Monitoring of Heavy Metals in the Bottelary River Using *Typha capensis* and *Phragmites australis*

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KEY WORDS

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ABSTRACT

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A preliminary investigation of heavy metal pollution in the Bottelary River with the focus on biomonitoring was conducted. *Typha capensis* and *Phragmites australis* were chosen as biomonitors due to their wide distribution and apparent ability to accumulate heavy metals in aquatic ecosystems. Plant, sediment and water samples were collected monthly at three different sampling sites along the Bottelary River from February to July 2005. The sites were chosen so as to compare the runoff from the farms (upstream of the tributary in the Bottelary River) with the water coming from a town and the Wastewater Treatment Works (tributary). The third site was just downstream of the first two.

Dissolved oxygen, temperature, pH, conductivity, ammonium, nitrite and nitrate were measured in water samples. Total nitrogen was measured in plants (Kjeldahl method). Cadmium, chromium, copper, iron, lead, manganese, nickel, vanadium, zinc and phosphate were measured in the water, sediments and plants. The concentrations of these nine metal elements were determined with an Atomic Absorption Spectrophotometer and the phosphate by a colorimetrical method.

The results showed that most of the element concentrations (Cd, Cr, Ni, Zn and P) in plants were higher than those in the sediments, and were much greater than those in the water. *T. capensis* was proved a more useful biological monitor as it accumulated most of the heavy metals; *P. australis* did however accumulate lead more readily. The roots consistently had higher metal concentrations than either the rhizomes or above–ground organs, except for lead in the stems of *P. australis* where concentrations were higher than in other organs. While chromium and nickel were higher in the tributary; copper, manganese and vanadium were higher upstream. Levels of cadmium, copper and zinc in water were above those legally acceptable. Over the study period, significant increases occurred in the concentrations of chromium, nickel, vanadium and zinc in plants.

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DECLARATION

I declare that *Monitoring of heavy metals in the Bottelary River using Typha capensis and Phragmites australis* is my own work, that it has not been submitted for any degree or examination at any other university and that all the sources I have used or quoted have been indicated and acknowledged by complete references.

Full name: Ying Ma

Date: 2 February 2005

Signed……………………….

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CHAPTER 1

Literature Review

Heavy Metals in the Environment — the Use of Plants to Monitor Heavy Metal Leads in Water

1.1 Introduction

Biomonitoring as indicative of the presence of pollutants is defined as the use of bio–organisms to obtain quantitative information on certain characteristics of the biosphere (Wolterbeek, 2002). Development of indicators of exposure is thus critical to evaluate risk from heavy metals and they are necessary as early warning systems for environmental deterioration (Anon a, 1983). It has the conceptual advantage that biotic responses may provide more direct measures of the biological significance of environmental contaminants (Markert, 1993). In the case of river metal contamination, utilizing macrophytes for biomonitoring may provide a relatively quick method for determining the spatial extent of metal contamination throughout a large area without the expense. Further, ecological risk assessments call for an understanding of the biological impacts of metal contamination, for which biomonitoring may provide more relevant information than do metal concentrations alone.

Plants play an important role in metal removal via filtration, adsorption, cation exchange, and through plant–induced chemical changes in the rhizosphere (Dunbabin & Bowmer, 1992; Wright & Otte, 1999). There is evidence that plants can accumulate heavy metals in their tissues. The use of these plants gave birth to a new technology termed phytoremediation, which is one of the most successful techniques that can be used to remove heavy metals. Phytoremediation is the use of plants to remove heavy metals or pollutants from the environment or to render harmless, hazardous material present in the water or soil (Salt *et al.*, 1998). The only problem with phytoremediation is that it needs to be done over a long period of time to clean contaminated soil or water properly. Phytoremediation of metal–contaminated water is readily achieved by the use of aquatic and terrestrial plants due to the non–bioavailability of target elements to other organisms. Terrestrial plants must first solubilize the target element in the rhizosphere and then have the ability to transport it to the aerial parts. There are no such problems in plants that grow naturally or are made to grow in or on an aqueous medium (Brooks, 1998).

Macrophytes may concentrate metals in their biomass to a higher level than their ambient environments, and “give a more time–integrated picture of contaminant concentrations” (Whitton *et al*., 1981). Therefore, macrophytes can be used in ecological surveys as *in–situ* bioindicators of water quality due to their ability to accumulate chemicals (Lewis, 1995). Rooted submerged macrophytes taking up pollutants represent the bioavailable, free–contaminant concentrations in the sediment interstitial water, as well as the contamination in the water column (Schrenk *et al*., 1998; Turgut & Fomin, 2002). They have been identified as a potentially useful group for bioremediation and biological monitoring (Biernacki *et al*., 1997; Salt *et al*., 1998).

1.2 Aim

1.2.1 General Aim

The use of plants to determine the degree of heavy metal contamination in water and sediments, in order to effectively monitor and provide possible recommendations to improve the water quality in the aquatic ecosystem of the Bottelary River. This mini–thesis represents the initial results in the particular study area.

1.2.2 Specific Aims

To quantify the extent of cadmium, copper, chromium, iron, lead, manganese, nickel, vanadium and zinc accumulation in aquatic plants growing in the Bottelary River, and examine relationships between water metal concentrations, sediment metal concentrations and plant metal content

To compare heavy metal concentration in roots, rhizome, stems and leaves

To compare *Typha capensis* and *Phragmites australis* as potential biomonitors of heavy metal pollution in the Bottelary River area

To evaluate the relative importance of contamination from the town (De Novo tributary from Kraaifontein) and from the farms (upstream)

1.3 Key Questions

- Which are the main heavy metals present in water, sediments and plants in the Bottelary River?
- Which heavy metals do *T. capensis* and *P. australis* absorb most effectively?
- In which part of the plants do heavy metals reach their highest concentrations?
- Which of these two macrophytes is more useful in monitoring heavy metals?
- Which pollutants come predominantly from the town (De Novo tributary) and which come from the farms (upstream)?

1.4 Heavy Metal Pollution in Water

Heavy metals are frequent waste products of anthropogenic activities and their emission often results in the contamination of the surrounding environment (Eeva & Lehikoinen, 2000). Industrial waste, geo–chemical structure of the catchment and mining create potential sources of heavy metal pollution in the aquatic environment (Lee & Stuebing, 1990). Under certain environmental conditions, metals may accumulate to toxic proportions and can cause ecological damage (Freedman, 1989). Heavy metals can even have effects on different aspects of water use, such as oxygen consumption by organisms in the environment (Ahern & Morris, 1999), water permeability (Rasmussen *et al*., 1995) and osmoregulation (Ahern & Morris, 1998).

1.5 Heavy Metal Treatments

Aquatic environments provide many ecosystem services to humans, yet they often receive high levels of pollutants from both point and non–point sources. The various conventional remediation technologies that are used to clean heavy metal polluted environments are; soil *in situ* vitrification, soil incineration, excavation and landfill, soil washing, soil flushing, solidification and stabilization electrokinetic systems. Each of the conventional remediation technology has specific benefits and limitations (Anon b, 1993; Anon c, 1997). Because analytical testing of chemical pollutants is time–consuming and expensive, there is great interest in finding environmental monitoring methods that are cost effective, fast and user friendly (Lovett–Doust et *al*., 1994a; Raskin *et al*., 1997). Furthermore, measuring concentrations of pollutants in water and sediments alone does not provide information on the potential impact of pollution on resident organisms (Lovett–Doust *et al*., 1994b).

As a potential low–cost remediation, phytoremediation refers to a diverse collection of plant–based technologies that use either naturally occurring or genetically engineered plants for cleaning contaminated environments (Cunningham *et al*., 1997; Flathman & Lanza, 1998). It consists of four different plant–based technologies each having a different mechanism of action for the remediation of metal–polluted soil, sediment, or water. These include: rhizofiltration, which involves the use of plants to clean various aquatic environments (Salt *et al*., 1995); phytostabilization, where plants are used to stabilize rather than clean contaminated soil (Salt *et al*., 1995); phytovolatilization, which involves the use of plants to extract certain metals from soil and then release them into the atmosphere through volatilization (Suszcynsky & Shann, 1995); and phytoextraction, where plants absorb metals from soil and translocate them to the harvestable shoots where they accumulate (Kumar *et al*., 1995).

The use of metal–accumulating plants to clean soil and water contaminated with toxic metals is the most rapidly developing component of this environmentally friendly and cost–effective technology. Aquatic macrophytes, being the principal mechanism for metal uptake and adsorption through roots, have been used during the last two decades for metal removal from water, competing with other secondary treatments (Denny & Wilkins, 1987). For aquatic macrophytes that possess roots but do not have a close physical association with sediments, the water is undoubtedly the principal source of elements. The uptake and subsequent release of trace metals during transmission to organisms of higher trophic levels represents a pathway detailing the cycling of trace metals in aquatic ecosystems.

1.6 Monitoring Heavy Metals in Water and Sediments

Concentrations of metals can be analyzed in water samples, sediment samples and biota taken from study sites. Using water samples to assess water quality in terms of metal contamination, and also to identify pollution sources, can be inconclusive due to fluctuations in dissolved constituents within short time intervals (Forstner, 1980). These fluctuations are due to daily and seasonal changes in water flow, changing pH and redox conditions, salinity, temperature and detergent concentrations in the water. Metal concentrations found in water are usually very low, resulting in analytical difficulties (Rainbow, 1995).

