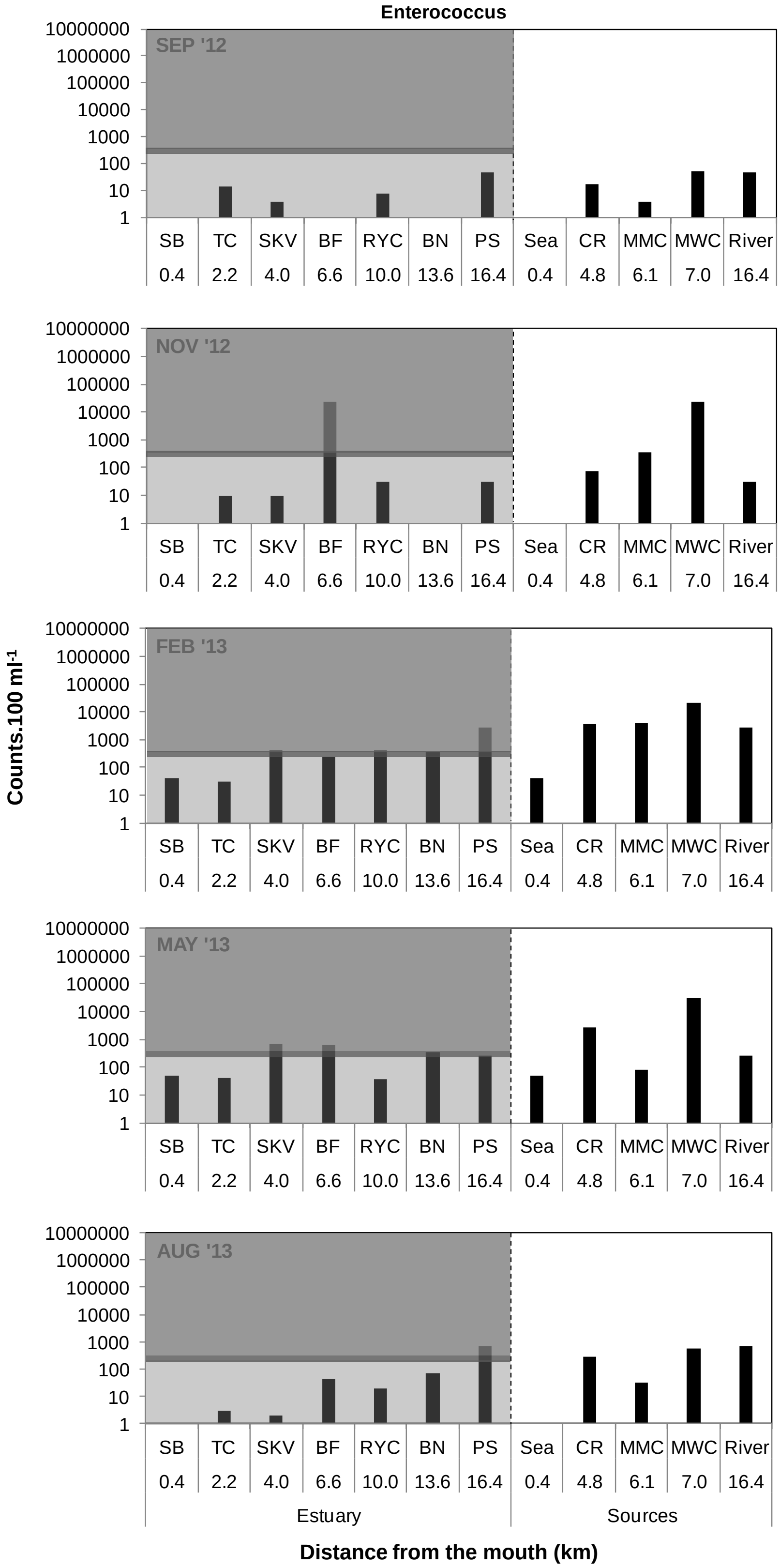


Figure 38: Spatial and temporal variability in enterococci counts measured along the length of the estuary and at points of entry into the estuary. (The shaded areas refer to the single value target values: < 240 counts 100 ml^{-1}, > 240 counts 100 ml^{-1} and > 380 counts 100 ml^{-1}).

The rise in *E. coli* counts recorded at Brickfields in November 2012 was associated with a pulse of flow from the Motherwell Canal (1.2 x 10^6; 44 – 5.8 x 10^6 counts 100 ml^{-1}), which marked the Motherwell Canal as a major source of faecal pollution to the estuary. Elevated *E. coli* levels at Swartkops Village in February and May 2013 were thought to originate from the Chatty River confluence with the estuary (0.8 km upstream from Swartkops Village) when faecal bacteria levels were the highest during the study. The Chatty River was thus a source of faecal pollution to the lower reaches of the estuary. Since 'wet-weather' conditions were not recorded in this study, comparisons could not be made with levels recorded during dry weather from runoff sources.

Enterococci counts (Figure 38) showed similar spatial and temporal patterns to that of *E. coli* measured in the estuary and at the sites which enter the estuary. The highest enterococci counts were recorded at Brickfields (4 888 counts 100 ml^{-1}; 0 – 24 196 counts 100 ml^{-1}) and Perseverance (349 counts 100 ml^{-1}; 30 – 1 500 counts 100 ml^{-1}). Enterococci counts were generally within acceptable limits (< 250 counts 100 ml^{-1}) in the lower and upper reaches of the estuary, except in November 2012 and February 2013, which was concomitant with elevated levels of *E. coli*. Several significant correlations were found between faecal bacteria levels and physico-chemical variables and nutrients (see Appendix D: Table 44). The data showed that *E. coli* measurements correlated positively to increased water column temperatures ($r = 0.34$; $p < 0.05$; $n = 35$) and low salinity ($r = -0.57$; $p < 0.05$; $n = 35$); thus confirming the effect of lower water column temperatures on the survival of *E. coli* in the waters. A high nutrient load in an aquatic environment with concomitant high bacteria counts is generally indicative of sewage pollution and in the present study this was illustrated by the significant ($p < 0.05$) correlation between all nutrients (see Appendix D: Table 44) and *E. coli* abundance, especially ammonium.

b) *Comparison with past data*

In a report produced by HKS (1974) it was concluded that the Swartkops River's self purification ability appears to be high and therefore "appears to be unaffected" by pollutant loads. However, it was only in the subsequent studies (Emmerson, 1985; Lord and Thompson, 1988; Lord and MacKay, 1993) that long-lasting concerns grew regarding this statement. Collated historical data indicated that the Swartkops River (Figure 39 and Figure 40) and the Motherwell Canal (Figure 39 and Figure 41) are still the two major sources of faecal bacteria to the estuary with counts ranging from 0 to 200 000 counts 100 ml^{-1} and from 0 to 40 x 10^{-6} counts 100 ml^{-1}, respectively and that the Chatty River is source of faecal bacteria to the lower estuarine reaches. The findings of the present study have been consistent with this trend, confirming bacteriological status of the estuary as poor. Figure 39 shows that *E. coli* counts are highest in places that are at the receiving end of wastewater treatment discharges. (Refer to Figure 31 and Figure 32 for discharge locations of WWTW in relation to sampling sites).

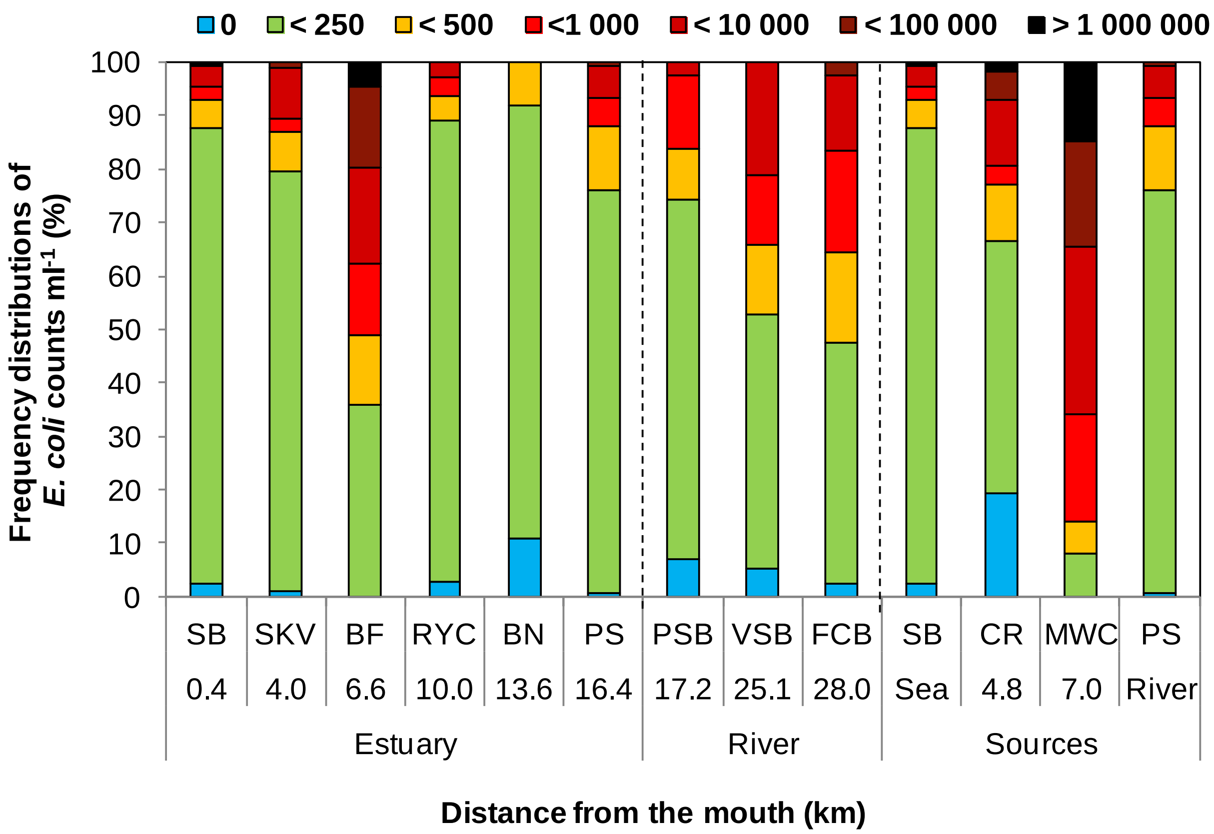


Figure 39: Spatial variability in frequency distributions of *Escherichia coli* counts recorded within the estuary from 1980 to 2013 in relation to counts recorded in the Swartkops River and at points of entry into the estuary.

This is because the Kelvin Jones WWTW discharges between the Frans Claasen and the Despatch Highway bridges, the Despatch WWTW discharges between the Despatch Road bridge and Perseverance and the KwaNobuhle WWTW discharges into the Brak River just above the river's confluence with the Swartkops River. The marked linear trend of decreasing *E. coli* levels with distance from the Frans Claasen Bridge to Bar None shows a clear dilution effect over a distance of approximately 14.4 km. However, this could also be due to mortality (osmotic shock due to salinity) and settling effects (Rozen and Belkin, 2001; De Brauwere *et al*., 2011).

Table 26 shows the results of a compliance assessment on *E. coli* data recorded from 1980 to 2013 for recreational water use. At Swartkops Village and Brickfields conditions have remained highly unsuitable for recreational water use (i.e. the water is of a "poor quality") since 1980. Compliance conditions at Redhouse Yacht Club have fluctuated since 1980, while in the Bar None area, water quality appears to have deteriorated. The bacteriological status of the water at Perseverance has remained poor since 1980. Sporadic increases in *E. coli* counts have rendered this region of the estuary unsafe for recreation, especially during the summer months.

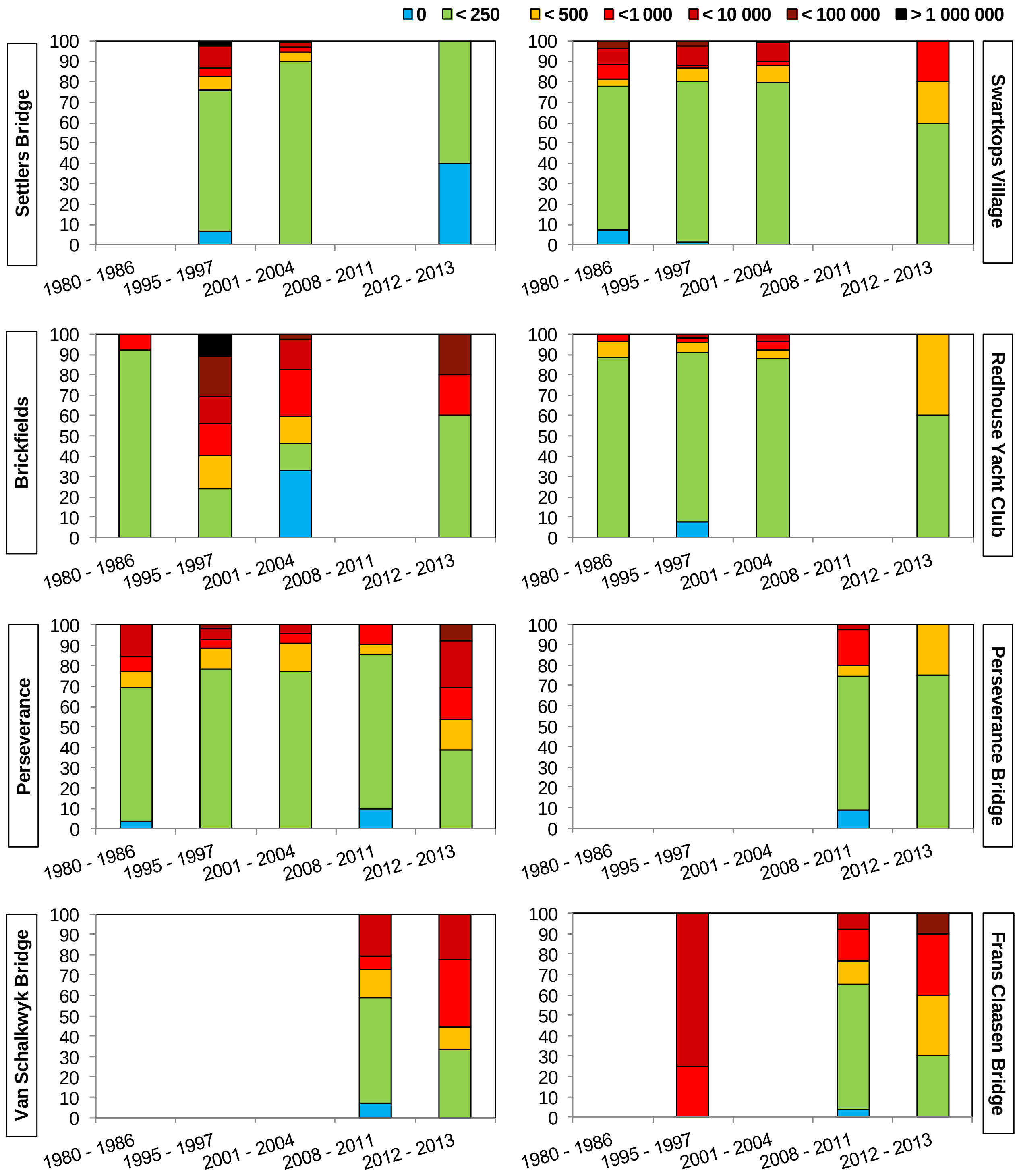


Figure 40: Spatial and temporal variability in *Escherichia coli* counts recorded in the Swartkops Estuary and in the Motherwell Canal from 1980 to 2013.