Sediment analysis is more useful to detect pollution problems and sources, especially for contaminants that are rapidly absorbed by particulate matter and consequently would not remain in water samples for long. When river flow is low, particulate matter suspended in the water settles to the riverbed and is incorporated into the sediment. Factors such as water flow (especially after a high discharge), and particle size need to be taken into account, when sediments are used to monitor metal pollution in rivers. For example a lower degree of contamination would be measured after a high discharge due to erosion of the river bed sediments (Forstner, 1980).

1.7 Plants as Heavy Metal Biomonitors

Aquatic macrophytes can absorb and accumulate various metals from the aquatic environment (Dean *et al*., 1972; Dietz, 1973; Erikson & Mortimer, 1975; Tremp & Kohler, 1995). They can be reliable indicators of metal pollution in freshwater ecosystems (Abo–Rady, 1980; Franzin & McFairlane, 1980; Mortimer, 1985; Ray & White, 1976). Macrophytes take up pollutants via their roots submerged in sediments (Biernacki *et al*., 1996; Salt, 1998) and also absorb chemicals from the water column through their leaves (Biernacki *et al*., 1996). The final metal concentration in plants is usually significantly larger than in the water column (Mal *et al*., 2002). Moreover, measuring components of basic plant traits, such as growth, survival, or reproduction is typically quite simple and repeatable. Therefore macrophytes are potentially useful for biomonitoring environmental concentrations of metals (Biernacki *et al*., 1996; Salt, 1998). The degree of metal uptake by some plants is largely dependent on the type of metal and the plant species involved (Mortimer, 1985).

The term ‘biomonitor’ describes species that accumulate trace metals or other substances in their tissues and therefore can be used to monitor the bioavailability of these substances in a particular environment. They can be used to establish geographical and temporal variations in the bioavailabilities of heavy metals in the aquatic environment. They can also offer time–integrated measures of the bioavailable levels of heavy metals which is a feature that makes them superior when compared to water or sediment samples (Rainbow, 1995).

Biomonitors generally accumulate trace metals to concentrations that are relatively easy to measure (Rainbow & Phillips, 1993), since they concentrate metals continuously. These metals often attain several orders of magnitude above ambient water concentrations. When sediments and water samples are analyzed for metals, it is the total metal concentration that is measured. However, the total metal concentration may not be a definitive way of analyzing whether metal uptake and toxicity are causing a problem at a site (Lovett–Doust *et al*., 1994b). Factors other than total metal concentration alter the free metal ion concentration of freshwater: pH, salinity (Turner *et al*., 1981) and concentration of humic materials are three such factors (Mantoura *et al*., 1978). To examine whether factors controlling metal speciation and uptake at a site (e.g. metal concentration, salinity, temperature, pH) are combining to produce toxic effects or metal accumulation in biota, it is necessary to measure aspects of the biota themselves, as well as metal concentrations in the sediment (Thawley *et al*., 2004).

Essential characteristics required of biomonitors include the capacity to accumulate pollutants without being killed by the encountered levels, and sedentariness in order to be representative of the area. Furthermore, biomonitors should be abundant in the study area, sufficiently long–lived and be of a reasonable size to provide enough tissue for analysis. They should be easy to sample and possess high concentration factors for the elements investigated (Phillips, 1977). *T. capensis* and *P. australis* were selected as possible biomonitors of heavy metal pollution in the Bottelary River area. This study suggests the possibility of using the bioconcentration of heavy metal ions in aquatic plants as a means for monitoring water resources affected by point and non–point source pollution.

1.8 Heavy Metal Accumulation by *Typha* Species and other Macrophytes

Typha capensis (cattail/bulrush) is a monoecious, tufted, perennial aquatic weed that belongs to the family Typhaceae and grows to up to 2m or more in height. The leaves are strap–like, twisted and spongy. It spreads and grows very fast in stream banks, ponds, canals and marshes throughout southern tropical Africa (Goldblatt & Manning, 2000). This plant can tolerate some degree of salinity, and both acidic and alkaline environments. *Typha* species have important properties such as a high natural productivity and they have the ability to accumulate large amounts of heavy metals and nutrients (Anon d, 2005).

Demirezen and Aksoy (2004) investigated concentrations of heavy metals (Cd, Cr, Cu, Ni, Pb and Zn) in bottom sediments, water, *Typha angustifolia* and *Potamogeton pectinatus* in Sultan Marsh, Turkey. They found that the tissues of *T. angustifolia* accumulate more heavy metals than the tissues of *P. pectinatus*. Considerably higher concentrations of cadmium were found in the emergent *T. angustifolia*, rather than in the submerged *P. pectinatus*. They also found that nickel and lead were accumulated by plants at a higher rate from bottom sediments than from water. Leaves of *T. angustifolia* accumulated less heavy metals than the corresponding roots. Therefore, they concluded that all plants could be used as biological indicators while determining environmental pressures; however *T. angustifolia* was proved to be a more appropriated biological indicator (Demirezen & Aksoy, 2004).

Mortimer (1985) treated the freshwater aquatic macrophytes of the Qttawa River as heavy metal monitors. The ability of freshwater aquatic vascular plants to accumulate heavy metals was examined in some detail during his five–year study. The differences in the uptake rate of metals by aquatic plants could depend on factors such as the plant species and the type of metal, as well as the seasonal growth rate changes and the metal ion being absorbed (Mortimer, 1985).

The concentrations of the heavy metals (Fe, Mn, Ni and Zn) in the water environment and stream sediments have also been investigated. Similarly their accumulation by two emergent aquatic plants, *Arundo donax* and *Typha capensis* under acidic and alkaline conditions in water and sediments of a stream polluted by goldmine and industrial effluents, at four selected localities in the Elsburgspruit Catchment area, was studied. It was evident from other related investigations that heavy metals could have deleterious effects on the composition and the presence of certain stream biota. Changes in the pH of the water would have a direct bearing on their water solubility, as well as the deposition capacity of such metals in the substrata of the standing and flowing water ecosystems (Van der Merwe *et al*., 1990).

The results showed that the deposition of zinc, manganese and nickel, but not iron, increased as the water became more alkaline. High concentrations of these metals were present in the tissues of both species, but they differed in their capacity to accumulate metals. There were no clear–cut similarities in the absorption capacity of the metals between the two plants, with zinc and iron being better accumulated under alkaline conditions by *T. capensis*, but with the reverse being the case for *A. donax*, where the absorption of manganese and nickel was more successful. The sequence in efficiency of *T. capensis* to accumulate copper in its various organs followed the same pattern as for iron and nickel, namely in descending order roots, white stems, green stems and green leaves. The concentrations in the organs of *T. capensis* confirm the ability of the roots to accumulate and be treated as the most important storage organ. Chromium was the only metal, which was accumulated in extremely low concentrations in all the organs of *T. capensis*, with the roots being the most efficient in the bioconcentration of this metal. Iron absorption was less affected by the pH of the water (Van der Merwe *et al*., 1990).

Manios *et al*. (2003) investigated removal of heavy metals from a metalliferous water solution by *Typha latifolia* plants. Four groups were irrigated with a solution containing different concentrations of copper, nickel and zinc. At the end of the ten week experimental period, the amounts of all three metals removed from the irrigation solution were substantial. The total amount of metals removed by the plants was considerably smaller than that of the substrate, due mainly to the small biomass development. A single factor ANOVA test (5% level) indicated that the build up in the concentration of metals in the roots, leaves and stems was due to the use of metalliferous water solution and not from the metals pre–existing in the substrate. The removing ability of the roots, leaves and stems was less than 1% (Manios *et al*., 2003).

Wetlands species are capable of removing large quantities of heavy metals from water. Most of the metals are stored in the sediments and some elements are taken up by plants. Metals have different behaviour patterns, for example copper and zinc can remain in the sediments only temporarily, whilst iron and lead tend to be strongly bound to the sediments. *T. latifolia* has similar patterns of uptake for a number of heavy metals. In *T. latifolia* the accumulation of heavy metals is higher in the roots and shoots (Quick, 1987).

The possibility of *Typha* species utilization to investigate waste–water treatment was done by Junrungreang and Jutvapornvanit at the Land Development Department (Bangkok, Thailand) in 1996. They found *Typha* species could produce large quantities of biomass, the total annual productivity being approximately 56.6 ton/ha. They indicated that the uptake ability of the *Typha* species above ground differed to that of below ground. *Typha* species have a rather high capacity to absorb heavy metals such as copper, manganese and zinc. They detected that the elements accumulated more in broad–leaf *T. latifolia* than narrow–leaved *T. angustifolia*. They also noted that seasonal dynamics affected the concentration of chemical elements and biomass in *Typha* species. They concluded that *Typha* species can be used for water purification and they should be harvested during summer to remove metals from waste–water (Anon d, 2005).

There was an investigation of cadmium, lead and zinc tolerance in four populations of *T. latifolia* raised from seed collected from metal–contaminated and uncontaminated sites. Metal concentrations in natural plant populations showed that cadmium, lead and zinc in the leaves and associated soil–sediments were maintained at low levels, while in the roots at relatively high levels accumulated, although concentrations varied widely. Seedlings from metal–contaminated populations accumulated considerably more metals in their roots than the uncontaminated population, in a pot trial (Ye *et al*., 1997a).

Taylor and Crowder (1983) investigated the uptake and accumulation of copper, nickel and iron by *T. latifolia* grown in a solution culture. They observed that the concentration of copper, iron and nickel in leaf and root tissue of *T. latifolia* grown in the solution culture containing 5–100 µg/g copper, 10–100 µg/g iron, 10–150 µg/g nickel, were correlated with concentrations of the same metals in the growth medium. The leaves of *T. latifolia* were tolerant to high concentrations 467 ± 50 µg/g of nickel within the plant tissue without signs of phytotoxicity. Concentrations of copper were up to 127 ± 28 µg/g in leaves of *T. latifolia* and toxicity symptoms appeared when copper reached 80 µg/g in the leaves. Iron showed high concentrations in the leaves, but there was less in the roots (Taylor & Crowder, 1983).