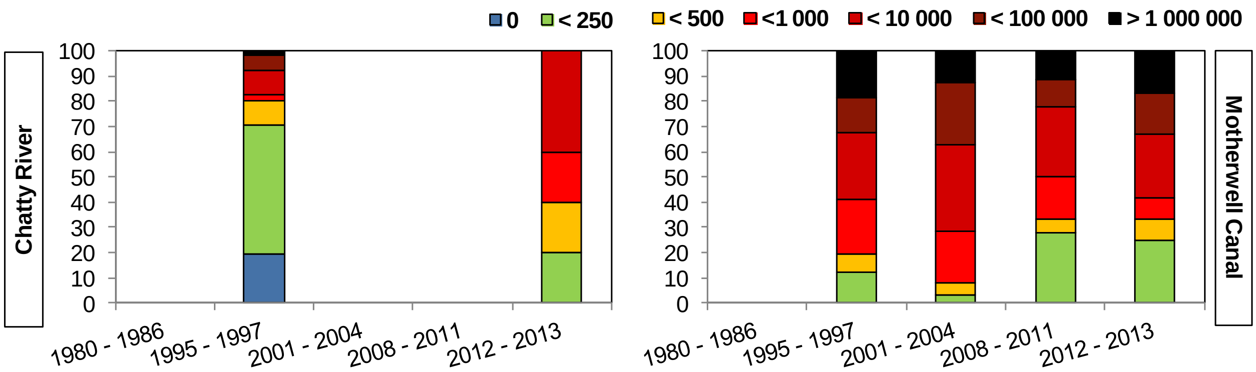


Figure 41: Spatial and temporal variability in *Escherichia coli* counts recorded in the Chatty River and in the Motherwell Canal from 1980 to 2013.

Table 26: Results of compliance assessment of *Escherichia coli* levels (counts 100 ml^{-1}) recorded within the Swartkops Estuary from 1980 to 2013. (The numbers refer to the percentile calculated within each category.

Site name and distance from the mouth [km]	**1980 - 1986**		**1995 - 1997**		**2001 - 2004**	**2012 - 2013**	
Settlers Bridge [0.4]	no data		1 787		264	46	
Swartkops Village [4.0]	916		1 710		942	586	
Brickfields [6.6]	500	199	116 000		30 560	25 452	
Redhouse Yacht Club [10.0]	448	270	400	181	345	402	384
Bar None [13.6]	no data		118		no data	356	352
Perseverance [16.4]	1 335		622		490	4 080	

Compliance criteria for recreational water use	Excellent	≤ 250 (95 percentile)
	Good	≤ 500 (95 percentile)
	Sufficient / Fair	≤ 500 (90 percentile)
	Poor	> 500 (90 percentile)

4.3.8. <u>Visual observations</u>

a) Short-term spatial and temporal study (2012 – 2013)

A considerable amount of litter was found in the estuarine and river reaches prior to and during the study period (2012 to 2013; see Appendix E: Figure 43 and Figure 44). The Motherwell Canal was found to be a major source of litter to the estuary, with litter noted from the point of entry into the estuary to the far upper limit of the stormwater canal. Litter dump sites were found along the northern bank of the canal as well as scattered along the length of the canal (see Appendix E: Figure 43). Although pollution traps are located at regular intervals within the Motherwell Canal, these canals were full and required emptying or had been stolen. Furthermore, bottles and plastic bags were found at the entry point of the Motherwell Canal into the Swartkops Estuary, thus suggesting that the pollutant traps are not effective in retaining all litter that enters its course. Litter along the length of the Markman Canal was minimal but localised in the lower regions of the canal and retained by vegetation within its course. According to Rump (pers. comm.), dumping of litter directly into the Motherwell Canal (possibly into the pollutant traps) and into the canals of the Uitenhage area are a common occurrence.

Free-floating aquatic species were noted from the tidal limit to the lower reaches of the Bar None area of the estuary. Dominant macrophyte species at the tidal limit included *E. crassipes*, *A. filiculoides* and *S. molesta*, although *E. crassipes* was also noted further downstream (see Appendix E: Figure 45). The persistent occurrence of the macrophyte species in the upper estuarine reaches of the estuary has indicated that the Swartkops River and estuary is eutrophic.

b) *Comparison with past data*

The present study indicated that the litter situation has not changed or improved since the time of the earliest reportings, i.e. litter is carried to the Swartkops Estuary during moderate and high flows from the Motherwell Canal and the Chatty River (MacKay, 1994). Additionally, the Markman Canal is still not a major source of litter and debris to the estuary as pollution is usually trapped by vegetation in the canal. The same has been reported for the river reaches in the vicinity of Despatch, Uitenhage and KwaNobuhle (MacKay, 1993).

This study was the first to report on invasive macrophyte species occurring in the upper estuarine reaches of the Swartkops system. The first published account of invasive aquatic weed species was written by Lansdell (1925), who made mention of a problematic infestation of *Eichornia crassipes* in the Swartkops River. It is known that invasive aquatic weed species can form dense mats under optimal nutrient conditions, such as water hyacinth (*E. crassipes*) and the Kariba weed (*Salvinia molesta*), thereby obstructing the natural flow of rivers, which in turn can increase the extent of flooding. This situation was illustrated in 1977 where dense mats of *E. crassipes* caused extensive flood damage in the Swartkops River (Vermaat, 2005). Furthermore, Baird *et al*. (1986) also noted that infestations of *E. crassipes* were found in the lower reaches of the Swartkops River, and related this to excessive nutrient loading from the Kelvin Jones WWTW. Other invasive aquatic plants previously noted in the Swartkops River generally between Uitenhage and Perseverance included the red water fern (*Azolla filiculoida*; CSIR, 1993; DWAF, 2007), *Salvinia molesta* (CSIR, 1993) and the Spanish reed (*Arundo donax*) (DWAF, 2007).

4.4. **Discussion**

The main objective of this study was to determine spatial and temporal variability in water quality characteristics of the Swartkops Estuary and to relate any changes to anthropogenic impacts, freshwater flow and tidal stage. Since the only functional flow gauge was located at Nivens Bridge (16.4 km upstream of the tidal limit), and because three WWTW discharge their wastewater into the Swartkops River at distances of between 20 and 29 km from the mouth, a discrepancy in river flow measured at Nivens Bridge compared to that at Perseverance was unarguable. However, it is assumed that the combined daily discharge of 29 Ml day^{-1} (DWAF, 1999) of effluent from the three WWTW has increased over the years since the discharge flow amount exceeding capacity at the Kelvin Jones, Despatch and KwaNobuhle WWTW is 75 Ml day^{-1}, 77 Ml day^{-1} and 96 Ml day^{-1}, respectively (DWA, 2009). It is therefore assumed, that these exceedances could have given rise to the observed increase in vertical stratification in the present study (maximum difference of 7.9 ppt) compared to past observations (1 to 2 ppt; MacKay, 1994). For comparison, little stratification in the Swartkops Estuary has previsouly been reported by several authors during low river flow (Wooldridge and Melville-Smith, 1979; Melville-Smith and Baird, 1980; Emmerson, 1985; MacKay, 1993). It is known that depending on the degree of stratification and the duration of such conditions, several water chemisitry and ecological changes may occur, such as phytoplankton succession (Adams and Bate, 1999), a build of nutrients due to reduced vertical mixing, hypoxic conditions and increased residence time of bottom waters, providing ideal conditions for phytoplankton to develop. As a result, the water body experiences eutrophication pressure, with phytoplankton cell densities greatly exceeding bloom-forming levels. These observable consequences (of vertical stratification and nutrient enrichment) were noted in the present study, indicating deterioration in the water quality status of the Swartkops Estuary.

Trophic status of the estuary

Most eutrophic systems have an over-supply of nitrogen and phosphorus compared to that needed to sustain primary production, with the result that neither nitrogen nor phosphorus is limiting (Burkholder *et al.*, 2006). This phenomenon is typical of estuarine systems which are subjected to impacts from several anthropogenic activities such as the Swartkops Estuary. The present study showed that the nutrient status of the estuary is favourable for the development of eutrophic conditions, as neither nitrogen nor phosphorus were limiting. For comparison, a study by Jafta (2010) found that the Bushmans Estuary is 'oligotrophic'; whereas other permanently open estuaries such as the Sundays (Kotsedi, 2011) and Gamtoos (Snow, 2007) estuaries are also eutrophic. However, the latter two estuaries have lower nutrient concentrations in comparison to the Swartkops Estuary and therefore also have lower levels of water column chlorophyll-*a*.

Trophic status of the river

In addition to receiving nutrient inputs from the Motherwell and Markman canals and the Chatty River, nutrient loads originating from upstream riverine sources significantly impacted nutrient levels in the upper reaches of the Swartkops Estuary. This study showed that DIN levels recorded in the Swartkops River have increased from 1995 to 2013; a trend which verified the high DIN levels recorded in the Swartkops Estuary during 2012 and 2013. Additionally, DIP levels in the Swartkops River have increased significantly over the same period, which substantiated the persistent eutrophic levels observed in the estuary during the present study. In other Eastern Cape rivers (such as Great Fish, Sundays, Gamtoos, Keiskamma and Great Kei), the converse has been noted (i.e. nutrient levels are declining or are stable in the riverine reaches); confirming the Swartkops River as the most eutrophic river in the Eastern Cape (see Appendix D: Table 46) and one of the most threatened freshwater systems in South Africa (De Villiers and Thiart, 2007). Furthermore, long-term temporal trends have indicated that DIP concentrations recorded from 1995 to 2013 have decreased significantly ($p < 0.05$) in the Gamtoos, Keiskamma and Great Kei rivers, whereas in the Great Fish and Sundays rivers, DIP levels have shown downwards trends (see Appendix D: Table 46). In 2007, De Villiers and Thiart (2007) derived the same conclusions from a comprehensive evaluation on nutrient levels recorded in the 20 largest river catchments in South Africa.

Nutrient data collected from the 1970s to 2005 showed that the most pronounced increase in DIP was in the Swartkops River, with levels increasing at 0.01 mg l^{-1} DIP per year (De Villiers and Thiart, 2007). According to De Villiers and Thiart (2007), this translated to a double time of less than five years. Additionally, DIN levels recorded over the same time period have indicated a significant ($p < 0.05$) increase in the Great Fish River and only subtle increases in the Sundays and Gamtoos rivers (see Appendix D: Table 46); while DIN levels recorded in the Keiskamma and Great Kei rivers have remained relatively constant.

Impact of freshwater flow on nutrient enrichment and eutrophication

Nutrient response studies to open and closed mouth states of temporarily open/closed estuaries can provide useful insight into the advantages of continuous freshwater flow, tidal flushing and residence times on nutrient build-up and chlorophyll-*a* concentrations. For example, the Mhlanga Estuary and to a lesser extent the Mdloti Estuary are, like the Swartkops Estuary, impacted by nutrient contributions from wastewater treatment works. Together, the Phoenix and Mhlanga WWTW discharge 20 Ml of treated sewage into the Mhlanga Estuary per day (Thomas *et al*., 2005), whereas the Mdloti Estuary receives a combined volume of 8 Ml of treated sewage per day from the Verulam and Mdloti WWTW (Thomas *et al*., 2005). Even though the Mdloti Estuary is temporarily open/closed, DIN levels recorded in the estuary during both open and closed mouth phases (0 – 1.5 mg l^{-1}) (Thomas *et al*., 2005) were lower than levels recorded in the Swartkops Estuary during the present study (0.01 – 5.2 mg l^{-1}) (Table 27). Similarly, DIP levels measured in the Swartkops Estuary (0 – 1.6 mg l^{-1}) were also higher than DIP concentrations recorded during both open and closed mouth phases in the Mdloti Estuary (0 – 0.4 mg l^{-1}). Subsequently, these high DIN and DIP levels contributed to even higher concentrations of chlorophyll-*a* recorded in the Swartkops Estuary (0 – 248 μg l^{-1}) compared to that recorded in the Mdloti Estuary (0.9 – 111 μg l^{-1}). This comparison indicates that, despite the fact that the Swartkops Estuary is permanently open to the sea and is therefore regularly flushed; it presents similar levels of nutrients and eutrophication pressure to that of the Mdloti Estuary.

Similar impacts have been observed in the Diep Estuary where major pollution effects from overloading of wastewater treatment works have been reported (Taljaard *et al*., 1992). Taljaard found that during strong river inflow, DIN concentrations appeared to be strongly linked to anthropogenic sources, whereas during mouth closure, DIN concentrations were near depletion (nitrite: < 0.01 mg l^{-1}, nitrate: 0.01 mg l^{-1}, total ammonia 0.07 mg l^{-1}), with DIP concentrations averaging 0.6 mg l^{-1}. During open mouth state nitrite levels were less than 0.10 mg l^{-1}, while nitrate ranged from 0.2 to 1.0 mg l^{-1} and total ammonia ranged from 0.3 to 0.6 mg l^{-1}. DIP were slightly also higher with values ranging from 0.3 to 0.9 mg l^{-1}.

The results of the current study also showed that chlorophyll-*a* concentrations in the Swartkops Estuary exceeded concentrations recorded in other permanently open estuaries, such as the Sundays Estuary (mean: 29 µg l^{-1}; maximum: > 100 µg l^{-1}) and Great Fish Estuary (mean: 20.5 µg l^{-1}; maximum: 210 µg l^{-1}) estuaries by Adams and Bate (1999), as well as in the Gamtoos Estuary (maximum: 115 µg l^{-1}) by Snow *et al*. (2000b) and Bate *et al*. (2002). However, chlorophyll-*a* concentrations in the Swartkops Estuary were found to be comparable with levels recently recorded in the Sundays Estuary (Kotsedi, 2011) – The author recorded 237 μg l^{-1} 4.1 km from the mouth where the water column was strongly stratified.

Table 27: Chlorophyll-*a* concentrations (µg l^{-1}) measured in selected temporarily open/closed estuaries and permanently open estuaries in response to freshwater inflow or mouth state and nutrient levels.

	Flow ($m^3\ s^{-1}$)		**Chl-*a* (µg l^{-1})**		**DIN (mg l^{-1})**		**DIP (mg l^{-1})**	
	Closed/ low flow	**Open/ high flow**	**Closed/ low flow**	**Open/ high flow**	**Closed/ low flow**	**Open/ high flow**	**Closed/ low flow**	**Open/ high flow**
Permanently open estuaries								
Swartkops (this study)	0.34	1.75	1.3 – 158	0 – 248	0.1 – 5.2	0 – 3.7	0.1 – 3.6	0 – 1.6
Bushmans[1]			< 9.0		< 0.4		< 1.7	
Sundays[2]			< 237		0 – 1.9		0 – 0.3	
Temporarily open/closed estuaries								
Mhlanga[3]			1.3 – 303	0.7 – 27	0.2 – 4.7	0.2 – 5.9	0.1 – 2.3	0.2 – 2.5
Mdloti[3]			3.7 – 111	0.9 – 18	0 – 0.9	0.4 – 1.5	0 – 0.1	0 – 0.4
Goukamma[4]			0.8 - 289	0.3 - 112	0 – 1.9	0 – 0.9	0 – 0.2	0 – 0.2
Sources: [1]Jafta (2011), [2]Kotsedi (2011), [3]Thomas *et al*. (2005), [4]Kaselowski (2012)								

For comparison, the highest chlorophyll-*a* concentration recorded in the Swartkops Estuary during the present study was measured 13.6 km from the mouth where the water column was noticeably less stratified than further downsteam between distances of 4.0 and 10.0 km from the mouth. It was found that high river flow due to rainfall events prior to sampling gave rise to higher phytoplankton chlorophyll-*a* biomass in both estuaries. Additionally, Kotsedi *et al*. (2012) found that the effect of reduced freshwater inflow resulted in an overall increase in DIN and DIP levels, whereas Scharler and Baird (2003) and the present study observed the opposite trend.

There is a limited understanding of the long-term changes in freshwater flow into the Swartkops Estuary and therefore, the flow conditions and residence times required to attain maximum phytoplankton biomass. Lord and MacKay (1993) estimated that a residence time of longer than 14 days is the norm in the Swartkops Estuary, however, since discharge volumes from the three WWTW have increased over the years, this is assumed to have changed. The present residence time of the Swartkops Estuary can be deduced, to some extent, based on patterns observed in other estuaries, such as the Sunday and Gamtoos. For example, it was previously noted that three spring tidal cycles, or 42 days, are required for phytoplankton to bloom and attain maximum biomass. In the case of the Gamtoos Estuary, a flow rate of just below 1 $m^3\ s^{-1}$ attained a phytoplankton maximum concentration of 115 $\mu g\ l^{-1}$. Scharler *et al*. (1997) showed that mean phytoplankton biomass recorded between Bar None and Perseverance ranged from 8.6 to 22.3 $\mu g\ l^{-1}$ between 1993 and 1994, whereas in the present study it ranged from 71.6 to 89.0 $\mu g\ l^{-1}$ (see Table 24). These observable changes indicate that the Swartkops Estuary has been severely impacted by changes in freshwater flow and subsequent nutrient loading from the river, and that the residence time can be expected to be greater than 14 days. However, the predicted increase in residence time does not align with the suspected increase in base flow (due to increases in wastewater discharges). This suggests that nutrient loads to the Swartkops Estuary have increased irrespective of changes in freshwater inflow, and possibly as a result of inadequate treatment of the domestic and industrial wastewater at the WWTW.

In contrast to the Swartkops Estuary where low flow conditions in February, May and August 2013, were associated with eutrophic levels of nutrients and phytoplankton blooms (> 10 000 cells ml^{-1}), Kotsedi *et al*. (2012) found that in the Sundays Estuary a consistently high river flow ensured that a build up of organic material did not occur in the estuary and thus hypoxic and anoxic events weren't observed. However, the authors concluded that should a reduction in river flow occur, an increase in the residence time of the water would lead to frequent occurrence of phytoplankton blooms. In the present study, eutrophic levels of nutrients and phytoplankton blooms were recorded during both low and high flow conditions. The increase in DO in the upper reaches of the Swartkops Estuary was related to an increase in phytoplankton biomass.

Phytoplankton biomass and community composition

Phytoplankton biomass contributes significantly to diurnal fluctuations in oxygen concentrations in an estuary, i.e. during the day, when phytoplankton photosynthesize, oxygen is produced, and in turn oxygen is consumed during bacterial degradation of phytoplankton, which is a common symptom of amplified phytoplankton growth in estuaries, such as the Swartkops. The severity of this becomes more apparent during the warm summer months, when nutrients fuel eutrophication, causing phytoplankton blooms. If decaying blooms are large enough and bottom water oxygen cannot be replenished by downward mixing of oxygenated surfaces waters then anoxia occurs (Newman and Unger, 2003). High inputs of organic material (i.e. sewage-derived effluent) can exaberate anoxia and hypoxia by elevating the biochemical oxygen demand (the oxygen consumed during the microbial decomposition of organic matter) and the chemical oxygen demand (the oxygen consumed through oxidation of ammonium and other inorganic reduced compounds). The scenarios described above were observed in the Swartkops Estuary during the present study under conditions of vertical stratification and high water temperatures.

It is to be noted that while ammonium and chlorophyll-*a* were significantly ($p < 0.05$) inversely correlated to salinity (which would suggest that as freshwater input increased, so did ammonium and chlorophyll-*a*), it can also be assumed that this is due to residence time/dilution. However, the present study concludes that there were no apparent effects of tidal stage and flow regime on chlorophyll-*a* in the estuary, since the highest chlorophyll-a concentrations were recorded on a neap flood tide (with higher flow conditions) and also on a spring ebb tide in February 2013 (with lower flow conditions).

It was previously stated that the onset and occurrence of eutrophic conditions are associated with decreases in diatoms and increases in flagellates (Stoemer and Smol, 1999) and that a shift from a diatom-dominated community towards flagellates can occur under conditions of phosphorus enrichment (Heisler *et al*., 2008). The results of the present study supported these two statements, in that the growth of flagellates was favoured in the present study. However, the study also showed that phytoplankton blooms of flagellates and diatoms generally occured over a salinity range of 8.8 ppt (at 13.6 km from the mouth) to 29.5 ppt (at 4.0 km from the mouth), although phytoplankton group dominance was mostly influenced by flow and nutrient availability. In contrast, phytoplankton blooms (> 20 µg l^{-1}) in the Sundays Estuary have been associated with low salinity in the middle to upper reaches only; which in this case, indicated that nutrient-rich freshwater inflow stimulated high chlorophyll-*a*. Also in the Sundays Estuary, Kotsedi (2011) found that high chlorophyte and diatom cell numbers were associated with low salinity water in the upper reaches of the estuary. This was true for flagellates, diatoms and chlorophytes in the upper reaches of the Swartkops Estuary. Kotsedi (2011) noted that flagellates were dominant throughout the estuary when nutrients were possibly depleted by other algal groups, whereas a dinoflagellate bloom was correlated with high chlorophyll-*a*, NH_4^+ and pH. In the Swartkops Estuary, a different response was observed, in that, elevated nutrient levels were associated with high flagellate cell numbers. Additionally, *Euglena* cells were poorly represented in the present study; however this group was limited to sites where nutrient concentrations were high (i.e. Perseverence and Markman Canal). Literature has indicated that the *Euglena* group is referred to as a pollution indicator species, and also the most tolerant genus of organic pollution/domestic waste (Person, 1989). They are also found in waters of high nitrogen concentrations, such as lagoons where sewage is treated and in stagnant pools (Person, 1989).

Since sampling days were not consecutive, sequences of phytoplankton succession could not be determined. However, a marked increase in freshwater inflow measured in November 2012 was followed by the highest phytoplankton biomass recorded in the estuary during the study. This was associated with a bloom of diatom cells occurring from Perseverance to Swartkops Village in the surface waters. The present study ascribed this observation to one of the following scenarios: (1) diatom species were more tolerant towards elevated flow rates (Hamilton, 2000), (2) diatoms correlated positively to increased turbidity and relatively low light at the time of sampling (Bormans and Webster, 1999; Marshall *et al*., 2006) and (3) diatoms were introduced into the estuary following increased freshwater inflow (Lucas, 1986; Snow *et* *al*., 2000b). Kotsedi (2011) monitored the Sundays Estuary over five consecutive weeks to determine short-term variability of phytoplankton composition and biomass in response to physical, chemical and climatic factors. Although chlorophyll-*a* concentrations were significantly ($p < 0.05$) variable between the different weeks, phytoplankton succession was not strongly exhibited in the study. However, Kotsedi (2011) found diatoms and flagellates to be the most abundant groups, succeeding each other in terms of dominance. Unlike the Swartkops Estuary, where high flow conditions, low DIP and high TOxN levels were associated with bloom densities (> 20 µg l^{-1}) of diatoms and flagellate blooms and also low flow conditions and higher nutrients levels (Table 43), no single factor was identified as the driver or trigger of phytoplankton blooms in the Sundays Estuary. However, it was observed that diatoms occurred in blooms during warm, calm conditions, which besides the warmer temperatures, was not consistent with the environmental conditions which fuelled diatoms to bloom in the Swartkops Estuary, since freshwater flow was at its highest on 20 November 2012.

Minne (2003) showed that NH_4^+ and salinity had significant effects on the distribution of diatoms, whereas DIP had no effect in a number of Eastern and Western Cape estuaries. For example, the same author found that in the Great Fish Estuary, diatom species were associated with low salinity whereas in the Breede, Bushmans, Kowie and Mpekweni estuaries diatoms were associated with high salinity. Within the context of this study, the diatom bloom in November 2012 was associated with low salinity ($r = -0.75$; $p < 0.05$) and showed a poor relationship with NH_4^+ ($r = 0.20$; $p > 0.05$). The former was not consistent with the findings by Minne (2003) where diatoms were associated with high salinity. Based on these findings it has become apparent that environmental factors which fuel diatom production in the Swartkops Estuary have been altered, and favour conditions of lower salinity. Moreover, Kotsedi (2011) found that wind-mixing and reduced temperatures promoted dominance of flagellates throughout the estuary. This too was not regarded as a driver of flagellate blooms in the present study since wind speeds ranged from 1 to 10.1 km $hour^{-1}$ when bloom densities were recorded in February, May and August 2013. Kotsedi (2012) noted that the highest cell densities occurred from 12.5 km from the mouth and that DIP possibily limited phytoplankton growth based on the high DIN:DIP ratios, particularly in the lower reaches of the estuary. This was not observed in the Swartkops Estuary, since bloom forming densities mostly occurred from 4.0 km from the mouth in response to eutrophic levels of DIP in the estuary.

Overall, the present study found that phytoplankton dominance by flagallates is not exclusive to the Swartkops Estuary. Adams and Bate (1994) concluded that the phytoplankton community in the Berg, Palmiet, Goukou, Gourits, Great Brak, Keurbooms, Gamtoos and Sundays estuaries are also dominated by flagellates. However, nutrient levels alone did not determine the species composition in these estuaries, but rather the stratified conditions of the water.

Trace metals

Based on past data, the present study has shown that contamination by trace metals has increased in the Swartkops Estuary. It was observed that levels of copper, lead, zinc, iron and cadmium had increased by approximately 93% within the estuary itself, at the tidal limit (Perseverance) and in the Motherwell Canal. The elevated levels of cadmium and lead, corroborate the results of a pollution study conducted on fish which showed that fish species "contained alarming amounts of dangerous metals" with extremely high concentrations of cadmium and, to a lesser extent, lead (see Appendix A: Article 8 and also Article 9). Concentrations of iron and cadmium were not considered safe for recreational use of the estuary (i.e. intake of water should not exceed "200 ml per day, that is, 100 ml per recreational session with two sessions per day"), whereas levels of copper, zinc, and cadmium exceeded acceptable levels of trace metals in coastal aquatic ecosystems. Total chromium (recorded at Tippers Creek) and dissolved selenium (recorded at Swartkops Village) were highlighted as metals of concern, requiring further investigation to confirm their levels of toxicity. Additionally, the high levels of chromium detected in the Markman (140 $\mu g\ l^{-1}$) and Motherwell (130 $\mu g\ l^{-1}$) canals can be a potential threat to the estuary in the future and therefore require further monitoring. Typical industry types which use large amounts of chromium, include electroplating operations, leather tanning, textile, paint and pulp (DWAF; 1996a; Kabir *et al*., 2012), all of which are located within the Markman industrial area and thus constitute suitable sources. Chromium may also be found in certain types of fertilisers, pesticides, vehicle exhausts, tyres and roof runoff. Due to the close proximity of the Tippers Creek area to the residential area of Amsterdamhoek (and Amsterdamhoek Drive), the present study considered vehicle exhaust fumes in the form of road runoff to be a source of chromium, as well as boating related sources and effluent seepage from conservancy tanks. In the Motherwell Canal, source identification was less obvious, though roof runoff could constitute a likely source of chromium due to the canal servicing a large area of low-cost housing with corrugated roofing. It is known that some types of corrugated roofs contain chromium (Berggren *et al*., 2004). Further investigations are required to substantiate these sources of chromium.

Due to project constraints, it was not possible to conduct a comprehensive study on temporal changes in trace metal concentrations and therefore, also, the influence of freshwater flow, and the effect of metal concentrations on nitrogen cycling (e.g. Talbot, 1988), however, this has been illustrated elsewhere. For example, Mzimela *et al*. (2003) investigated seasonal patterns of selected metals in water from the Mhlathuze Estuary which is located along the KwaZulu-Natal coast and within close proximity of the Richards Bay Harbour. Like the Swartkops, rapid development has taken place in the catchment area, including a large array of different industries and discharges of domestic and industrial wastewater into the estuary and rivers in its catchment. The authors found pronounced seasonal variations with the highest metal concentrations recorded during summer. These metals were aluminium (26 200 $\mu g\ l^{-1}$) and iron (23 500 $\mu g\ l^{-1}$); both which coincided with extremely high freshwater inflow from the Mhlathuze River. Other metals such as chromium (14 – 226 $\mu g\ l^{-1}$), copper (14 – 76 $\mu g\ l^{-1}$), lead (82 – 448 $\mu g\ l^{-1}$) and zinc (52 – 112 $\mu g\ l^{-1}$) remained relatively constant during the study period. The study found that lower concentrations generally coincided with reduced riverine runoff from the catchment of the estuary at the time of sampling. In comparison to concentrations recorded in the Swartkops Estuary (this study), only copper (10 – 70 $\mu g\ l^{-1}$) and zinc (50 – 90 $\mu g\ l^{-1}$) concentrations appeared within comparable ranges. All other metals were higher in the Mhlathuze Estuary. The higher levels of aluminium, iron, chromium and lead recorded in the Mhlathuze Estuary were believed to be indicative of the different types of industrial activites occurring within the catchment area.

It was previously noted that freshwater inflow into an estuary can have a significant impact on the toxicity of trace metals in surface waters (e.g. Kariega Estuary, Mhlathuze Estuary and Nerbioi-Ibaizbal River Estuary); sometimes to the extent that trace metal concentrations do not comply with water quality criteria. Had freshwater flow at the time of sampling in the present study exceeded 2.14 $m^3\ s^{-1}$, the observed increases in copper and cadmium with distance from the tidal limit may have been flushsed out or diluted to concentrations below the criteria for recreation and coastal marine ecosystems. Additionally, the studies by Orr *et al*. (2007) and Amigo *et al*. (2012) illustrated that an estuarine flow requirement study is required to determine the optimal flow in the Swartkops Estuary at which trace metals comply with water quality criteria. Since several factors limited temporal variability and the spatial extent of trace metals in the Swartkops Estuary, it is strongly encouraged that metals are routinely monitored under conditions of high and low flow.

<u>Faecal bacteria</u>

In the present study, contamination by faecal bacteria in the Swartkops River and estuary was high, with bacteria counts indicating marked seasonal patterns. Additionally, elevated nutrient levels in the Swartkops Estuary and at the points of entry into the estuary were concomitant with high bacteria counts (e.g. Daly *et al*., 2013). This was expected since domestic waste is naturally high in nutrients, especially nitrate and ammonium. Historical data of faecal bacteria collated with data from the present study showed that *E. coli* levels in the Motherwell Canal and the Swartkops River have been persistently high, and are the two major sources of faecal bacteria to the estuary; counts ranged from 0 to 40 x 10^{-6} counts 100 ml^{-1} and from 0 to 200 000 counts 100 ml^{-1} respectively. Elevated *E. coli* counts occurring in the upper reaches of the estuary and in the lower and middle reaches of the Swartkops River were attributed to the wastewater treatment works that discharge effluent into the river. A distinct peak in *E. coli* was consistently measured in the middle reaches of the Swartkops Estuary in the vicinity of Brickfields, which was consistent with high levels of faecal bacteria in the Motherwell Canal. The Motherwell Canal was and still is a significant source of faecal bacteria to the middle reaches of the estuary, mostly due to leaks in the sewer system within the Motherwell Township area, resulting in domestic waste entering the Motherwell stormwater canal. MacKay (1993) noted that during wet or dry weather, elevated levels of *E. coli* can be found in the canal and that a constant low flow prevails. These findings corresponded with the findings of the current study (2012 - 2013). This study also noted the importance of tides and river discharges as illustrated by a dilution effect on *E. coli* concentrations downstream of WWTW discharge sites. Although De Brauwere *et al*. (2011) found that tide is crucial to explaining increased concentrations upstream of discharge sites in the Scheldt River estuary, in this study river flow was found to play a greater role considering the estuary as a whole. This is because the combined volume of wastewater from the three upstream WWTW has a greater impact on upstream sites (due to poor or no tidal flushing), than lower estuarine sites (i.e. Brickfields) which are subjected to vigourous tidal flushing and osmotic shock due to increased salinity.