McNaughton *et al*. (1974) found that *Typha* species from metal contaminated and non–contaminated environments can grow equally as well in substrates of elevated metal concentrations. There were also no differences noted in *T. latifolia* and *P.*

australis plants originating from either non–contaminated or contaminated environments (McNaughton *et al*., 1974).

1.9 Heavy Metal Accumulation by *Phragmites* Species and other Macrophytes

Phragmites australis (common reed, Poaceae) is one of the most widely distributed species in the world. It is a robust, reedlike perennial that grows up to three meters tall. The leaves are cauline and lanceolate. It spreads and grows in marshes, streams and seeps (Glodblatt & Manning, 2000). It is a valuable soil reclamation plant because it is probably indigenous to South Africa, and therefore flourishes permanently and spreads easily without posing a threat to the natural biota. It thrives in watercourses, decreasing water flow rate and increasing sediment deposition, and it will grow through after being covered in sediment from floods (O'Keeffe, 1986). This plant can withstand extreme environmental conditions, including the presence of toxic heavy metal contaminants, such as cadmium, lead and zinc (Schierup & Larsen, 1981a). In the last two decades, *P. australis* has been widely used in constructed wetlands for treatment of industrial wastewaters containing metals (Dunbabin & Bowmer, 1992). With respect to *P. australis*, Ye *et al*. (1997b) studied the uptake of cadmium, lead and zinc by plants raised from seeds collected from a contaminated and a clean site, using single metal treatments. He indicated that *P. australis* could withstand extreme environmental conditions, including the presence of toxic heavy metal contaminants. *P. australis* has been used for many years in constructed wetlands for treatment of industrial wastewaters containing metals (Ye *et al*., 1997b).

A study of heavy metal uptake by marsh plants in Lake St. Clair revealed that increased plant growth during the growing season resulted in increased heavy metal concentrations in several plant species (Mudroch & Capobianco, 1978). The result showed that plant concentrations of nickel, chromium, and copper increased during vegetative growth of *T. latifolia*; cadmium and zinc increased during vegetative growth of *Myriophyllum heterophyllum*; cobalt and lead increased during vegetative growth of *Chara* spp. Increased heavy metal uptake resulting from increased plant biomass production appears to be real and may significantly contribute to heavy metal cycling by marsh plants, especially *P. australis*.

P. australis plays an important role in the cycling of nutrients. Lindsay (1972) found the major proportion of heavy metals is bound rapidly in the sediments, where chemical parameters such as organic matter content, pH and cation exchange capacity (CEC) determine the type and stability of metal precipitation and thus availability of these elements to rooted macrophytes.

Chiaudani (1969) investigated the distribution of copper in *P. australis* from different lakes in Italy, and found that most of the copper accumulated in the rhizome and roots. This distribution pattern has been observed in a number of other species, which suggests that only a small amount of heavy metals taken up by plant roots are translocated to the parts of the plant above the ground. Sharpe and Denny (1976) found that macrophytes take up heavy metals mainly through roots although the uptake through the leaves may also be significant.

Schierup and Larsen (1981a) investigated *P. australis* cycling of zinc, copper, lead, and cadmium in two Danish lakes, one being polluted, the other not. They found that uptake of these heavy metals by population of *P. australis* was greater in the unpolluted lake than in the polluted lake. They indicated that a differentiation in uptake could be established between real uptakes of heavy metals and chelation, compared to active functional groups in the cell walls and phenolic OH groups. The difference in uptake was dependent on edaphic factors rather than on the amount of heavy metals present in the sediment. Two of the edaphic factors thought to influence heavy metal availability are sediment pH, and sediment oxidation–reduction (redox). The status was sufficiently different between the two lakes to explain the difference in uptake, i.e. uptake increased as redox increased or pH decreased.

Auclair (1979) also found that the tissue concentration of several emergent hydrophytes of which *P. australis* is one dependent on edaphic factors. He indicated plant growth affected nutrient and heavy metal tissue concentrations as well.

The availability and uptake of heavy metals were greater in Lake Hampen than in Lake Sorteso, possibly as a result of differences in the prevailing chemical sediment conditions, e.g. pH, cation exchange capacity (CEC) and organic carbon content. Copper, lead and cadmium were accumulated mainly in the roots of *Phragmites*, with a lower in concentration in the rhizomes and above ground parts. Zinc accumulated both in the roots and in the aerial plant parts. In Lake Hampen the distribution of heavy metals between the sediment and different parts of *Phragmites* was similar, irrespective of whether concentration or actual amounts were used as a basis of comparison. In Lake Sorteso, a larger quantity of heavy metals was accumulated in the rhizomes than in the roots and above ground biomass (Schierup & Larsen, 1981a).

Schierup and Larsen (1981b) found that the concentrations of zinc and copper in the leaves were more or less constant during the study period. Although heavy metals are generally considered as being rather immobile in plants, the two essential micronutrients, copper and zinc may be considered to be mobile in plants. The decrease in the concentration of both metals, after the growth season, may be due to translocation of these elements to the rhizomes for storage, or to leaching. The concentration of lead in the leaves increased during and after the growth season (Schierup & Larsen, 1981b).

Iron was clearly the best accumulated of all metals in *P. australis* from Florida Lake in South Africa. This was followed by manganese and zinc. Copper followed by chromium was accumulated in the lowest concentrations. In *P. australis* from Blesbokspruit, iron was also found to be the best bioconcentrated in various organs. Copper and manganese were the two metals best taken up from the sediments by this plant. Nickel, zinc, and lead occurred at lower concentrations in the different organs analyzed. The roots were clearly the most efficient in the accumulation and storage of all seven metals investigated with iron, manganese and copper concentrations as in the Florida Lake, being higher in the plants than in the sediments. The white stems embedded in the substrate contained the second highest concentration of the metals after the roots. The green stems and leaves were less efficient in the accumulation and storage of metals (Schoonbee *et al*., 1995).

Weis *et al*. (2003) found that upper leaves of *P. australis* and *Spartina alterniflora* tended to have lower levels of metals than lower leaves. Since lower leaves are produced earlier in the season and are older than upper leaves, there are two possibilities: (1) leaves produced earlier in the season have more metals than those produced later and/or (2) individual leaves continue to accumulate metals during their lifespan. They tested these two hypotheses by numbering all leaves as they emerged from both species of plants growing in Cu–, Pb–, or Zn– contaminated vermiculite in a greenhouse. They removed tissue from the new odd–numbered leaves for metal analysis, and then analyzed the remaining half–leaves later in the season when they had senesced. They compared initial and final metal levels in individual leaves and compared the final metal levels in leaves that emerged early with those that emerged later. They found that hypothesis two was supported, since initial levels in individual leaves tended to be lower than the final levels. Hypothesis one was not generally supported for either species. They also found very great variation in both initial and final metal levels in leaves on the same plant and in different plants of the same species exposed to the same metal concentration. This suggests that scientists who are sampling leaves of these species in the field should collect many leaves and pool them to get accurate data, since individual leaves are not representative of the plant as a whole (Weis *et al*., 2003).

1.10 Conclusions

The ability of aquatic weeds to absorb and accumulated various metals that they take from the aquatic environment has been demonstrated by a number of research workers (Dean *et al*., 1972; Dietz, 1973; Erikson & Mortimer, 1975; Tremp & Kohler, 1995). It also been shown that aquatic macrophytes can be reliable indicators of metal pollution in freshwater ecosystems. Rooted macrophytes have been identified as a potentially useful group for bioremediation and biological monitoring. They take up pollutants via their roots submerged in sediments (Biernacki *et al*., 1996; Salt, 1998) and also absorb chemicals from the water column through their leaves (Biernacki *et al*., 1996). The degree of metal uptake by plants is largely dependent on the type of metal and the plant species involved.

It has been claimed numerous times that *T. capensis* and *P. australis* have the ability to accumulate large amounts of heavy metals and nutrients. It can therefore be concluded that *T. capensis* and *P. australis* are considered good candidates for heavy metal biomonitoring because they are cosmopolitan, well-studied species and there is evidence that they accumulate chemicals from both water and sediments. The concentrations in the organs of *T. capensis* confirm the ability of the roots to accumulate and be treated as the most important storage organ.

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CHAPTER 2

Monitoring for Heavy Metals with *Typha capensis* and *Phragmites australis* in the Bottelary River, Cape Town, South Africa

2.1 Introduction

Heavy metals are defined by Passow *et al*. (1961) as those metals having a density greater than five and including about thirty–eight elements (Antonovics *et al*., 1971). The metals of most immediate concern are: cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel and zinc. Heavy metals are added to fresh water streams and ponds, deliberately as components of herbicides (Brown *et al*., 1979) or as byproducts of different human activities (Brown *et al*., 1979; Vesk *et al*., 1997). If they reach high enough levels, they may become toxic to aquatic organisms. Heavy metal concentrations in the environment are of great concern due to their serious effects through the food chain on animal and human health.

The distribution and performance of many aquatic macrophytes is often correlated with water quality (Agami *et al*., 1976; Romero & Onaindia, 1995). Detecting environmental pollution by using biological materials as indicators is a cost effective, reliable and simple alternative to the conventional sampling methods (Zurayk *et al*., 2001). A number of organisms such as mosses and periphyton, fish and vascular plants have been successfully utilized (Porvari, 1995).