For comparison with other South African estuarine systems, it was found that studies on bacteria levels are limited and only a few are available for comparison. However, poorly maintained and inadequate wastewater treatment facilities, are not limited to the Swartkops catchment area and are in fact considered to be major environmental health risks within every South African municipality. For example, the Kleinrivier Estuary, like the Swartkops Estuary, is an important recreational assest to the town of Hermanus. It is currently graded as the 5th most important estuary in South Africa and is intermittently polluted with *E. coli* and faecal coliforms, especially when the water level is low, and during the summer months (Hamilton-Attwell, 2007). Most of the residential estates along the estuary are not linked to the municipal sewerage reticulation system and therefore residences rely on either septic or conservancy tank systems. Hamilton-Attwell (2007) noted that maximum bacteria counts of 2 419 counts 100 ml^{-1} are often recorded at different stations in the estuary. The study pointed out that a WWTW contributes 50% towards the inflow of treated effluent into the Klein River, while the other 50% is attributed to untreated effluent seeping from septic tanks from properties alongside the river on both banks. The situation described above is comparable with the current bacteriological status of the Swartkops River and estuary, although sewage effluent that enters the Swartkops is amplified due to three wastewater treatment works discharging into the Swartkops River. In addition, several high-density residential areas also found within close proximity to the river and estuary which don’t have modern day sanitation facilities or experience leaks in their sewer systems. Consequently, this has led to maximum bacteria counts that are far higher than those recorded in the Klein River. The Plankenburg River (Stellenbosh) and the Berg River (Paarl) are two rivers in the Cape Metropolitan-Boland area that are regarded as highly polluted with recorded bacteria levels similar to those recorded in the Swartkops catchment area. Informal settlerments inhabit the banks of these rivers and stormwater drainage pipes from these settlements flow directly into both rivers (Barnes, 2003). Barnes (2003) recorded an *E. coli* count as high as 2.44 x 10^9 counts 100 ml^{-1} at the a stormwater drainage pipe that discharges to the Berg River. Like the Swartkops River catchment, raw sewage spills from sewer pump stations in Wellington, overstressed sewer mains and stormwater effluent from informal settlements were identified as possible sources of pollutions. Several years later, Paulse *et al*. (2007) recorded *E. coli* counts ranging from 36 to 1.7 x 10^7 counts 100 ml^{-1} at a site where stormwater drainage pipes from the informal settlement flow directly into the Berg River. These results were comparable with *E. coli* counts recorded within the Motherwell stormwater canal (5.8 x 10^6 counts 100 ml^{-1}) in November 2012, and which once discharged to the estuary, were significantly reduced (42 x 10^3 counts 100 ml^{-1}), yet significantly higher than the guideline value. In another study, Morrison *et al*. (2001) revealed that raw sewage discharges resulting from the inadequate Keiskammahoek wastewater treatment plant and a malfunctioning pump station, contributed to increased oxygen demand and nutrient loading of the receiving water body. This in turn led to eutrophication and algal blooms; much the same as the conditions reported in the Swartops Estuary in the present study. These examples illustrate that faecal pollution of aquatic ecosystems is a generic concern throughout South Africa and unless addressed, will continue to affect the ability of the these natural systems to support a health aquatic life.

It is widely observed that faecal pollution presents a major health risk for recreational use of water. Recreational waters in particular require consistent monitoring since bacteria levels may increase or decrease by several orders of magnitude within a sort space of time due to environmental effects (Sinton *et al*., 2002; Neger, 2002; Alam and Zafar; 2012; Blaustein *et al*., 2013). Additionally, routine monitoring of sediments for faecal bacteria is often disregarded, mainly due to sampling difficulties and financial constraints associated with analysis of sediments for faecal bacteria. This type of analysis will hold significant value when recreational waters have to be assessed for faecal pollution prior to sporting events such as the Redhouse River Mile swimming event, as disturbed sediments can result in resuspension of faecal pollution. For example, Alam and Zafar (2012) investigated spatial and temporal variability in *E. coli* in water and soil in relation to changes in physico-chemical characteristics of the water column of the Karnafuly Estuary in Bangladesh. The authors found that *E. coli* were positively correlated ($p < 0.01$) with water temperature and negatively correlated with salinity, pH and dissolved oxygen, though interestingly *E. coli* levels in the sediment were 3.7 times greater than levels recorded in the water column. Since the later did not form part of the present study, future studies should investigate riverine and estuarine sediments as potential sources of *E. coli*, especially since the Swartkops River and estuary is relatively shallow. This feature renders the Swartkops Estuary prone to re-suspension of *E. coli* into the water column, especially when the water column and sediments are disturbed during times of increased recreational activity.

<u>Visual observations</u>

The present study indicated that litter in the form of plastic bags and bottles is a major concern especially along the banks of the Motherwell Canal and within the canal itself; and also in the freshwater reaches of the Swartkops River. Visual observations clearly indicated that litter centered around densely populated areas, such as the Motherwell Township and certain areas of Despatch and Uitenhage i.e. at the junction of the Middle and Kat canals (Figure 54; also see Appendix A: Article 6), which is consistent with previous observations (MacKay, 1994). Litter traps are located in parts of the Kat and Middle Street canals, however unlike the Motherwell Canal they are effective in retaining the litter, thus preventing it from entering the estuary. Although litter within the Markman Canal was minimal during the present study, litter washed into the Markman Canal would be retained by the natural vegetation (MacKay, 1994) under wet weather conditions. Litter carried to the estuary from the Chatty River was not a concern in the current study; however, in the study of MacKay (1994) the Chatty River was noted as a source of litter especially during moderate and high flows. It was beyond the scope of the current study to determine whether litter carried to the estuary has increased or decreased in the subsequent years, however litter was generally localised to residential areas.

Invasive aquatic plants in the Swartkops River have persisted since the earliest publication by Lansdell (1925) and have been attributed to nutrient loading from the Kelvin Jones WWTW. Three weed species, namely *E. crassipes*, *S. molesta* and *A. filiculoida* were recorded at the tidal limit of the estuary, whereas *E. crassipes* was also noted further downstream as far as Bar None. The present state conditions of the Swartkops Estuary were such that large stands of these floating aquatic macrophytes populated the upper reaches of the estuary indicating eutrophic conditions.

4.5. **Conclusions**

Both the quality and quantity of sewage-derived effluent, industrial effluent and urban runoff can impact receiving freshwaters as well as the marine environment, impacting ecosystem functioning. The most obvious of ecological impacts result from increases in nutrient loads in the receiving environment (with associated growth of phytoplankton and free-floating invasive species). In the present study, sewage effluent resulted in several water quality related problems, i.e. eutrophication, infestation of invasive plant species and elevated faecal bacteria.

The main objective of this study was to determine the spatial and temporal variability in water quality characteristics of the Swartkops Estuary. It was clear that spatial and temporal variability in nutrient levels, trace metals, phytoplankton biomass and faecal bacteria existed. The input of nutrients and faecal bacteria to the Swartkops River and estuary via wastewater treatment works and stormwater systems, respectively, were severe. Prior to 1986 the water quality of the estuary was generally considered to be suitable for recreation, but the sensitivity of the estuary to domestic and industrial effluent has become more pronounced, as indicated by the lasting effects of nutrient enrichment (i.e. persistent phytoplankton blooms) observed in the present study. Additionally, the study showed that neither reduced nor increased flow to the Swartkops Estuary resulted in reduced phytoplankton blooms from occurring.

Aquatic systems with long residence times (and low flow regimes) are particularly susceptible to developing algal blooms under conditions of nutrient enrichment whereas estuaries with short residence times are characterized by increased flow which usually prevent eutrophication by flushing nutrients out of the system (Grall and Chauvaud, 2002). This study found that the Swartkops Estuary was in a permanent state of eutrophication regardless of changes in residence times and associated flow conditions. This observation emphasized the need for improved management of effluent originating from sewage treatment facilities and commercial industries, improved management of urban runoff and determination of estuarine flow requirements.

The overall hypothesis tested was that there has been a general deterioration in water quality of the Swartkops Estuary as evidenced by an increase in nutrients, trace metals, faecal bacteria, phytoplankton biomass and phytoplankton cell densities. In view of the following, this hypothesis was not rejected:

- The estuary was found to be well oxygenated (~7.2 mg l^{-1}, 1.3 to 18.2 mg l^{-1}), however differences between surface and bottom concentrations of dissolved oxygen were notably larger ($p < 0.05$) in this study. A review of past data indicated that vertical gradients for oxygen in the Swartkops Estuary are not distinct (McLachlan, 1972; Emmerson, 1985; Scharler *et al*., 1997) and oxygen depletion in bottom water is of rare occurrence (Scharler *et al*., 1997). The observed changes in the present study would have occurred due to increased rates of photosyntehsis occurring in the surface waters in response to increased phytoplankton biomass. Additionally, there was no correlation between DO, temperature and salinity. These observations suggested that an anthropogenic source(s) was responsible, and that elevated DO levels occurred in response to nutrient over-enrichment of the surface waters, resulting in increased photosynthesis and in turn elevated DO concentrations.

- All nutrients ($NH4^+$, TOxN, DIN, and DIP) increased with distance from the mouth. Ammonium concentrations showed localised increases in the middle reaches of the estuary (due to stormwater discharges from the Motherwell Canal) and at the tidal limit (February 2012: 1.93 mg l^{-1}, May 2013: 2.65 mg l^{-1}). Watling (1982) concluded that occasional high NH_4^+ levels occur in the presence of anoxic waters as a result of failures occurring at the Kelvin Jones WWTW. Results of the current study corroborate this. Overall, the Motherwell Canal and to a lesser extent the Markman Canal were the greatest sources of nitrogen compounds to the estuary, and contributions from the Chatty and, Swartkops rivers were smaller in comparison. The Swartkops River was the primary source of DIP throughout the study, with levels decreasing steadily from the tidal limit to the mouth. These trends had been observed in past studies (MacKay, 1993; Scharler *et al*., 1997), however, the present study found an overall increase in nutrient levels in the Swartkops Estuary when compared to past data.

- Trace metal concentrations were spatially variable, though not all metals were elevated at sampling sites located within close proximity to diffuse sources (i.e. the Motherwell and Markman canals, Tippers Creek and the Swartkops River). Preliminary data on trace metals suggested that levels of copper, zinc, iron and cadmium have increased by at least 90% since the study by Watling and Watling (1982), in the Swartkops Estuary, at the tidal limit of the estuary and in the Markman and Motherwell canals. Further investigations are required to substantiate that these increases constitute as tangible evidence that support a general deterioration in water quality of the Swartkops Estuary.

- Chlorophyll-*a* in the Swartkops Estuary showed an increase with time when compared to historical data (Scharler *et al*., 1997; Binning, 1999). This finding supported the general hypothesis of a general deterioration in water quality of the Swartkops Estuary in response to an overall increase in nutrient levels. Additionally, phytoplankton biomass was highest in the upper reaches of the estuary. This was a reflection of increased concentrations of nutrients with distance from the mouth.

- Temporal and spatial variability in faecal bacteria was evident during the present study. Faecal bacteria counts were highest in the summer months, and at estuarine sites which are impacted by anthropogenic activities, i.e. Perseverance and Brickfields. Overall, it was clear that high faecal bacteria counts are still present in the estuary with no improvements observed since the studies by Emmerson (1985), Lord and Thompson (1988) and Lord and MacKay (1993). It can therefore be said that faecal bacteria continues to have a negative impact on the health status of the estuary and therefore also the recreational value of the estuary.

From the findings given above, it was clear that the Swartkops Estuary remains severely impacted by the release of pollutants from land-based activities and stormwater discharges. Additionally, the present study highlighted the lack of consideration which has been given to the various facets that are necessary for efficient water quality management. This was particularly true for the estuarine monitoring sites which showed exceptionally low monitoring frequencies by the Department of Water Affairs (http://www.dwa.gov.za/iwqs/wms/data/000key.asp) and also inconsistencies in analytical methods used. The study was thus valuable in stressing the need for a significant improvement in consistent data collection, standardisation amongst laboratory methods, and improved data management.

4.6. **Recommendations**

4.6.1. Long-term monitoring

a) <u>Spatial and temporal monitoring requirements</u>

During the present study, collation of historical data from the Department of Water Affairs emphasised the need for improved spatial and temporal monitoring frequencies of data collection for nutrient and trace metal levels within the estuary. Nutrient studies conducted on the waters of the estuary by academic institutions have been ongoing; however the lack of routine monitoring (i.e. continuous monthly data) by water quality regulatory officials was evident. If officials were to rely on governmental (internal) data records for trend analyses and therefore management decisions concerning nutrient levels and trace metals within the estuary, then the present study proved this to be impossible.

Nutrient trend analyses of the estuarine water is of particular importance due to increased rates of urbanisation and poorly maintained sewerage systems within the Swartkops catchment area, resulting in increased urban and stormwater runoff – all of which have been found to influence nutrient levels within the estuary (i.e. Motherwell Canal and Chatty River). In addition to this, effluent discharges from wastewater treatment works located above the tidal limit have for some time influenced the nutrient status of the upper reaches of the Swartkops Estuary. It is, thus, recommended that spatial and temporal monitoring is conducted on a monthly basis at site-specific locations within the estuary and at all points of entry into the estuary, in such a manner that statistically-proven trends on nutrients, trace metals and faecal bacteria levels are achievable. In essence, this is required to measure the magnitude of human influences on the water quality of the Swartkops Estuary.

b) Data assimilation and transfer

The present study brought to light another concern with regards to water quality data availability; that is the preparation and dissemination of water quality monitoring data. The Department of Water Affairs encourages and stresses the need for data acquisition, data management and storage, and information generation and dissemination (DWAF, 2004). In the present study, both field data records and water quality laboratory reports were found in storage boxes of a water regulatory office which required capturing prior to use in the present study. Furthermore, data which is available online (http://www.dwa.gov.za/iwqs/wms/data/000key.asp) was found to be updated only every four to six months. The significance of this is that delays occur when water quality data are needed for decisions regarding development proposals and problem specific remedial action plans. It is advised that water quality data be captured following its release from the laboratory.

c) Faecal bacteria

Escherichia coli levels in the Swartkops Estuary and river have been elevated far beyond water quality criteria for recreational water use with sporadic episodes often greater than 10 000 counts 100 ml^{-1}. However, since the implementation of the revised South African water quality guidelines for faecal bacteria of coastal marine waters (RSA DEA, 2012), *E. coli* is no longer regarded as the primary indicator of faecal contamination, but instead enterococci counts. For baseline or short-term water quality surveys this has meant that compliance criteria are no longer applicable. Moreover, there are currently no long-term records of enterococci levels recorded by the Department of Water Affairs to relate to current levels of enterococci; which for temporal analyses and water resource management is a concern. It is thus strongly advised that the Department of Water Affairs continues with the collection of *E. coli* data (for temporal analyses and until such time that long-term enterococci data are available), and monitor enterococci levels in the Swartkops Estuary (RSA DEA, 2012).

d) Freshwater inflow gauges

Presently, the only freshwater flow gauge is located at Nivens Bridge. This is approximately 16.4 km upstream of the tidal limit i.e. Perseverance. It is recommended that if statistically correct interactions between freshwater inflow, residence times and nutrient levels in the estuary are to be determined, then a freshwater flow gauge positioned at the tidal limit would be beneficial for long-term studies on nutrients, phytoplankton biomass and bacteria inputs from the Swartkops River.