Aquatic macrophytes may accumulate considerable amounts of heavy metals in their tissues (Kovács *et al*., 1984). Therefore, aquatic macrophytes were thus proposed as pollution–monitoring organisms (Greger & Kautsky, 1993; Shine *et al*., 1998; Zhulidov, 1996). Uptake and accumulation of elements by plants may follow two different paths, i.e. via the root system and through the foliar surface (Sawidis *et al*., 2001). Metal bioaccumulation depends upon numerous biotic and abiotic factors, such as temperature, pH and dissolved ions in water (Demirezen, 2002; Lewander *et al*., 1996).

According to Robach *et al*. (1996) the occurrence of aquatic macrophytes is unambiguously related to water chemistry and using these plant species or communities as indicators or biomonitors has been an objective for surveying water quality. Aquatic plants are used in water quality studies to monitor heavy metals and other pollutants of water and submerged soils. Their selective absorption of certain ions combined with their sedentary nature is a reason for using hydrophytes as biological monitors (Sawidis *et al*., 1995). Aquatic macrophytes and sediments can well present further information about the metal content of their aquatic environment (Demirezen, 2002).

The suggestions from the literature above show that the plants, particularly *Typha* and *Phragmites* are good candidate species as heavy metal biomonitors in wetlands. In this study, a preliminary investigation of their use to monitor the pollution situation with respect to heavy metals in the Bottelary River and the De Novo tributary is presented. Two aquatic macrophytes *Typha capensis, Phragmites australis* and corresponding water and sediment samples were collected from the Bottelary River and the De Novo tributary and analyzed for their heavy metal contents (Cd, Cr, Cu, Fe, Mn, Ni, Pb, V and Zn) in order to ascertain their indicator value and degree of contamination.

2.2 Study Area and Sampling Sites

2.2.1 Description of Study Area

The Bottelary River (Figure 2.1) is a tributary of the Kuils River and originates in the Bottelary Hills between Kraaifontein and Stellenbosch, Western Cape, South Africa. It flows along a defined channel for some fourteen kilometers until it joins with the Kuils River in the suburb of Kuils River. The latter river in turn flows through a series of wetland and vleis prior to joining the Eerste River and then flowing into False Bay (Van Driel, 2003). The river is a wetland transitional zone of reedbeds or other wetland plant communities. It is dominated by the emergent macrophytes such as *T. capensis a*nd *P. australis* (Haskins *et al*., 2000).

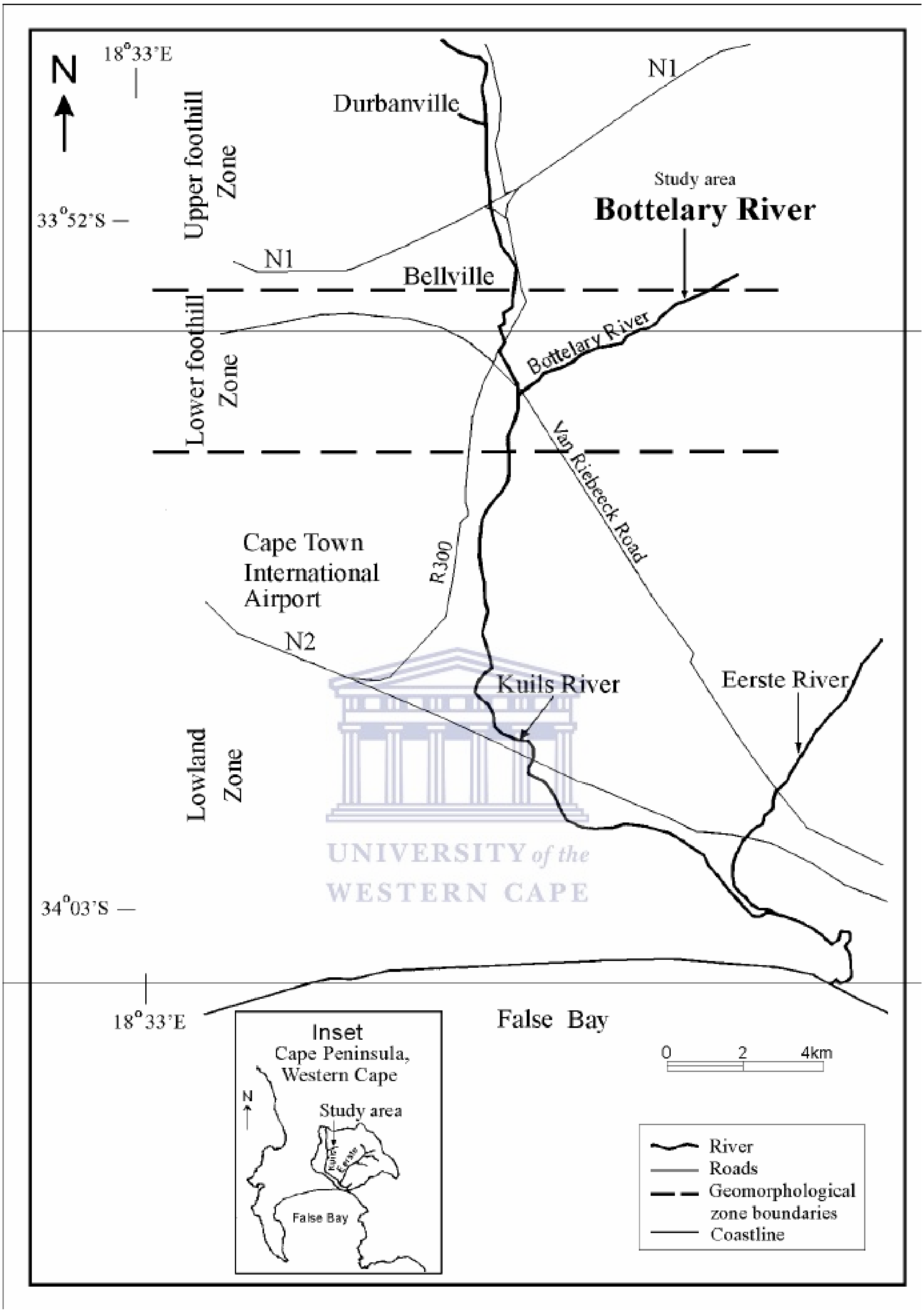

Figure 2.1 Location of the Bottelary River (Adapted from Ninham & Chittenden, 1999)

The Bottelary catchment covers a surface area of approximately eighty square kilometers. It consists of agricultural land, predominantly vineyards in the upper and middle reaches. The river flows through low–density residential suburbs in its lower reaches (Van Driel, 2003). This catchment is within the winter rainfall area of the Western Cape (Ninham, 1979). During my study period, the total monthly rainfall at the airport nearby ranged from 2.0 to 95.3 mm (Figure 2.2). The highest monthly rainfall occurred in April. The rainfall pattern displayed significant temporal variations (Figure 2.2). The rainfall data is shown in (Appendix 4.1).

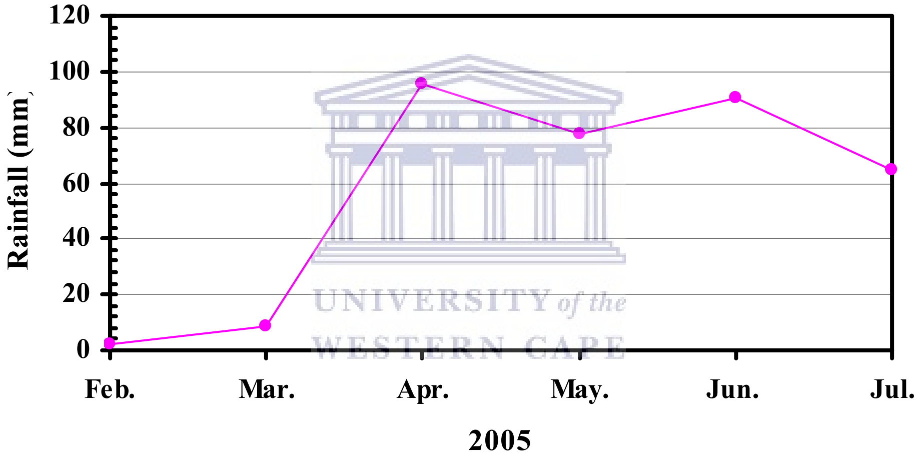

Figure 2.2 Total monthly rainfall for Cape Town International Airport during the study period (Data from the South African Weather Service)

The water flow is strongly seasonal in the Bottelary River. High natural water flows are generally recorded in winter due to the abundant precipitation. During heavy rains, urban storm runoff may exceed the capacity of the sewer systems, causing them to overflow and deliver pollutants to nearby surface waters. River bank erosion occurs due to moderately high flow velocities at peak times of discharge. In summer, supplementary irrigation is required as there is not enough rainfall. The river water is used for livestock watering, crop irrigation purposes, the Kuils River Golf course, sports grounds and watering of domestic gardens, which results in a decrease in the water flow, thereby stressing the aquatic ecosystem (Ninham, 2001).

The De Novo stream is a tributary of the Bottelary River. The De Novo tributary originates from the streets in Brackenfell and Wallacedene. It flows towards the south and joins with the Bottelary River. The De Novo tributary is strongly seasonal, almost dry during the summer and in full flow during the winter. It also receives the treated sewage effluent from the Scottsdene Wastewater Treatment Works (WWTW) which extends the flow period. It flows through vegetable farms.

2.2.2 Description of Sampling Sites

According to the distribution of *T. capensis* and *P. australis* along the Bottelary River and the possible sources of pollution, the three representative sampling sites were chosen at the junction of the De Novo tributary of the Bottelary River (Table 2.1; Figure 2.3). All sampling sites were within 500m of each other.

Table 2.1 List of sampling sites on the Bottelary River

Sites/Position	Farm
Upstream	Vredehoek
De Novo Tributary	Willemsrus
Downstream	Groenland

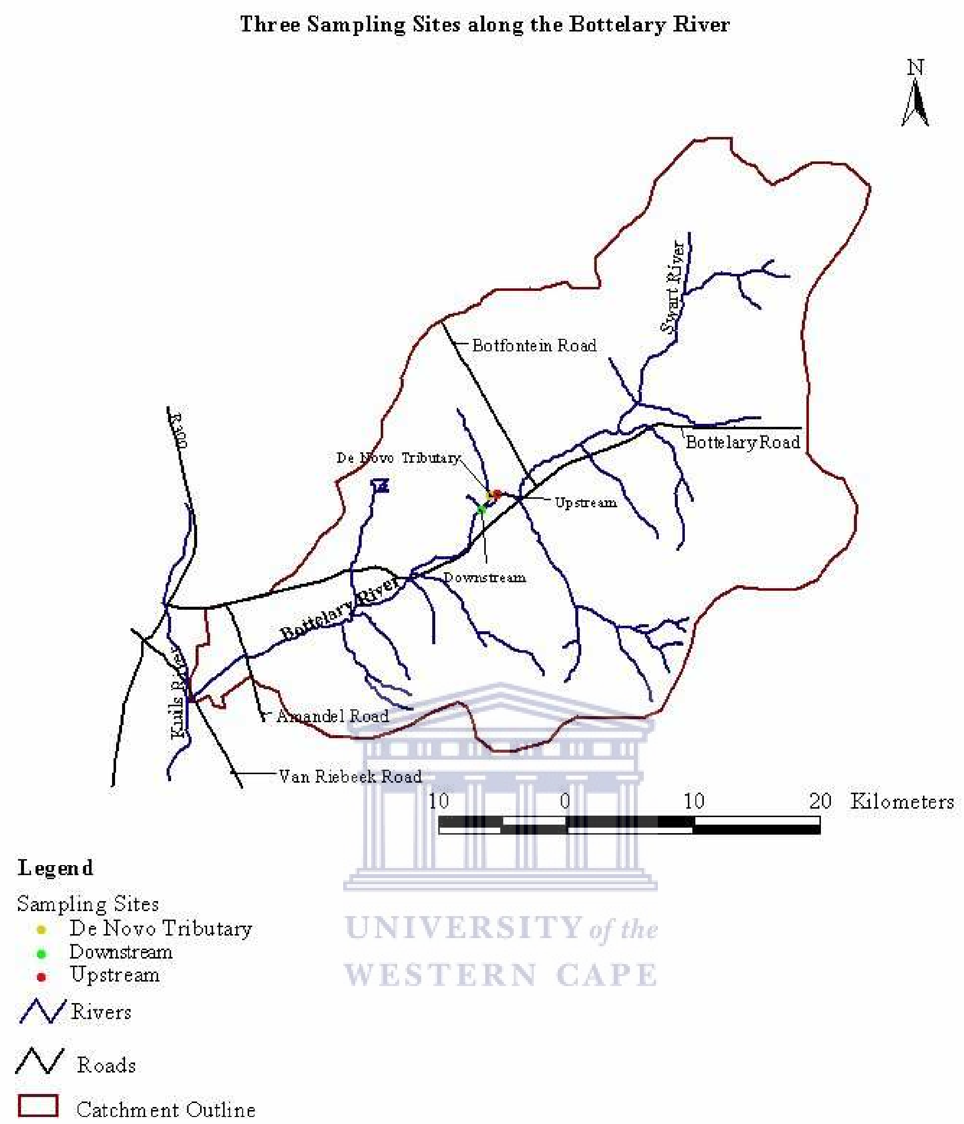

Figure 2.3 Location of the three sampling sites on the Bottelary River

2.3 Point Sources and Non–point Sources in Bottelary River

Heavy metals are emitted into the environment in different ways, i.e. transportation of heavy metals from one location to another, through release from processes in industry, through burning of fossil fuels, processes in agriculture, and other human activities (Aksoy *et al*., 2000). Water pollutants are categorized as emitted from point or non–point sources.

Point sources are distinct and confined, such as pipes from industrial or municipal sites that empty into rivers (Haslam, 1995). The major point sources discharging into the Bottelary River are the Scottsdene Wastewater Treatment Works (WWTW) and the Scottsdene canal. The WWTW has a present capacity of 7.5 mega liters per day. It is located in the upper reaches of the Bottelary River. The WWTW receives wastewater from the urban areas of Scottsdene, Wallacedene and part of Kraaifontein as well as a small light industrial area (Ninham & Chittenden, 1999). The final treated effluent from the WWTW is usually well within DWAF permit guidelines and is discharged into the De Novo tributary and one of the minor dams (Ninham, 2001). Additionally, there are other point discharges such as the storm water outlets. Collectively, all these point sources would influence water quality in the Bottelary River (Haskins *et al*., 2000).

Non–point sources are diffused and influenced by factors such as land use, climate, hydrology, topography, native vegetation and geology (Botkin & Keller, 2000). Urban runoff and agricultural runoff are mainly non–point sources, which contribute to the pollution in the Bottelary River. Cultivation is the main land–use activity in the catchment area beyond the urban boundary. The major sources of pollution from agriculture are fertilizers containing superabundant nutrients such as nitrogen and phosphorus, and heavy metals such as cadmium, copper, lead and zinc (Alloway & Ayres, 1997; Withers & Lord, 2002). Some urban surface runoff enters the Bottelary River system via overland flow. In the farmland area, pollution e.g. high nutrient runoff occurs from the various agricultural practices washes into the river (Haskins *et al.*, 2000). Furthermore, the water quality in this river may also be altered by other factors, such as livestock manure, human waste, and atmospheric deposition.

2.4 Methodology

The research methodology involved field tests as well as laboratory analyses. The analyzed parameters include water temperature, dissolved oxygen (DO), electrical conductivity (EC), pH, nitrate (NO_3^-), nitrite (NO_2^-), ammonium (NH_4^+), total nitrogen (TN), phosphate (PO_4^{3-}), cadmium (Cd), chromium (Cr), copper (Cu), iron (Fe), lead (Pb), manganese (Mn), nickel (Ni), vanadium (V), and zinc (Zn).

2.4.1 Field Procedures

The field monitoring was carried out in the Bottelary River. Three different sampling localities were selected, distributed as follows: in the De Novo tributary, and both upstream and downstream of it. Every month, from February 2005 until July 2005, samples of water (500ml), sediments (500g) and entire plants were collected at each site. Sediment samples were collected from root zone of the plants. *T. capensis* and *P. australis* plants rooted below the water level were collected. Dissolved oxygen and water temperature were measured at each site using a handheld dissolved oxygen meter (YSI Model 55 Handheld Dissolved Oxygen System, Japan).

2.4.2 Laboratory Procedures

Water samples were first filtered then measured for hydrogen ion activity using a pH meter (PHM 64 Research pH meter, Radiometer Copenhagen) and for electrical conductivity using a conductivity meter (Metrohm 644 conductometer, Switzerland). Aquamerck Reagent Kits were used to estimate levels of nitrate, nitrite and ammonium ions in the laboratory. Nitrate was measured using a Merck Nitrate Kit (Aquamerck ^{R} 8032 Nitrate MERCK), which uses the reduction of nitrate to nitrite through the formation of an azo dye with sulphanilic/gentisic acid. The reagent kit included a colour scale. Comparison with the colour scale permits a rapid and semiquantitative assessment of the nitrate content present. Nitrite was estimated by using a Merck Nitrite Kit (Aquamerck^{R} 8025 Nitrite MERCK), which uses colorimetric determination with sulfanilic acid and N– (1–naphty1) ethylenediammonium dichloride reagent, with a colour scale. In acidic solution, nitrite ions react with sulfanilic acid to form a coloured diazonium salt which then reacts with N– (1–naphty1) ethylenediammonium dichloride to form a red–violet azo dye. The colour formation was used to estimate the nitrite concentration by visual comparison of the colours of the solution with the colour standards. Ammonium was determined using a Merck Ammonium Kit (Aquamerck^{R} 11 117 Ammonium Test MERCK), which uses colorimetric determination with Nessler's reagent with a colour scale. Ammonium ions react with Nessler's reagent to form a characteristic yellow to brown dye. Again, a colour scale enabled estimations of ammonium levels. Then a 0.5 ml aliquot of hydrochloric acid (HCl) was added per 100 ml of water before storage in a cold room.

The sediments samples were air–dried for at least two days until totally dry then passed through a two mm sieve. The harvested plants were washed and separated into roots, rhizomes and leaves for *T. capensis*, or roots, rhizomes, stems and leaves for *P. australis*. The plant organs were oven–dried at 70°C to constant weight and ground using a Hammer mill. Drying of the collected materials protects the plant material from microbial decomposition and also ensures a constant reference value (Markert, 1993; Aksoy *et al*., 1999).

2.4.3 Chemical Analyses of Heavy Metals

2.4.3.1 Digestion of Plant Samples and Sediments

The sediment samples were digested using aqua regia solution, HCl: HNO_3 (3:1). One gram of sediment was weighed and placed into a digestion tube with 12 ml digestion mixture. Samples were digested for three hours at 110°C. After evaporation to near dryness, the tubes were removed and allowed to cool. The samples were diluted with 20 ml of 2 % (v/v with H_2O) nitric acid. It was then quantitatively transferred into a 100 ml volumetric flask after filtering through Whatman no.42 filter paper and diluted to volume with distilled water.