4.6.2. Additional research

a) Influence of seaward nutrient inputs from Fishwater Flats Wastewater Treatment Works

The present study noted sporadic elevations in faecal bacteria counts at the mouth of the Swartkops Estuary and attributed this to sewage outfalls from the Fishwater Flats WWTW and sewage contributions from the Papenkuils River (which receives sewage effluent from the treatment plant itself). It is recommended that future water quality investigations consider conducting dye dispersion studies to determine the extent of faecal bacteria dilution and transport from the Papenkuils River mouth and the Fishwater Flats discharge pipeline (which extends 0.17 km into the Algoa Bay) (Anonymous, 2004) into the mouth of the Swartkops Estuary.

b) Trace metals

Several studies have related the accumulation of trace metals in aquatic species (Silva *et al*., 2001; Mzimela *et al*., 2003; Montes Nieto *et al*., 2010; Jepkoech *et al*., 2013) and their presence in sediments, groundwater and surface water to anthropogenic sources (Mzimela *et al*., 2003; Jackson *et al*., 2007; Newman and Watling, 2007; Jackson *et al*., 2009), whilst others have considered terrigenous sources (Newman and Watling, 2007; Alagarsamy and Zhang, 2010). Limited data is available concerning the bioavailability and resuspension of trace metals in the Swartkops Estuary, including their sources and the effect of tide. Consequently, not enough is known about the temporal and spatial variability and the influence which changes in salinity (in response to tide) may have had on concentrations of previously recorded concentrations in the water column, especially since all previous studies have only analysed single water samples. It is therefore recommended that future water quality surveys emphasise the influence of salinity and include analysis of trace metals contained within aquatic plant species, filter feeders, the water column, marine and terrigenous sediments and the groundwater of the Swartkops Catchment, including regular intervals within the Motherwell and Markman canals and the Chatty River. To differentiate naturally occurring and anthropogenically introduced concentrations of trace metals, geochemical normalisation studies are also recommended, as is illustrated by Newman and Watling (2007). It is also recommended, that the study be coupled with the effects of both an ebb and flood tide to gain a better understanding of the relationship between submarine groundwater discharge and recharge and trace metal concentrations in the Swartkops River and estuary.

c) Surface and groundwater interactions

Reviewed international literature (Winter *et al*., 1998; Hinsby *et al*., 2012; Carol *et al*., 2013; Sawyer *et al*., 2013; Unland *et al*., 2013; Wong *et al*., 2013) has illustrated the interactions between groundwater quality and estuarine water quality. In particular, studies in the United

States have found that much of the groundwater contamination in the shallow aquifers is directly connected to surface water (Winter *et al*., 1998). In South Africa, direct linkages between surface water bodies and ecosystems services with groundwater storage, recharge and discharge are not yet recognised and valued in decision making and in the management of water resources and river basins (Pietersen *et al*., 2011). It is therefore, not surprising that not much work has been done to assess the potential contribution of groundwater to freshwater flow to the Swartkops Estuary. Following a water quality monitoring exercise in 1993, it was observed that agricultural application of fertilisers and manure are potential contributors of nutrients to the groundwater of the Swartkops catchment (Haigh, 2002). Similarly, seepage from landfill waste sites, industrial evaporation ponds, sewage maturation ponds and sludge lagoons have previously been reported as being potential contributors of nutrients to the groundwater of the Swartkops catchment. Although, no significant levels have been measured in the groundwater below the Kruisrivier farming area near Uitenhage at the time of the study, altered water chemistry was noted in the surface and groundwater at Nivens Bridge, mostly as a result of industrial activities. In addition, elevated nitrate/nitrite concentrations have been observed below Grahams Poultry in Uitenhage, where chicken manure is used as fertiliser. Another potential source of nutrient contamination is the sewage sludge from the wastewater treatment works which are disposed of on drying beds upstream of the Nic Claasen Bridge, on the western bank of the Swartkops River (Haigh, 2002). Elevated nitrate and nitrite concentrations have also been found below the residential areas of Despatch and Redhouse and have been thought to originate from leaking sewer pipes and conservancy tanks (Maclear, 1995). For more clarity on historical groundwater data, elevated ammonium levels has been recorded at boreholes located within close proximity to Nivens Bridge (Kruisriver: Gubb & Inggs, 79.4 – 105.7 mg NH_4^+ l^{-1}), Despatch WWTW (Wagens Drift: 97.1 – 105.7 mg NH_4^+ l^{-1}), KwaNobuhle WWTW (Rioolwerk – Cape Good of Good Hope ponds: 32.7 – 51.2 mg NH_4^+ l^{-1}), and Redhouse (12.8 – 31.9 mg NH_4^+ l^{-1}), whereas elevated dissolved inorganic phosphorus levels have previously been recorded at Despatch WWTW (Wagens Drift: 9.6 – 30.3 mg DIP l^{-1}) (http://www.dwa.gov.za/iwqs/wms/data/M_reg_WMS _boreh.htm).

With several possible sources of contamination to the primary aquifer and therefore the water of the Swartkops River and estuary, it is recommended that future studies investigate the impact of groundwater on the waters of Swartkops River and estuary. The monitoring exercise which was conducted in 1993 has already identified routine sampling points of the primary aquifer (Haigh, 2002).

Chapter 5: References

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APPENDIX A: NEWSPAPER ARTICLES

1

Sewage flows in streets

Posted by: Saving Water SA (Cape Town, South Africa) - partnered with Water Rhapsody conservation systems – 02 April 2010

A new warning has been issued to the public not to swim or fish in the Swartkops River as the water quality has deteriorated further.

Last month, the 2010 Redhouse River Mile had to be moved from the Swartkops River to Cannonville on the Sundays River due to dangerously high levels of pollution.

Swartkops River Mouth. Photo by Graham Hobbs.

Jenny Rump of the Zwartkops Trust said yesterday that effluent had been flowing into the river for days because broken sewerage pipes had not been repaired. "The trust has received complaints from the residents of Aloes and Wells Estate that sewage has been running past their houses.

"It has been flowing in their streets for days now," added Rump, The Herald General Motors Citizen of the Year. "This is dangerous for the people's health. They have to live with the sewage smell for days," she said.

Rump said the Swartkops River, popular among swimmers and anglers, attracted a lot of visitors over the Easter weekend. Some of the stormwater drains that run through Wells Estate, which borders the sea, were broken, Rump added.

The deluge of litter that comes from the drains also joins the Swartkops River, causing two types of pollution.

"Sewage sometimes also bubbles onto the streets of Motherwell, apparently from collector stations where pumps have broken down, and then finds its way via the stormwater drains into the river," she said.

Municipal spokesman Kupido Baron could not comment last night on the latest pollution development at the Swartkops River. "All I know is that we have been working on the issue since the time the River Mile event scheduled to take place there had to move.

"I know there is a strategy in place, and we have weekly testing of the bacteria level at the river," he added without disclosing the nature of the plan.

Source: **Weekend Post**

April 2nd, 2010 | Tags: Motherwell, pollution, sewage, stormwater, Swartkops River, water quality | Category: Environment

2

Pollution of Bay's rivers must be stopped

03 November 2011

I WANT to commend the Zwartkops Conservancy for laying a criminal complaint against the metro for repeatedly failing in its duty to prevent pollution in the Swartkops River ("Action over pollution", October 27).

This matter has now been coming on for months (if not years) and needs to be resolved now, not tomorrow. It is a disgrace to the metro that the legendary Redhouse River Mile will once again not be swum in the Swartkops, after it having a year to rectify the situation.

But this is still the same ongoing pathetic saga of our metro – never responding to anything!

The summer season is upon us, and the Swartkops and Chatty rivers will once again be the playground for many a child living next to its banks. I shudder to think which consequences it holds for their health!

Please Mr Mayor and our new acting municipal manager, stop sewage from pouring into our beautiful rivers. Water is our source of life and needs to be conserved.

Please fix all the faulty pump stations that need to be repaired. I am convinced metro officials are fully aware of what is happening and what needs to be done to prevent a catastrophe.

We, as residents of this beautiful metro, demand action now!

DA councillor Brenda Matthee, member of the public health standing committee

3

Disaster as Swartkops River fouled again

28 February 2012

JUST as the Markman Canal was really looking much better after last year's breakdown of both the Studebaker and Aloes pump stations in Markman, it has happened once again.

On Saturday February 11, the pump at Studebaker again broke down and was only repaired four days later, with a temporary electrical cable!

Needless to say, the Markman Canal, which empties into the Swartkops River is once again in a terrible mess!

Both raw sewage and the dark, oily industrial pollution from nearby industries were gushing into the canal for four days! I have requested the infrastructure and engineering directorate to tell us when the electrical faults at the Studebaker and Aloes pump stations will be permanently repaired and maintained, and also suggested back-up generators be kept in case of another emergency.

I also eluded to the fact that the residents of this metro have the right to be kept informed of the e-coli count in the river on a regular basis.

The Bill of Rights in South Africa's constitution (Act 108 of 1996) Section 24 states: "Everyone has the right a) to an environment that is not harmful to their health or well-being; and b) to have the environment protected, for the benefit of present and future generations, through reasonable legislative and other measures that: i) prevent pollution and ecological degradation; ii) promote conservation; and iii) secure ecologically sustainable development and use of natural resources, while promoting justifiable economic and social development".

As a member of the public health standing committee, I am very concerned regarding the ongoing pollution that ends-up in the Swartkops River, especially through both the Motherwell, as well as Markman Canals.

This state of affairs infringes directly on the constitutional rights of many residents in this metro.

It is a crying shame and a very bad reflection on the metro that the Redhouse River Mile was once again held at Sundays River, for the third time, this year, due to the terrible state of the Swartkops River.

Something drastic has to be done as a matter of urgency so that our "sick" Swartkops estuary has a chance to recover and that the Redhouse River Mile could be brought back to Redhouse, where it belongs!

Brenda Matthee, DA PR public health councillor, NMBM

4

Buckets unemptied for weeks

20 April 2012

Neo Bodumela

RESIDENTS in Joe Slovo informal settlement in Uitenhage are threatening to throw the contents of their ablution buckets into the offices of their ward councillor, Kenneth Kohl, if the buckets are not collected.

The ward 48 residents said the buckets had not been emptied since the beginning of the month and they had resorted to dumping the waste into a nearby canal which fed into the Swartkops River.

This comes a week after protesters in Motherwell dumped the buckets' contents at Missionvale ward 32 councillor Sandra Fillis's office.

An angry resident known only as Meki, said they could no longer wait for the buckets to removed. "They had better come to fetch these buckets otherwise we will dump them at the councillor's office.

"We have had to dump these buckets at the [informal] dumping site and into canal," she said.

"We are tired of collecting these buckets ourselves because we are not the lorry they have hired to collect them and we are not getting paid by them to collect it.

"We will fight with the councillor to get them cleaned because we cannot sit with these buckets in our yards any more."

Community member Catherine Gillian said the waste was a health hazard for the children.

"These buckets are a danger to the kids and we are worried about it because what if they get sick? Never mind the smell. They have not been here [to collect the buckets] for the past three weeks," she said.

An official in Kohl's office, Manie Pienaar, speaking on behalf of the councillor, said they had tried over the past weeks to resolve the problem.

"And we have written and contacted the municipality since April 10 to try and sort out the issue, with no luck. We have been struggling and every time we get hold of somebody, they say that they are going to come and sort it out but they have never come out.

"These people are angry and they are threatening to dump the waste into the councillor's offices. We have even written to the mayor and nothing has come out from this."

Municipal spokesman Roland Williams said the relevant municipal department would be working through the weekend to resolve the problem.

"We appeal to communities that if at any stage they are experiencing delays in the delivery of basic services like water, electricity and sanitation, they immediately call the service delivery line at 0800205050".

5

Swartkops river water test results unsatisfactory

Tweet 0   g+1 9

22-03-2012 -
The overall results of the testing of samples of the Swartkops River taken at the Settlers Bridge, Swartkops Village, Motherwell Canal, Redhouse and Perseverance were "unsatisfactory," according to a report submitted to the Public Health Committee.

A total of 62 samples were taken.

The report says that storm water entering the river at the Motherwell Canal continued to be the source of pollution while the Markman Canal also led to the river being contaminated with high E.coli counts as two sewerage pumps at Aloes and the Studebaker pump stations were defective.

This led to raw sewerage being pumped into this canal and so into the Swartkops River. Both pumps have now been repaired.

As far as the North End Lake is concerned, the report states that a total of 89 samples were taken and the results indicated that the water was "unfit for full contact recreational purposes" adding that as the lake serves as a storm water catchment area it was "never intended to be used for full contact water sport".

Moves are now underway to rehabilitate the North End Lake as part of the urban renewal development of the precinct surrounding the Nelson Mandela Bay Stadium.

METRO MINUTES is an electronic up-to-date daily newsletter (Monday to Friday) covering decisions taken by the Nelson Mandela Bay Council and its committees, in addition to business developments within the Metro and occasionally in other parts of the Eastern Cape. The newsletter is free from October to the end of January. To receive this newsletter, send an e-mail to metminutes@iafrica.com

Eastern Cape, Nelson Mandela Bay, Port Elizabeth, Metro Minutes, Swartkops River, Settlers Bridge, Swartkops Village, Motherwell Canal, Redhouse, E.coli, Nelson Mandela Bay Stadium, Studebaker pump stations,

6

Clean up polluted Swartkops River

24 April 2012

E-mail your letter to The Herald

WITH reference to the article, "Buckets unemptied for weeks" (April 20), as an environmentalist involved with ongoing campaigning to stop pollution from entering the Swartkops River, this is again a huge setback and I am very angry!

This time around, the Motherwell and Markman canals are not the culprits, but the notorious Kat canal in Uitenhage!

The photograph taken by Eugene Coetzee that accompanied the article indicates two very young children carrying their unemptied bucket to dump the contents into the nearby Kat canal that is "supposed" to be a stormwater canal!

This is confirmed in the article by the angry resident, known as Meki, because the residents have no other choice as to use the canal due to the total lack of service delivery.

On February 23, I sent an e-mail to the Nelson Mandela Bay Municipality's director of water and sanitation to bring the many problems regarding pollution into the Swartkops River to his attention and asked when the metro's action plan to stop pollution flowing into the Swartkops River, as demanded from the Department of Water Affairs, will be brought before council.

After two follow-ups, I received a one-liner reply "that water and sanitation is looking into this matter and will provide you with a response shortly".

How long must we still wait while our environment is being run into the ground?

Where is everyone's constitutional right to live in an environment that is not harmful to their health and wellbeing?

In the attached photographs I took on Friday at the Kat canal's entrance into the Swartkops River, it can be seen that the metro's attempt to divert the sewage into the system is failing miserably! On the river side of the canal a huge sewage pond has formed that is absolutely disgusting.

What else must happen – must a person first die of disease – before the metro addresses this problem?

Please, Mr Mayor, our metro is being run into the ground.

Please request all your relevant departments involved to do something, and do something urgently to stop this terrible mess!

As a DA councillor on the public health committee, I will not stop campaigning that our beautiful Swartkops River and estuary be restored to its former unpolluted glory.

Brenda Matthee, DA PR councillor and member of the public health standing committee, NMB

7

Officials quizzed as investigation into pollution nears end

24 April 2012

Brian Hayward

WITH the Green Scorpions investigating the pollution of Nelson Mandela Bay's rivers, councillors yesterday grilled top officials over why they were not acting to stem the problem.

The quizzing of public health executive director Dr Mamisa ChabulaNxiweni and her counterpart in infrastructure and engineering, Ali Shaidi, took place at a municipal public accounts (MPAC) committee meeting.

The Swartkops, Baakens and Chatty rivers were most polluted, councillors heard, mainly by effluent from communities which were not serviced properly, as well as from toxic waste from the Markman industrial area.

Chabula-Nxiweni said that although the Green Scorpions investigation into which industries were to blame was progressing well, the unit had yet to brief her on the outcome.

But she did reveal: "The Green Scorpions have identified the culprit companies. They are determining whether the companies are legitimately positioned or not.

"They will give us their report when they are done with the investigation.

"On Thursday we want to meet with the officials responsible. The main cause of the pollution is sewage, as well as broken [sewerage] pumps." Shaidi said a delay in his directorate compiling its part of the report was the reason why it had not yet been delivered, although this would be done by the next MPAC meeting.

He said another reason for the pollution was inadequate and aging infrastructure, which required an upgrade costing at least R30-billion.

Councillor Joy Seale (ANC) asked what steps had been taken to stem the pollution since the report had been requested last November.

Shaidi said a multipronged approach had been formulated, including getting relevant officials in both directorates to liaise closely with councillors about which areas were most problematic.

Councillor Leon de Villiers (DA) blasted the directors, saying Uitenhage residents had been forced to dump bucket-system sewage into a river after the sanitation department failed to collect it.

The grilling of the directorate heads, summoned to address councillors face to face, was a result of a new approach which entails stringent and rigorous questioning,

MPAC councillors from around the province attended a three-day course at Pine Lodge in the Bay last week held by the Association of Public Accounts Committees (Apac) as part of Operation Clean Audit.

Another part of the strategy is a decision to caucus as a committee instead of splitting into party caucuses.

8

Warning on eating of Swartkops fish

March 6, 2014 | Filed under: News, | Posted by: Allan Williams

TOP marine biologists warned yesterday against eating fish from the polluted Swartkops River mouth.

This follows an independent pollution study done by Nelson Mandela Metropolitan University (NMMU) marine biologists which showed that fish from the area contained alarming amounts of dangerous metals.

NMMU marine biology lecturer Dr Nadine Strydom said the study had found extremely high concentrations of cadmium and, to a lesser extent, lead in the tissue of certain species of fish in the estuary.

"The levels measured exceed international safe standards for food," she said. "My research team discovered fish with livers that were discoloured and had formed ulcers." - Riaan Marais

9

From cleanest to dirtiest water

March 14, 2014 | Filed under: LifeStyle, | Posted by: Allan Williams

‹ Winning couple prove a popular choice VW AutoPavilion celebrates decade ›

AS AN estuarine angler, I have the immediate choice of four estuaries to fish in and they are no more than an hour's drive from my home. The same goes for my hunting passion. Here in the Eastern Cape, the surrounds of Port Elizabeth, we are so privileged as we live in the hunting and fishing paradise of the world.

UNSAFE WATERS: Pollution in the Swartkops River, including dangerous heavy metals from nearby factories, has turned it into a health hazard

The Swartkops, which remains the river of choice is the closest but unfortunately is also polluted.

It is amazing to see how it has stood up to all this pressure and yet survives and is seemingly productive.

This would not be the case should the river not get its daily flush of sea water. A toxicology study of the Swartkops has now become common knowledge and the results are simply shocking.

The consumption of fish from this river should be avoided at all costs as harmful heavy metals are present in the flesh of these fish and over time will cause many health complications.

Samples taken from live fish have proven their toxic levels are higher than that which is considered safe for human consumption. There is no detoxing of these metals either as they accumulate in the body and eventually it is the straw that breaks the camel's back.

Heavy metals such as cadmium and lead causes flu-like symptoms with a sore throat, coughing, joint pain and diarrhoea. Eventually liver and kidney disease take hold.

Stricter measures need to be implemented to prevent industry from discharging these pollutants into the estuary by way of leaks from holding tanks, that allow stormwater to ferry them into the river system.

Some cases may also be deliberate actions where dumping directly into the stormwater canals takes place because of insufficient monitoring.

The Kromme River is still my favourite as it seldom disappoints and this I believe is because the system is actually an arm of the sea.

The fact that very little freshwater enters the Kromme allows species to frequent there that will not enter the other estuaries.

One of its primary bait species is the ink fish or "chummy", which are not fresh-water tolerant so they are in abundance in the Kromme. The Sundays and Gamtoos rivers are much the same, as they are fed with water from the agricultural irrigation schemes overflow.

These are essentially cob rivers as these fish are tolerant to freshwater, where they venture far in to the saline-free environment.

This was established with the Telemetry study conducted there a few years back. Transponders were surgically inserted into cob that were caught and released.

Their movements were monitored by listening stations strategically placed all the way up the Sundays and along our coast. Invaluable data has and is still collected about the movement of these fish. – Reel Time, with Wayne Rudman

APPENDIX B: HISTORICAL DATA RECORDS

Table 28: Industries and their respective wastewater disposal methods.

Industry	Area	Quantity Water Used - Source	Industry Type	Disposal Manner	Permitted Quantity	Permitted Quality
East Cape Tanning	UIT	500 $m^3 d^{-1}$ - UCC 170 $m^3 d^{-1}$ - KJ WWTW	Tannery	Evaporation ponds	370 $m^3 d^{-1}$	None
Gubb & Inggs	UIT	1500 $m^3 d^{-1}$ - UCC 450 $m^3 d^{-1}$ - river	Wool processing	Evaporation ponds	820 $m^3 d^{-1}$	None
Perseverance Wool Pullery	PS	273 $m^3 d^{-1}$ - DCC & DS WWTW 67 $m^3 d^{-1}$ - river	Wool processing	Evaporation ponds	127 $m^3 d^{-1}$	None
Cape of Good Hope Woolcombers	UIT	1137 $m^3 d^{-1}$ - UCC	Wool processing	Evaporation ponds	280 $m^3 d^{-1}$	None
Marina Sea Salt	PS	river	Salt works	Evaporation in salt pans		
Transnet Cuyler Manor	UIT	740 $m^3 d^{-1}$ UCC 40 $m^3 d^{-1}$ KJ WWTW	Transportation	Settling dams and alum then to river	-	-
Grahams Poultry	KR	-	Chicken products	Irrigated on land/ponds		
Industex	UIT	200 $m^3 d^{-1}$ - PECC	Towelling manufacturing	KJ WWTW and WasteTech	-	-
Gearmax	UIT	80 $m^3 d^{-1}$ - UCC 8 $m^3 d^{-1}$ - KJ WWTW (irrigation)	Car parts & machinery	Settling tanks and KJ WWTW	-	-
National Standard	UIT	800 $m^3 d^{-1}$ - PECC, UCC & springs	Wire manufacturing	KJ WWTW and reuse	-	-
Volkswagen	UIT	3300 $m^3 d^{-1}$ - UCC & PECC	Car manufacturing	KJ WWTW	-	-
Hella	UIT	220 $m^3 d^{-1}$ - UCC	Car parts & plating	KJ WWTW	-	-
Guestro Uitenhage Dorbyl	UIT	225 $m^3 d^{-1}$ - PECC, UCC & springs	Machinery	KJ WWTW	-	-
Tycon	UIT	2273 $m^3 d^{-1}$ - UCC	Tyre manufacturing	KJ WWTW	-	-
Union Cotton Mills	UIT	640 $m^3 d^{-1}$ - UCC	Cotton mill	KJ WWTW	544 m^3/d	-
Rocklands Poultry	UIT	450 $m^3 d^{-1}$ - UCC	Chicken products		-	-

Source: adapted from DWAF (1998). DCC = Despatch City Council, UCC = Uitenhage City Council, PCC = Port Elizabeth City Council, DS = Despatch, KJ = Kelvin Jones; UIT = Uitenhage, PS = Perseverance' KR = Kruistiver

Table 29: Annual historical record from the Department of Water Affairs showing the number of sampling events for dissolved inorganic nitrogen (DIN) determinations in the Swartkops River catchment.

	Estuarine sites								**Freshwater sites**					
	Settlers Bridge (SB)	**Swartkops Village (SKV)**	**Chatty River (CR)**	**Brickfields (BF)**	**Motherwell Canal (MWC)**	**Redhouse Yacht Club (RYC)**	**Bar None (BN)**	**Perseverance (PS)**	**Perseverance Bridge (PSB)**	**Van Schalkwyk Bridge (VSB)**	**Frans Claasen Bridge (FCB)**	**Nivens Bridge (NB)**	**Elands River (ER)**	**Groendal Dam (GD)**
1995											12	10	21	10
1996											1	27	7	9
1997	3		2	3		3						50	24	10
1998												44	21	12
1999												17	6	12
2000												16	16	11
2001	3	3				3	3					13	10	12
2002												11	7	12
2003												19	9	12
2004												18	9	12
2005											1	23	16	10
2006											2	22	21	11
2007									3			23	21	8
2008												9	8	6
2009					1			1	4	3	4	1	4	5
2010					6			6	6	7	7	8	7	9
2011					6			7	8	9	9	12	13	6
2012					4			5	6	7	8	12	12	1
2013					2			3	2	2	2	8	9	
Totals	**6**	**3**	**2**	**3**	**19**	**6**	**3**	**22**	**29**	**28**	**46**	**343**	**241**	**168**

Table 30: Annual historical record from the Department of Water Affairs showing the number of sampling events for dissolved inorganic phosphorus (DIP) determinations in the Swartkops River catchment.

	Estuarine sites								**Freshwater sites**					
	Settlers Bridge (SB)	**Swartkops Village (SKV)**	**Chatty River (CR)**	**Brickfields (BF)**	**Motherwell Canal (MWC)**	**Redhouse Yacht Club (RYC)**	**Bar None (BN)**	**Perseverance (PS)**	**Perseverance Bridge (PSB)**	**Van Schalkwyk Bridge (VSB)**	**Frans Claasen Bridge (FCB)**	**Nivens Bridge (NB)**	**Elands River (ER)**	**Groendal Dam (GD)**
1995								4			19	10	21	10
1996											1	27	7	9
1997	3		2	3		3						50	24	10
1998												44	21	12
1999												17	6	12
2000												16	16	11
2001	3	3				3	3					13	10	12
2002												11	7	12
2003												19	9	12
2004												18	9	12
2005												23	16	10
2006												22	21	11
2007									3			23	21	8
2008											1	10	11	9
2009					1			1	4	4	4	13	9	7
2010					6			6	6	7	7	9	7	9
2011					6			7	8	9	9	12	13	6
2012					4			5	6	7	8	12	12	1
2013					2			2	2	2	2	8	9	
Totals	**6**	**3**	**2**	**3**	**19**	**6**	**3**	**26**	**29**	**29**	**51**	**357**	**249**	**173**

Table 31: Annual historical record from the Department of Water Affairs showing the number of sampling events for *Escherichia coli* load determinations in the Swartkops River catchment.

	Estuarine sites								Freshwater sites					
	Settlers Bridge (SB)	**Swartkops Village (SKV)**	**Chatty River (CR)**	**Brickfields (BF)**	**Motherwell Canal (MWC)**	**Redhouse Yacht Club (RYC)**	**Bar None (BN)**	**Perseverance (PS)**	**Perseverance Bridge (PSB)**	**Van Schalkwyk Bridge (VSB)**	**Frans Claasen Bridge (FCB)**	**Nivens Bridge (NB)**	**Elands River (ER)**	**Groendal Dam (GD)**
1980		3		3		3		3						
1981		9		8		8		8						
1985		9		9		9		9						
1986		6		6		6		6						
1995	14	46	25	47	61	62	14	57				10		
1996	28	44	21	44	60	61	15	61			4	15		
1997	4		5		3	3	3	3				3		
1998														
1999														
2000	1	1		1	1	1		1						
2001	32	32		32	31	31		34						
2002	39	40		40	39	42		42						
2003	69	68		67	67	65		64	1					
2004	72	72		70	75	73		72						
2005														
2006									3		1			
2007									5	1	1			
2008					4			5	8	7	4			
2009					3			3	6	6	6			
2010					6			6	6	7	7			
2011					5			6	7	8	8			
2012					5			5	6	7	8			
2013	5	5	5	5	7	5	5	8	2	2	2			
Totals	**259**	**303**	**51**	**301**	**362**	**338**	**32**	**362**	**44**	**38**	**41**	**28**	**0**	**0**

Table 32: Freshwater flow recorded at Nivens Bridge 45 days prior to sampling.

Sampling date	18 SEP '12	20 NOV '12	12 FEB '13	21 MAY '13	14 AUG '13
	1.35	0.35	0.41	0.46	0.35
	1.33	0.38	0.41	0.41	0.55
	1.41	0.62	0.43	0.36	0.63
	8.90	0.72	0.42	0.31	0.45
	34.00	1.27	0.38	0.28	0.42
	21.57	1.24	0.32	0.26	0.48
	13.01	1.20	0.29	0.24	0.46
	8.94	1.17	0.28	0.23	0.44
	7.14	1.18	0.29	0.23	0.40
	5.24	5.24	0.29	0.23	0.39
	4.02	39.31	0.27	0.22	0.35
	2.97	298.92	0.24	0.20	0.32
	2.13	212.88	0.22	0.19	0.30
	1.64	361.74	0.21	0.21	0.30
	1.41	548.00	0.21	0.22	0.31
	1.34	183.82	0.20	0.22	0.30
	1.33	104.12	0.39	0.22	0.29
	1.32	59.57	0.24	0.22	0.30
	1.30	39.67	0.25	0.21	0.29
	1.28	27.71	0.23	0.21	0.29
	1.28	21.56	0.21	0.20	0.31
	1.27	18.13	0.20	0.20	0.29
	1.25	14.63	0.19	0.20	0.27
	1.21	13.41	0.17	0.19	0.25
	1.19	37.04	0.16	0.16	0.24
	1.16	33.98	0.14	0.15	0.25
	1.11	24.35	0.13	0.15	0.24
	1.05	18.39	0.11	0.25	0.23
	1.00	14.71	0.10	0.18	0.22
	1.02	11.71	0.09	0.20	0.21
	0.97	9.61	0.09	0.19	0.20
	0.99	7.60	0.09	0.18	0.20
	1.21	5.88	0.08	0.24	0.19
	1.28	4.96	0.07	0.24	0.18
	1.28	3.99	0.06	0.23	0.18
	1.24	3.37	0.06	0.23	0.17
	1.20	2.61	0.07	0.23	0.18
	1.17	2.25	0.07	0.23	0.16
	1.06	2.14	0.06	0.23	0.15
	0.97	2.09	0.06	0.22	0.15
	0.98	1.77	0.06	0.21	0.16
	0.95	1.43	0.07	0.20	0.17
	0.94	1.35	0.64	0.20	0.17
	0.94	1.35	0.35	0.20	0.16
	0.89	1.34	0.23	0.20	0.15
Range	0.89 – 34.00	0.35 – 548.00	0.06 – 0.64	0.15 – 0.46	0.15 – 0.63
Median/mean	1.28 (3.32 ± 0.90)	5.88 (47.75 ± 16.22)	0.21 (0.21 ± 0.02)	0.22 (0.23 ± 0.01)	0.27 (0.28 ± 0.02)