About 0.5 g of plant material was accurately weighed into a silica crucible. The crucible was then placed in a muffle furnace and heated to 400°C, until the samples became a white ash. After cooling the ash was dissolved in 4 ml of hydrochloric acid and warm water. This was then filtered and diluted to volume in a 100 ml volumetric flask, with a final hydrochloric acid concentration of 1 %.

2.4.3.2 Plant, Sediments and Water Analysis

The solutions were analyzed for Cd, Cr, Pb and V using a Unicam Solaar M Series Atomic Absorption Spectrometer (AAS) with GF95 Graphite Furnace System. The concentrations of Cu, Fe, Mn, Ni and Zn were determined by using the same instrument with an air/acetylene flame system. Water was analysed for the same elements as plants and sediments (Allen *et al*., 1986).

2.4.4 Nitrogen Analysis

The plant materials were analyzed for total nitrogen concentration by using the Kjeldahl method. The plant samples were digested using sulphuric–peroxide mixture.

About 0.4 g of plant material was accurately weighed and placed into a suitable Kjeldahl flask with 4.4 ml of digestion mixture. The digestion started at a low temperature of 100°C for half an hour until initial reaction subsided avoiding loss of sulphuric acid fumes. Thereafter, the temperature was raised consecutively to 250°C, 300°C, and 350°C at half–an–hour intervals and finally to 380°C until a colorless solution was obtained. It was then quantitatively transferred into a 100ml volumetric flask after filtering through Whatman no.42 filter paper, the digest was then and diluted to volume with distilled water. The sample solution (10 ml) was transferred into a Buchi tube. Excess sodium hydroxide (NaOH) and distilled water was added to the sample and then processed on a Buchi nitrogen distillation unit. The distillate was collected in a 50 ml Erlenmeyer flask containing 5 ml of 2 % boric acid. The boric acid mixture was then combined with five drops of methyl red indicator solution and titrated with 0.01 M standardized hydrochloric acid (HCl) solution until a pale pink colour was obtained (Allen, 1989). The nitrogen was calculated using the following formula:

N (mg/g) (plant) = Volume HCl * Normality HCl * 140 *100 *1000/100* aliquot (ml) * sample mass (g) (Cyster, 1979; Moore & Chapman, 1986)

2.4.5 Phosphorus Analysis

The Murphy and Riley (1962) method was used to determine the total phosphorus concentration of plants, sediments and water. The Murphy and Riley solution was made with the following reagents: sulphuric acid, ammonium molybdate, ascorbic acid, potassium antimonyl tartrate and a pale yellow solution was obtained. During the Murphy and Riley procedure, a standard curve, using (2; 4; 8; 16; 20; 30µg phosphorus) was prepared before running the digested samples. One ml of digested plant sample, one ml of water sample or 0.5 ml of digested sediment sample were added respectively to fifty ml volumetric flasks with eight ml of Murphy and Riley solution and diluted to volume with distilled water. One hour was allowed for colour development of standards and samples. The absorbance was measured at wavelength of 882 nm using a Shimadzu 160–A UV visible spectrophotometer. The phosphorus concentration was calculated using the following formula:

P (mg/g) (plant and soil) = Concentration (mg) * solution volume (ml) / aliquot size (ml) * sample mass (g) (Moore & Chapman, 1986)

P (mg/l) (water) = Concentration (mg) * 1000 / aliquot (ml) (Moore & Chapman, 1986)

2.4.6 Statistical Analysis

Statistical analysis was carried out using SAS Version 8.2 computer package (SAS, 1999). Anova and t-tests were used to compare the means of samples at $p \leq 0.05$. Pearson correlation coefficients were determined to examine the relationship between the levels of elements in plants, sediments and water, again at $p \leq 0.05$.

2.5 Results and Discussion

2.5.1 Plant Species Comparisons

In order to determine which of the plant species were the best accumulators of heavy metals, the analytical results of the different organs of the two species were pooled and the results for the plant species presented in Table 2.2.

Table 2.2 Significant differences ($p \leq 0.05$) in element concentration (mg/g) between *Typha capensis* and *Phragmites australis* growing in the Bottelary River study area

Species	Fe	Mn	Pb	Zn
T. capensis	1.8756	0.1364	0.000007	0.12899
P. australis	1.1887	0.0847	0.000406	0.0815

The data shows that there were significant differences observed in the concentrations of iron, manganese, lead and zinc between these two species. *T. capensis* had much higher concentration of iron, manganese, zinc and far lower lead concentrations than *P. australis*. *T. capensis* readily accumulated iron but not lead.

2.5.2 Localization (accumulation) at Organ Level:

Table 2.3 shows the relative concentrations of elements in the different plant organs. In this table, data for roots, rhizomes and leaves for both species were pooled, while stems refer to *P. australis* only.

Table 2.3 Average concentration of elements in organs of pooled *Typha capensis* and *Phragmites australis* from the Bottelary River system. Concentrations followed by the same letter do not differ significantly ($p \leq 0.05$)

Organs	Cd µg/g	Cr mg/g	Cu mg/g	Fe mg/g	Mn mg/g	Ni mg/g	Pb mg/g	V mg/g	Zn mg/g	TN mg/g	P mg/g
Leaves	0.029b	0.0066b	0.0042b	0.3378b	0.1164b	0.0038b	0.0001b	0.0030b	0.0619b	2.5618a	2.4327b
Stems	0.025b	0.0067b	0.0037b	0.1989b	0.0298c	0.0042b	0.0005a	0.0002b	0.0583b	1.3062c	1.723b
Rhizomes	0.04b	0.0146a	0.0078a	0.6851b	0.036c	0.0065b	0.0001b	0.0008b	0.0733b	1.7474b	2.9249ab
Roots	0.199a	0.0212a	0.0087a	4.0683a	0.2069a	0.0093a	0.0003ab	0.0066a	0.1920a	2.0666b	4.5019a

Macrophytes take up heavy metals mainly through the roots although uptake through the leaves may also be of significance (Baker & Walker, 1990). The storage capacity of metals can vary between organs and species. The general evidence from Table 2.3 is that most elements (Cd, Cr, Cu, Fe, Mn, Ni, V and Zn) were accumulated mainly in the roots of the plants growing along the Bottelary River, with a decrease in concentration in the rhizomes and above ground parts. However, more lead was accumulated in the stems than in the roots. Hwang *et al*., (1996) found nitrogen was the major nutrient in plant tissues, reaching maximum concentrations in the above-ground tissues. The same results were also noted for these two species along the Bottelaty River (Table 2.3).

2.5.3 Position Comparisons

In order to evaluate the sources of various contaminants: from the town (De novo tributary) or from the farms (upstream), the analytical results of pollutant concentrations in plant species growing at the three sampling sites are presented in table 2.4.

Table 2.4 Differences in element concentrations for pooled *Typha capensis* and *Phragmites australis* plants in the Bottelary River study area comparing the three sampling sites. Concentrations followed by the same letter do not differ significantly ($p \leq 0.05$)

Position	Cd µg/g	Cr mg/g	Cu mg/g	Mn mg/g	Ni mg/g	V mg/g	TN mg/g
Upstream	0.063b	0.0101b	0.0085a	0.1748a	0.0043b	0.0030a	1.831b
Tributary	0.067b	0.0186a	0.0050b	0.0808b	0.0079a	0.0019ab	1.925b
Downstream	0.110a	0.0105b	0.0059b	0.0651b	0.0064ab	0.0018b	2.24a

The results showed there were significant differences in the plants' element concentrations of cadmium, chromium, copper, manganese, nickel, vanadium and total nitrogen between these three sampling sites. Higher concentrations of copper, manganese and vanadium were observed in the upstream samples. The main urban pollution entering the De Novo tributary as a point source caused the high values of chromium and nickel in the Bottelary River (Table 2.4). Agricultural runoff also significantly contributed to high concentrations of heavy metals, nitrogen and phosphorus in the Bottelary River. The probable sources of the chemical pollutants in the Bottelary River include both point sources from the Wastewater Treatment Works (WWTW) and the Scottsdene canal, and diffuse sources from urban run off and agricultural runoff in the catchment area.

2.5.4 Seasonal Concentration Trends in Water, Sediments and Plants

2.5.4.1 Cadmium Concentrations

Cadmium is a toxic element, which occurs naturally in plants, sediments and water, and exists along with zinc in nature. The sources of cadmium in the aquatic environment are from natural weathering processes, mining, metal smelters, industries, agricultural use of sludges, fertilizers and pesticides, burning of fossil fuels, and the deterioration of galvanized materials and cadmium-plated containers (Anon b, 1996). Air, water and soil cadmium concentrations in areas unaffected by human activities are usually very low, in contrast to concentrations in polluted areas (Raspor, 1980). Cadmium is both metabolically and passively taken up and it is easily transported in the plant. Plants in unpolluted environment contain 0.01–0.3 µg/g cadmium (Allen, 1989).

The results are presented as averages of the three sites, which were sampled during the same period. There were significant differences in the water and sediments over time (Figures 2.4 & 2.5). A sudden increase in cadmium concentration was observed in the water and sediments between days 56 and 84. This may be due to the flushing of pollutants by the April rains (Figure 2.2).

There was no significant variation in the cadmium concentrations in the plants overall during the study period. The average cadmium concentrations in plants were in the range of 0.059–0.097 µg/g which is in the range of unpolluted regions (Allen, 1989). Comparing the amount of cadmium in water, sediments and plants, the plants had much higher levels of cadmium than the sediments and water. *T. capensis* had higher cadmium concentrations than *P. australis* at the end of the study period (Figure 2.6).

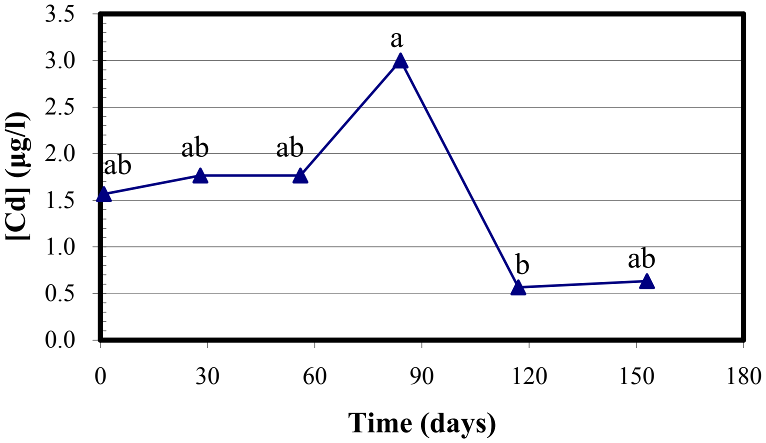

Figure 2.4 The variation in cadmium concentrations in water from the Bottelary River study area over the sampling period in 2005. Concentrations marked by the same letter do not differ significantly ($p \leq 0.05$)

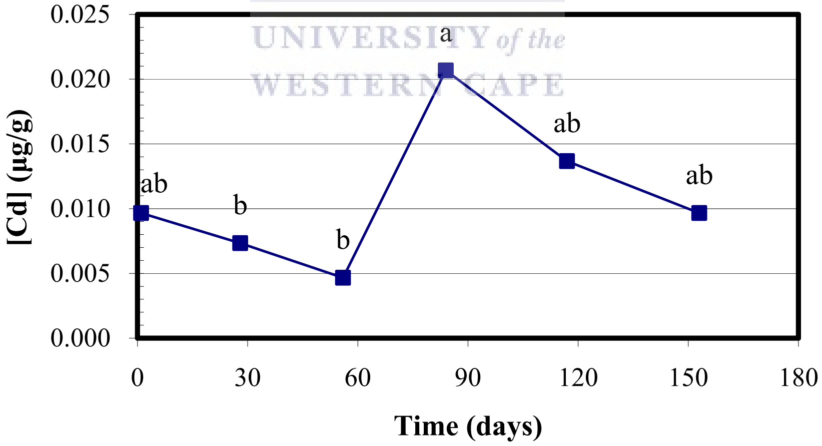

Figure 2.5 The variation in cadmium concentrations in sediments from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

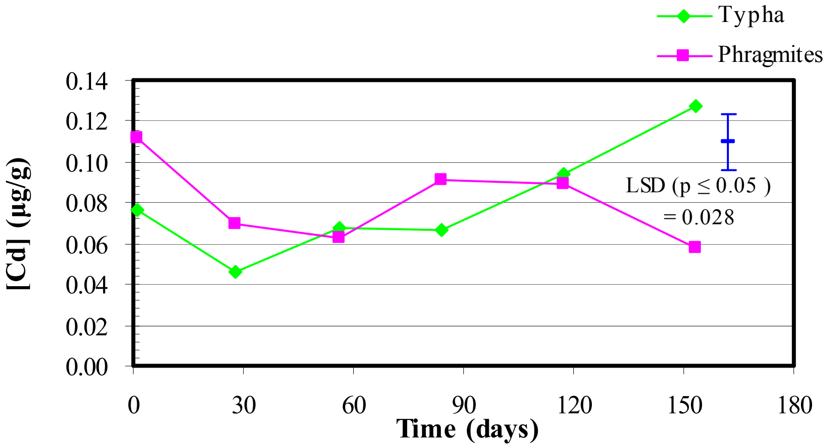

Figure 2.6 The variation in cadmium concentrations in *Typha capensis* and *Phragmites australis* from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.2 Chromium Concentrations

Chromium is a relatively scarce metal. Its occurrence and amounts in aquatic ecosystems are usually very low. Most elevated levels of chromium in aquatic ecosystems are a consequence of industrial activity (Anon b, 1996).

During the six of months in which plants was sampled, there were significant variations in the chromium concentrations in water and plants (Figures 2.7 & 2.8). The chromium concentrations in water showed an increase towards the end of the sampling period. There was no significant difference in the concentrations of chromium in the sediments over time. The mean concentrations of chromium in sediments were in the range of 0.002–0.005 mg/g. The levels of chromium in plants were higher than those in the sediments, and were much greater than those in the water. Chromium accumulated to higher concentrations in *P. australis* than in *T. capensis* during the study period (Figure 2.9).

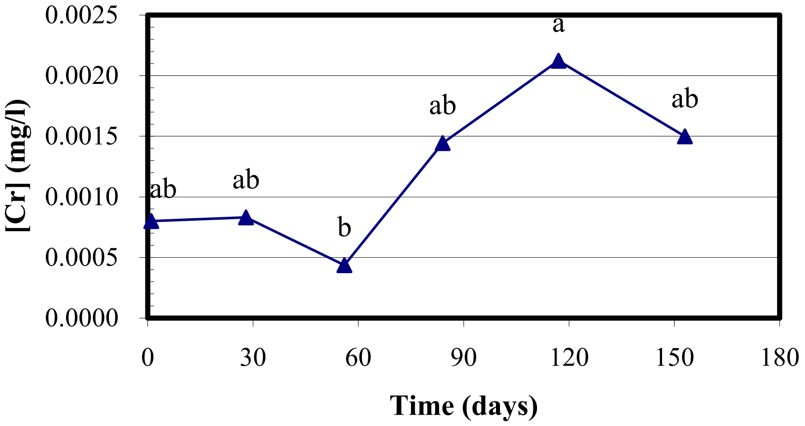

Figure 2.7 The variation in chromium concentrations in water from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

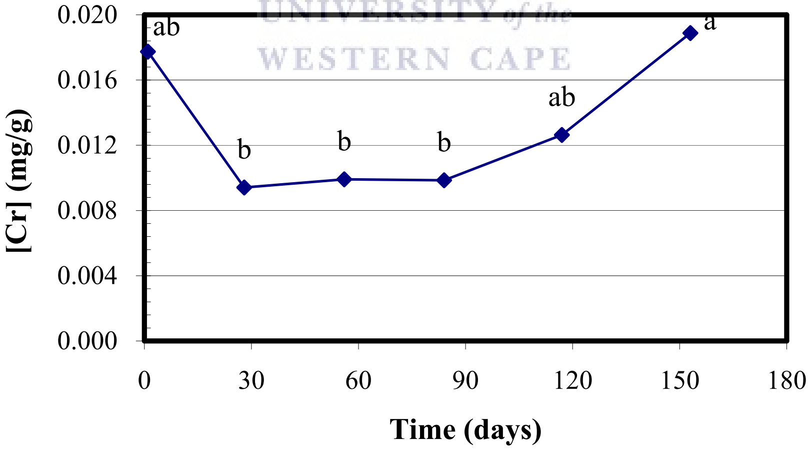

Figure 2.8 The variation in chromium concentrations in pooled plants from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

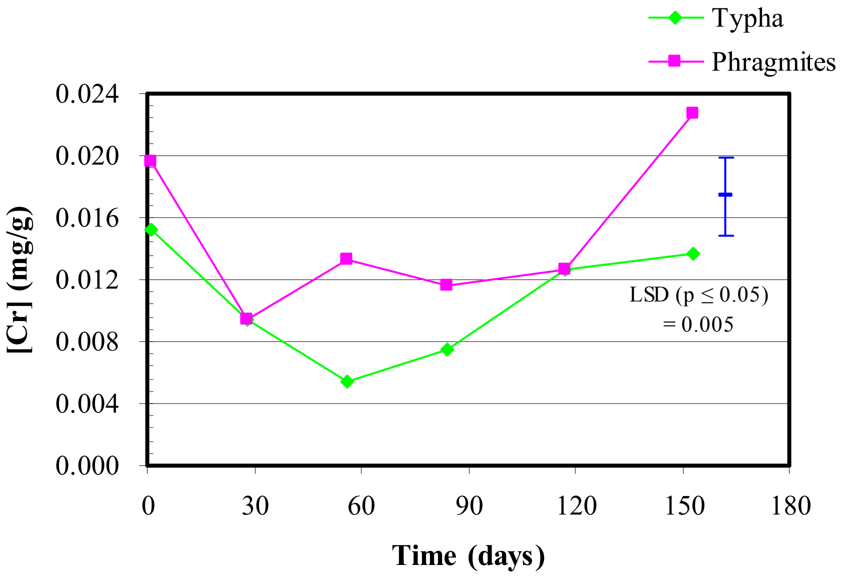

Figure 2.9 The variation in chromium concentrations in *Typha capensis* and *Phragmits australis* from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.3 Copper Concentrations

Copper is one of the world's most widely used metals (Anon b, 1996), and is regarded as a potential hazard (Anon e, 2003). It is also an essential micronutrient required by all organisms (Avenant–Oldewage & Marx, 2000) and is rapidly accumulated by plants and animals (Anon b, 1996). Copper is generally more soluble in acidic waters at pH values below 6.5. It is easily adsorbed and precipitated in sediments at alkaline pH. The copper concentrations in plants above 0.005–0.02 mg/g are regarded as poisonous (Jones & Belling, 1967).