APPENDIX C: TREND ANALYSES

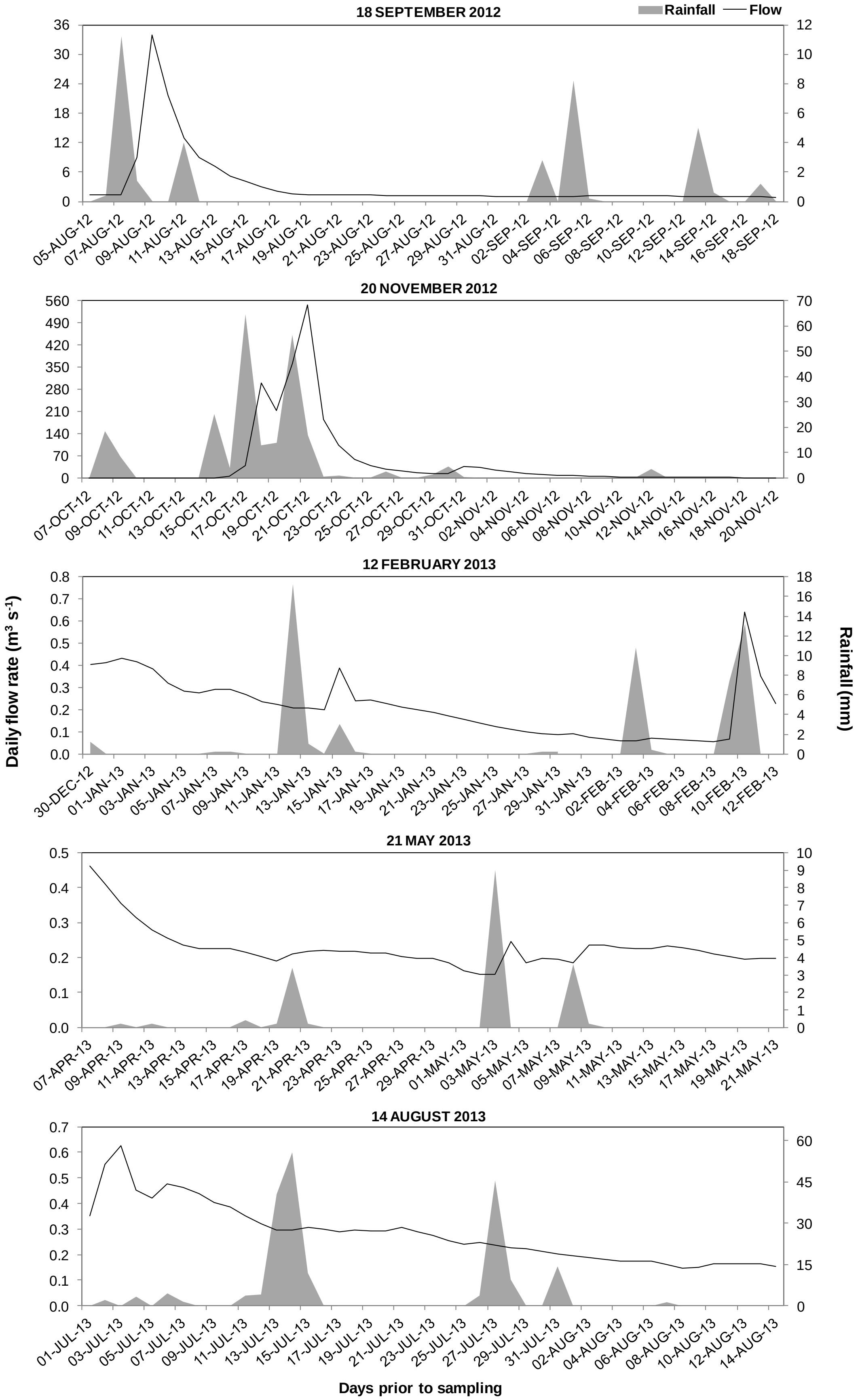

Figure 42: River flow rate and rainfall measured prior to sampling.

Table 33: Historical and current measurements of salinity (ppt) for three reaches of the Swartkops Estuary, including the Swartkops River.

Time series	Mean flow ($m^3 s^{-1}$)	Lower reaches	Middle reaches	Upper reaches	Swartkops River	Source
1979 – 1981	*no data*	30.4	22.0	20.6^{a}	15.7	Emmerson (1985)
1993 – 1994	1.52 ± 2.14	33.1	21.4	14.7^{b}	0	Scharler *et al*. (1997)
2012 – 2013	0.91 ± 0.83	34.4	22.1	8.4^{b}	1.7	This study (depth averaged)

Note: Lower reaches = Settlers Bridge, middle reaches = Brickfields, upper reaches = [a]Redhouse Yacht Club or [b]Bar None, Swartkops River = Perseverance. (Mean ± SD).

Table 34: Historical and current measurements of temperature (°C) reported for three reaches of the Swartkops Estuary, including the Swartkops River.

Time series	Mean flow ($m^3 s^{-1}$)	Lower reaches	Middle reaches	Upper reaches	Swartkops River	Source
1979 – 1981	*no data*	19.2	19.7	19.3^{a}	19.9	Emmerson (1985)
1993 – 1994	1.52 ± 2.14	17.4	18.7	20.0^{b}	19.5	Scharler *et al*. (1997)
2012 – 2013	0.91 ± 0.83	18.4	19.5	20.9^{b}	20.0	This study (depth averaged)

Note: Lower reaches = Settlers Bridge, middle reaches = Brickfields, upper reaches = [a]Redhouse Yacht Club or [b]Bar None, Swartkops River = Perseverance. (Mean ± SD).

Table 35: Most frequently observed pH ranges reported for the Swartkops Estuary.

Minimum	Maximum	Mean	Source
-	-	8.0	McLachlan (1972)
6.7	9.2	8.1	Emmerson (1985)
7.6	8.0	7.8	Scharler *et al*. (1997)
7.0	9.0	7.5 - 8.5	Binning (1999)
7.5	8.1	7.9	SRK Consulting (Pty) Ltd (2011)
7.1	8.8	8.0	This study

Table 36: Most frequently observed dissolved oxygen (mg l^{-1}) reported for three reaches of the Swartkops Estuary, including the Swartkops River.

Time series	Mean flow ($m^3 s^{-1}$)	Lower reaches	Middle reaches	Upper reaches	Swartkops River	Source
1979 – 1981	*no data*	7.4	7.2	7.2	7.0	Emmerson (1985)
1993 – 1994	1.52 ± 2.14	7.0	7.2	6.9	9.2	Scharler *et al*. (1997)
2012 – 2013	0.91 ± 0.83	7.1	6.2	9.2	6.4	This study (depth averaged)

Note: Lower reaches = Settlers Bridge, middle reaches = Brickfields, upper reaches = [a]Redhouse Yacht Club or Bar None, Swartkops River = Perseverance.

Table 37: Past and present total suspended solids (mg l^{-1}) reported for three reaches of the Swartkops Estuary, including the Swartkops River.

Time series	Mean flow (m^3 s^{-1})	Lower reaches	Middle reaches	Upper reaches	Swartkops River	Source
NOV '12	0.34	404	328	654	92.0	SRK Consulting (Pty) Ltd (2011)
FEB '13	0.50	71.2	50.8	56.0	26.5	This study (surface only)

Note: Lower reaches = Settlers Bridge, middle reaches = Brickfields, upper reaches = Redhouse Yacht Club, Swartkops River = Perseverance.

Table 38: Spatial and temporal variability in ammonium (NH_4^+; mg l^{-1}) recorded in the water column of the Swartkops Estuary and at two points of entry into the estuary, namely Chatty River and Motherwell Canal. The mean, miniumum and maximum values are indicated, including the number of data points in parentheses.

	1971 to 1992	1993 – 1994[a]	1997 – 2001[b]	2009 – 2011[c]	2012 – 2013[d]
SB	0.04 [10] 0 – 0.21	0.10 [62]	0.08 [6] 0.04 – 0.14		0.06 [5] 0 – 0.10
TC					0.15 [5] 0.02 – 0.50
SKV			0.08 [3] 0.04 – 0.12		0.08 [5] 0.02 – 0.20
BF		0.11 [68]	0.11 [3] 0.10 – 0.12		0.16 [5] 0.7 – 0.36
RYC	0.05 [10] 0 – 0.42		0.10 [3] 0.02 – 0.14		0.18 [5] 0.04 – 0.33
BN		0.09 [62]	0.09 [3] 0.06 – 0.12		0.30 [5] 0.01 – 1.0
PS		0.10 [14]		0.59 [16] 0.04 – 3.52	0.56 [13] 0.04 – 1.93
CR					1.06 [5] 0.26 – 2.32
MMC					1.00 [5] 0.41 – 2.40
MWC				1.43 [15] 0.04 – 12.30	1.18 [11] 0.04 – 3.85

Note: Shaded rows refer to points of entry into the estuary.
Site names: Settlers Bridge (SB), Tippers Creek (TC), Swartkops Village (SKV), Brickfields (BF), Redhouse Yacht Club (RYC), Bar None (BN), Perseverance (PS), Chatty River (CR), Markman Canal (MMC), Motherwell Canal (MWC).
Sources: [a]Hilmer (1984), [b]Scharler et al. (1997), [c]DWAF, [d]DWAF and SRK Consulting (Pty) Ltd (2011), [e]DWAF and this study.

Table 39: Spatial and temporal variability in total oxidised nitrogen (TOxN; mg l^{-1}) recorded in the water column of the Swartkops Estuary and at two points of entry into the estuary, namely Chatty River and Motherwell Canal. The mean, minimum and maximum values are indicated, including the number of data points in parentheses.

	1979 – 1981[a]	**1984[b]**	**1993 – 1994[c]**	**1997 – 2001[d]**	**2009 – 2011[e]**	**2012 – 2013[f]**
SB	0.24 [30] 0.01 – 1.12	0.07 [10] 0.02 – 0.21	0.18 [62]	0.09 [6] 0.04 – 0.14	0.03 [1]	0.06 [5] 0 – 0.13
TC						0.13 [5] 0 – 0.50
SKV	0.30 [30] 0.07 – 1.32			0.23 [3] 0.13 – 0.34	0.10 [1]	0.32 [5] 0.02 – 0.88
BF	0.51 0.12 – 1.65		0.21 [68]	2.08 [3] 0.15 – 4.69	0.10 [1]	0.71 [5] 0.25 – 1.15
RYC	0.61 [30] 0.11 – 1.85	0.15 [10] 0.02 – 0.44		0.22 [6] 0.05 – 0.52	0.01 [1]	0.80 [5] 0.30 – 1.62
BN			0.18 [64]	0.24 [3] 0.02 – 0.36		1.04 [5] 0.53 – 1.86
PS	0.85 [30] 0.11 – 2.66		0.34 [14]		1.24 [16] 0.08 – 2.70	2.03 [13] 0.05 – 5.09
CR				0.05 [2] 0.05 – 0.06	0.08 [1]	1.44 [5] 0.40 – 3.54
MMC						5.13 [5] 2.35 – 11.79
MWC					3.27 [14] 0.01 – 7.91	7.81 [11] 2.56 – 11.80

Note: Shaded rows refer to points of entry into the estuary.
Site names: Settlers Bridge (SB), Tippers Creek (TC), Swartkops Village (SKV), Brickfields (BF), Redhouse Yacht Club (RYC), Bar None (BN), Perseverance (PS), Chatty River (CR), Markman Canal (MMC), Motherwell Canal (MWC).
Sources: [a]Emmerson (1985), [b]Hilmer (1984), [c]Scharler et al. (1997), [d]DWA database, [e]DWAF and SRK Consulting (Pty) Ltd (2011), [f]DWAF and this study.

Table 40: Spatial and temporal variability in dissolved inorganic nitrogen (DIN; mg l^{-1}) recorded in the water column of the Swartkops Estuary and at two points of entry into the estuary, namely Chatty River and Motherwell Canal. The mean, minimum and maximum values are indicated, including the number of data points in parentheses.

	1983[a]	**1993 – 1994[b]**	**1997 – 2001[c]**	**2009 – 2011[d]**	**2012 – 2013[e]**
SB	0.11 [10] 0.02 – 0.43	0.28 [62]	0.17 [6] 0.09 – 0.24		0.11 [5] 0.05 – 0.16
TC					0.27 [5] 0.07 – 0.52
SKV			0.31 [3] 0.20 – 0.38		0.41 [5] 0.04 – 0.96
BF		0.31 [68]	2.18 [3] 0.24 – 4.79		1.02 [5] 0.41 – 1.29
RYC	0.20 [10] 0.02 – 0.87		0.29 [6] 0.07 – 0.66		0.98 [5] 0.52 – 1.85
BN		0.27 [64]	0.33 [3] 0.08 – 0.47		1.34 [5] 0.81 – 2.01
PS		0.43 [14]		1.95 [14] 0.45 – 6.22	2.59 [13] 0.09 – 6.04
CR			2.50 [2] 0.70 – 5.17		2.50 [5] 0.70 – 5.17
MMC					6.13 [5] 3.16 – 12.61
MWC				4.94 [13] 1.18 – 12.31	8.98 [11] 2.91 – 13.74

Note: Shaded rows refer to points of entry into the estuary.
Site names: Settlers Bridge (SB), Tippers Creek (TC), Swartkops Village (SKV), Brickfields (BF), Redhouse Yacht Club (RYC), Bar None (BN), Perseverance (PS), Chatty River (CR), Markman Canal (MMC), Motherwell Canal (MWC).
Sources: [a]Hilmer (1984), [b]Scharler et al. (1997), [c]DWAF, [d]DWA database and/or SRK Consulting (Pty) Ltd (2011), [e]DWAF and this study.

Table 41: Spatial and temporal variability in dissolved inorganic phosphorus (DIP; mg l^{-1}) recorded in the water column of the Swartkops Estuary and at two points of entry into the estuary, namely Chatty River and Motherwell Canal. The mean, minimum and maximum values are indicated, including the number of data points in parentheses.

	1979 – 1981[a]	**1984[b]**	**1993 – 1994[c]**	**1997 – 2001[d]**	**2009 – 2011[e]**	**2012 – 2013[f]**
SB	0.28 [30] 0.10 – 0.41	0.06 [10] 0.03 – 0.09	0.02 [62]	0.11 [6] 0.04 – 0.27	0.13 [1]	0.06 [5] 0.02 – 0.10
TC						0.08 [5] 0.05 – 0.11
SKV	0.55 [30] 0.19 – 0.93			0.36 [3] 0.09 – 0.77	0.75 [1]	0.18 [5] 0.07 – 0.34
BF	1.05 0.62 – 1.75 [30]		0.08 [68]	0.17 [3] 0.14 – 0.19	1.07 [1]	0.41 [5] 0.17 – 0.79
RYC	1.62 [30] 0.68 – 3.80	0.33 [10] 0.18 – 0.57		0.63 [6] 0.34 – 1.09	1.55 [1]	0.44 [5] 0.05 – 1.19
BN			0.12 [64]	1.05 [3] 0.50 – 1.36		0.98 [5] 0.39 – 1.41
PS	3.22 [30] 1.26 – 9.95		0.21 [14]		3.96 [15] 0.43 – 8.16	1.25 [13] 0.17 – 3.58
CR					0.57 [1]	0.40 [5] 0.26 – 0.73
MMC						0.14 [5] 0.09 – 0.22
MWC					0.33 [14] 0.02 – 1.13	0.19 [11] 0.05 – 0.33

Note: Shaded rows refer to points of entry into the estuary.
Site names: Settlers Bridge (SB), Tippers Creek (TC), Swartkops Village (SKV), Brickfields (BF), Redhouse Yacht Club (RYC), Bar None (BN), Perseverance (PS), Chatty River (CR), Markman Canal (MMC), Motherwell Canal (MWC).
Sources: [a]Emmerson (1985), [b]Hilmer (1984); [c]Scharler et al. (1997), [d]DWA database, [e]DWAF and/or SRK Consulting (Pty) Ltd (2011), [f]DWAF and this study.

APPENDIX D: ADDITIONAL DATA

Table 42: Trace metal measurements recorded in November 2012 and analysed by Talbot and Talbot Laboratories.

	Channel												Sources		
Site ID	**SB**		**TC**	**SKV**		**BF**		**RYC**		**BN**		**PS**	**CR**	**MMC**	**MWC**
Distance from the mouth (km)	**0.4**		**202**	**4**		**6.6**		**10**		**16.6**		**16.4**	**4.4**	**6.1**	**7**
Depth	**S**	**B**	**S**	**S**	**B**	**S**	**B**	**S**	**B**	**S**	**B**	**S**	**S**	**S**	**S**
Cyanide (total) ($\mu g\ l^{-1}$)	<20	<20	<20	<20	<20	<20	<20	<20	<20	<20	<20	<20	<20	<20	<20
Dissolved arsenic ($\mu g\ l^{-1}$)	1.2	2.8	1.7	<1	2.1	3.2	2.6	2.0	2.1	2.0	1.8	2.0	3.7	11.1	7.7
Dissolved mercury ($\mu g\ l^{-1}$)	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
Dissolved selenium ($\mu g\ l^{-1}$)	25.0	12.5	17.1	17.7	15.3	2.6	1.5	1.3	2.5	1.1	5.0	1.4	3.0	3.3	8.3
Fluoride ($\mu g\ l^{-1}$)	2 000	1 990	1 960	1 770	1 960	1 040	1 610	850	1 080	610	690	580	1 360	1 530	1 480
Total iron ($\mu g\ l^{-1}$)	520	500	600	400	590	3 280	1 960	270	320	4 870	3 910	2 660	1 480	1 650	150
Total aluminium ($\mu g\ l^{-1}$)	<300	<300	500	300	400	<300	400	500	600	600	700	300	1 400	1 200	300
Total cadmium ($\mu g\ l^{-1}$)	90	90	80	60	90	20	40	10	10	<10	<10	<10	10	10	10
Total chromium ($\mu g\ l^{-1}$)	<110	<110	130	<110	<110	<110	<110	<110	<110	<110	<110	<110	<110	140	130
Total copper ($\mu g\ l^{-1}$)	60	70	60	50	70	20	40	10	20	<10	<10	<10	10	10	10
Total lead ($\mu g\ l^{-1}$)	<50	<50	<50	<50	<50	<50	<50	<50	<50	<50	<50	<50	<50	<50	<50
Total zinc ($\mu g\ l^{-1}$)	80	90	80	80	80	70	70	50	60	60	50	50	70	70	60

Note: 'S' = surface, 'B' = bottom

Table 43: Spearman rank correlations (r) relating physico-chemical variables to nutrients and biological variables (n = 60). Coefficients that are significantly correlated ($p < 0.05$) are shaded in grey.

	pH	Temperature	Salinity	DO	TSS	Chl-*a*	NH_4^+	TOxN	DIP	Chl-*a*	Flagellates	Diatoms	Dinoflagellates	Cyanobacteria	Chlorophytes	Unknown	*Euglena*
Distance	0.12	0.21	-0.93	0.08	-0.37	0.51	0.78	0.81	0.89	0.78	0.38	0.34	-0.03	-0.13	0.34	-0.11	0.21
pH		-0.09	-0.03	0.66	-0.13	-0.04	-0.10	-0.09	-0.04	0.15	0.04	0.07	0.04	-0.31	-0.16	-0.18	0.17
Temperature			-0.39	0.04	-0.09	0.29	-0.06	0.05	0.11	0.54	-0.04	0.71	0.13	0.12	0.56	-0.08	-0.01
Salinity				-0.14	0.42	-0.46	-0.77	-0.80	-0.83	-0.83	-0.35	-0.46	-0.11	0.08	-0.39	0.07	-0.18
DO					-0.32	-0.20	-0.07	-0.05	-0.11	0.13	-0.22	0.28	0.36	-0.20	-0.04	0.02	0.12
TSS						-0.11	-0.31	-0.35	-0.11	-0.25	-0.08	-0.36	-0.34	-0.05	0.05	0.05	-0.18
NH_4^+							0.34	0.52	0.56	0.51	0.42	0.21	-0.41	-0.10	0.38	-0.29	0.07
TOxN								0.96	0.77	0.51	0.43	0.03	0.07	0.01	0.05	0.01	0.22
DIN									0.80	0.57	0.44	0.08	0.01	-0.02	0.14	-0.05	0.22
DIP										0.67	0.49	0.15	-0.20	-0.11	0.39	-0.07	0.21
Chl-*a*											0.44	0.59	0.00	-0.16	0.53	-0.33	0.13
Flagellates												0.04	-0.41	-0.12	0.18	-0.38	0.11
Diatoms													0.01	0.00	0.50	-0.17	0.12
Dinoflagellates														0.11	-0.20	0.20	-0.12
Cyanobacteria															0.15	0.43	-0.05
Chlorophytes																-0.08	0.20
Unknown																	-0.06

Table 44: Spearman rank correlations (r) relating physico-chemical variables to nutrients and faecal bacteria in the surface waters (n = 60). Water quality parameters that are significantly correlated ($p < 0.05$) are shaded in grey.

	pH	Temperature	Salinity	DO	TSS	Chl-*a*	NH_4^+	TOxN	DIN	DIP	Flagellates	Diatoms	Dinoflagellates	Cyanobacteria	Chlorophytes	Unknown	*Euglena*	*E. coli*	Enterococci
Distance	0.09	0.19	-0.93	0.22	-0.39	0.81	0.57	0.82	0.86	0.92	0.42	0.40	-0.05	-0.21	0.32	-0.18	0.26	0.52	0.32
pH		0.11	-0.07	0.76	-0.16	0.23	-0.10	-0.12	-0.11	-0.01	0.04	0.13	0.22	-0.37	-0.20	-0.14	0.20	-0.20	-0.13
Temperature			-0.38	0.24	-0.17	0.52	0.29	-0.06	0.08	0.06	-0.08	0.67	0.22	0.10	0.46	-0.13	-0.03	0.34	0.33
Salinity				-0.27	0.47	-0.83	-0.53	-0.79	-0.84	-0.83	-0.38	-0.54	-0.08	0.17	-0.31	0.17	-0.20	-0.57	-0.34
DO					-0.28	0.30	-0.05	0.01	0.07	0.10	-0.14	0.31	0.44	-0.35	-0.06	0.03	0.12	-0.19	-0.07
TSS						-0.31	-0.18	-0.31	-0.34	-0.18	-0.16	-0.46	-0.29	0.02	0.06	0.00	-0.20	-0.12	0.06
Chl-a							0.56	0.56	0.64	0.68	0.41	0.62	0.03	-0.30	0.42	-0.45	0.15	0.61	0.44
NH_4^+								0.41	0.61	0.58	0.41	0.24	-0.38	-0.10	0.45	-0.33	0.10	0.69	0.64
TOxN									0.95	0.81	0.48	0.16	0.02	-0.10	0.11	-0.05	0.29	0.46	0.22
DIN										0.85	0.49	0.22	-0.04	-0.09	0.25	-0.14	0.29	0.55	0.35
DIP											0.55	0.24	-0.20	-0.16	0.44	-0.14	0.25	0.53	0.39
Flagellates												0.08	-0.46	-0.11	0.28	-0.32	0.17	0.45	0.31
Diatoms													0.04	-0.10	0.39	-0.26	0.14	0.34	0.16
Dinoflagellates														0.09	-0.34	0.21	-0.16	-0.26	-0.20
Cyanobacteria															0.15	0.32	-0.08	-0.03	0.02
Chlorophytes																-0.08	0.26	0.41	0.50
Unknown																	-0.09	-0.38	-0.30
Euglena																		0.10	0.22
E. coli																			0.78

Table 45: Surface and bottom water quality characteristics recorded during bottom water hypoxic conditions at Brickfields and Bar None in February and August 2013.

		Salinity (ppt)	Chl-*a* ($\mu g\ l^{-1}$)	DO ($mg\ l^{-1}$)	NH_4^+ ($mg\ l^{-1}$)	TOxN ($mg\ l^{-1}$)	DIN ($mg\ l^{-1}$)	DIP ($mg\ l^{-1}$)	DIN:DIP	Flagellates ($cells\ ml^{-1}$)	Diatoms ($cells\ ml^{-1}$)	Chlorophytes ($cells\ ml^{-1}$)
Hypoxic conditions: 18 February 2013												
Brickfields	Top	21.10	23.68	3.81	0.12	0.28	0.40	0.84	0.50	13 935	6 609	1 831
	Bottom	23.18	10.66	2.71	0.20	0.22	0.42	0.74	0.60	11 457	2 973	0
Bar None	Top	1.44	158.06	9.93	1.46	0.85	2.31	1.41	1.60	6 812	5 822	5 883
	Bottom	1.98	58.61	1.30	0.55	0.20	0.74	1.32	0.60	5 827	2 787	4 459
Hypoxic conditions: 21 May 2013												
Bar None	Top	10.90	40.26	5.91	0.13	2.63	2.76	1.06	2.60	46 820	4 793	0
	Bottom	18.83	18.00	1.92	0.17	1.09	1.26	0.46	2.70	16 164	478	186
Normal conditions: 14 August 2013												
Bar None	Top	16.04	16.28	7.81	0.12	0.67	0.76	1.43	0.53	2 637	27	0
	Bottom	18.44	16.64	5.00	0.14	0.71	0.85	1.39	0.61	2 588	27	0

Table 46: A comparison of dissolved inorganic nitrogen and dissolved inorganic phosphorus levels recorded at representative monitoring sites of six Eastern Cape rivers from 1995 to 2013, including Elands River and Groendal Dam located within the Swartkops Catchment for comparison with the Swartkops River monitoring site at Nivens Bridge. (Percentage agriculture is indicated in parentheses).

	Great Fish (~10%)	Sundays (~ 6%)	Gamtoos (<5%)	Keiskamma (<5%)	Great Kei (<5%, subs)Ψ	Swartkops (~15%)
Dissolved inorganic nitrogen						
n	418	144	589	258	237	343
Mean (mg l^{-1})	0.58	0.20	0.10	0.32	0.25	1.20
SE	0.03	0.02	0.01	0.02	0.02	0.09
Median (mg l^{-1})	0.45	0.14	0.06	0.26	0.18	0.57
Minimum (mg l^{-1})	0.03	0.04	0.03	0.03	0.03	0.04
Maximum (mg l^{-1})	4.40	1.17	4.67	2.62	2.91	14.40
Annual variability	S***	S**	S***	S***	S**	S***
Monthly variability	S***	NS	NS	S***	NS	S***
Correlation (r)	0.61	0.34	0.22	-0.05	-0.10	0.28
Dissolved inorganic phosphorus						
n	432	144	615	262	238	357
Mean (mg l^{-1})	0.11	0.10	0.02	0.04	0.03	0.16
SE	0.01	< 0.01	< 0.01	< 0.01	< 0.01	0.02
Median (mg l^{-1})	0.08	0.09	0.02	0.03	0.02	0.04
Minimum (mg l^{-1})	< 0.01	0.01	< 0.01	< 0.01	0.01	0.01
Maximum (mg l^{-1})	2.20	0.34	0.23	0.49	1.00	3.08
Annual variability	S*	S***	S***	S***	S***	S***
Monthly variability	S***	S***	S***	S***	NS	NS
Correlation (r)	-0.40	-0.09	-0.58	-0.57	-0.56	0.51

Note: S, $p < 0.05$; S**, $p < 0.01$; S***, $p < 0.001$; $\alpha = 0.05$; Ψsubs = subsistence. "r" is calculated based on annual median values and refers to linearity with time. Shaded values indicate $p < 0.05$). Shaded values indicate $p < 0.05$)*

APPENDIX E: ADDITIONAL PHOTOS

Figure 43: Litter trapped within two canals and in a stormwater drain located in Uitenhage.

Figure 44: Litter found in the upper reaches of the Swartkops River downstream of Nivens Bridge and at selected sites of the Motherwell and Markman canals.

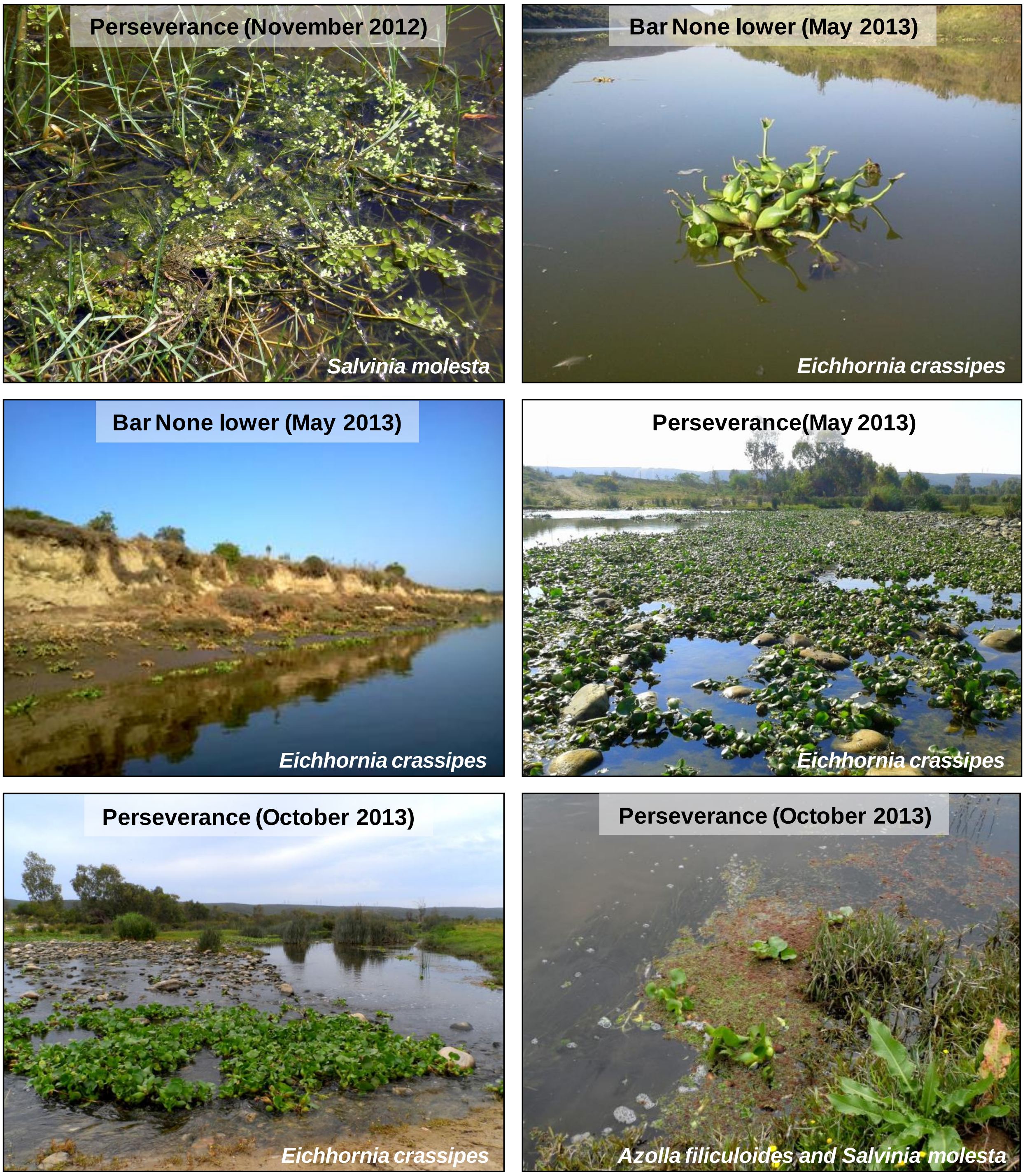

Figure 45: Floating macrophyte species encountered during 2012 and 2013.