Significant differences in copper concentrations in water were observed over the course of the study period (Figure 2.10). The peak in the copper concentration in the water seems to parallel the flushing with the first major rains (Figure 2.2). There was no significant variation in the copper concentrations in the sediments and plants over time. The mean concentrations of copper in sediments were in the range of 0.001–0.008 mg/g. The mean concentrations of copper in plant samples ranged from 0.006 to 0.008 mg/g. The concentrations of copper were high in the plants and sediments and lower in the water. The copper concentrations tended to be higher in *T. capensis* than in *P. australis* (Figure 2.11).

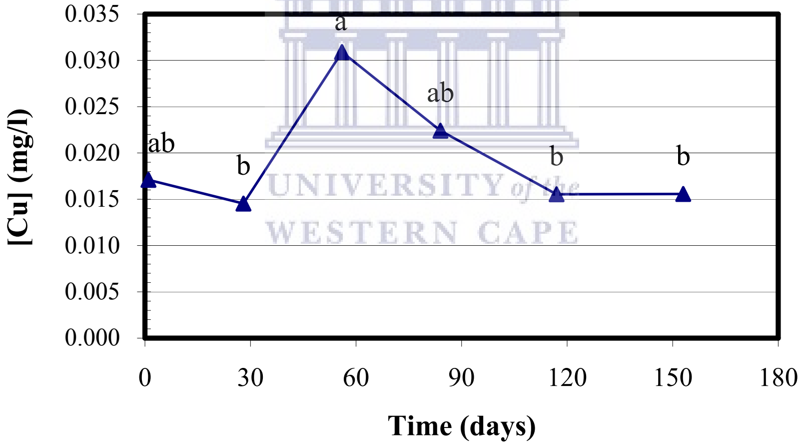

Figure 2.10 The variation in copper concentrations in water from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

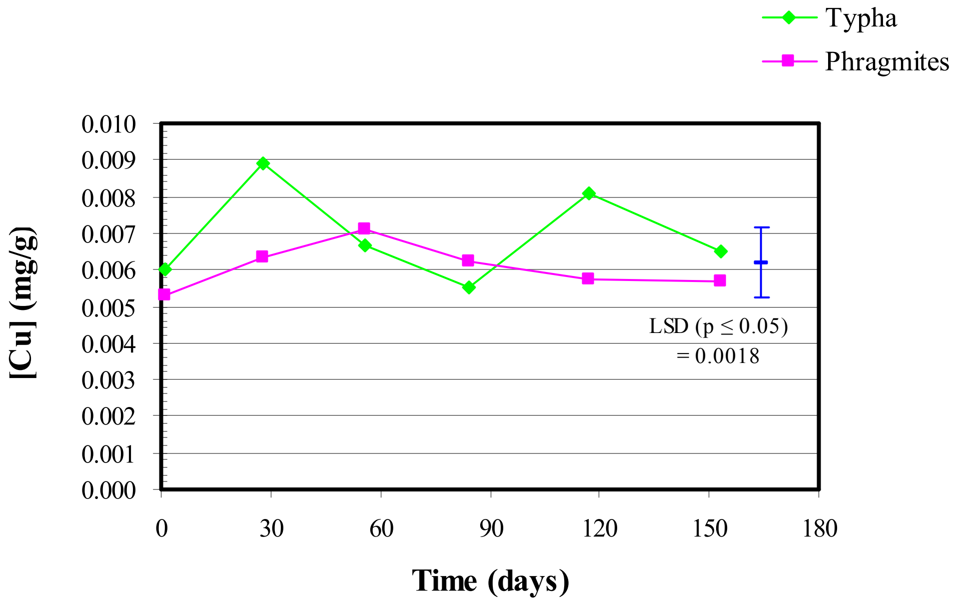

Figure 2.11 The variation in copper concentrations in *Typha capensis* and *Phragmites australis* from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.4 Iron Concentrations

Iron is the fourth most abundant element in the earth's crust and may be present in natural waters. It is an essential micronutrient for all organisms. Iron is released into the environment by natural process, such as weathering of sulphide ores and igneous, sedimentary and metamorphic rocks and by human activities, mainly from the burning of coke and coal, acid mine drainage, mineral processing, sewage, landfill leachates and the corrosion of iron and steel. In the aquatic environment, the chemical behavior of iron is determined by oxidationreduction reactions, pH and the presence of coexisting inorganic and organic complexing agents (Anon b, 1996).

During the period of investigation, there was no significant difference in the iron concentration in the water and sediments. The mean concentrations of iron in water ranged from 0.22 to 0.63 mg/l. Most iron in oxygenated waters occurs as ferric hydroxide in particulate and colloidal form and as complexes with organic, especially humic, compounds. Ferric salts are insoluble in oxygenated waters, and hence iron concentrations are usually low in the water column (Anon b, 1996). The mean concentrations of iron in sediments were in the range of 2.13–11.77 mg/g. There were significant variations in the iron concentration found in the plants, which ranged from 1.05 to 2.08 mg/g (Figure 2.12). The highest value of iron was noted in April, which coincided with the occurrence of the highest rainfall (Figure 2.2). The iron concentration levels in sediments were higher than those in the plants, and were much greater than those in the water. Iron accumulated to higher concentrations in *T. capensis* than in *P. australis* during the study period. The peak in iron concentrations appears to parallel the flush with the first rains. *P. australis* responded more rapidly than *T. capensis* (Figure 2.13).

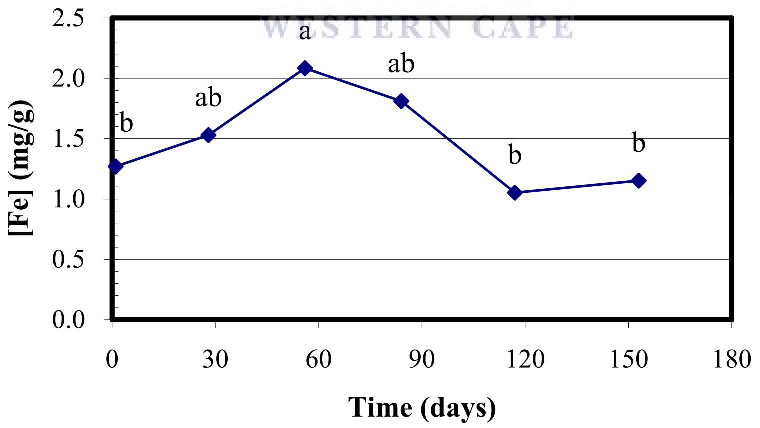

Figure 2.12 The variation in iron concentrations in pooled plants from the Bottelary River study area over the sampling period (2005). Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

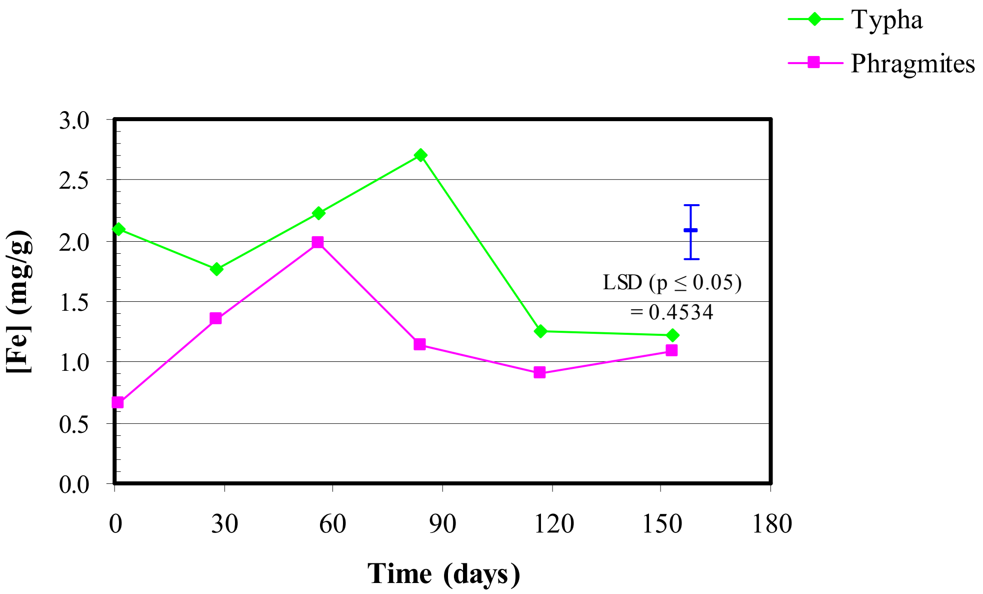

Figure 2.13 The variation in iron concentrations in *Typhy capensis* and *Phragmites australis* from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.5 Lead Concentrations

Lead occurs as metallic lead and inorganic and organometallic compounds (Anon b, 1996). It is considered to be potentially hazardous and toxic to most forms of life, and relatively accessible to aquatic organisms (Anon e, 2003). The major sources of lead into aquatic environments are street runoff, industrial and municipal wastewater discharge, mining, milling and smelting of lead and combustion of fossil fuels. Divalent lead is accumulated by aquatic biota, while decreasing pH increases the bioavailability of divalent lead. At a constant pH, solubility decreases with increasing alkalinity (Anon b, 1996). Plants normally absorb only a little lead, which does not move beyond the roots (Adler, 1996). According to Roos (1994), 0.03–0.3 mg/g lead concentrations in plants are considered toxic.

The lead concentrations in water were below our detection level.

The results showed no significant difference in the lead concentration in the sediments and plants during the period of study. The mean lead concentrations in sediments were in the range of 0.0012-0.0032 mg/g. The mean concentrations of lead in plant samples ranged from 0.0001 to 0.0003 mg/g. High lead concentrations were found in *P. australis* and sediments. *T. capensis* accumulated very little lead (Figure 2.14).

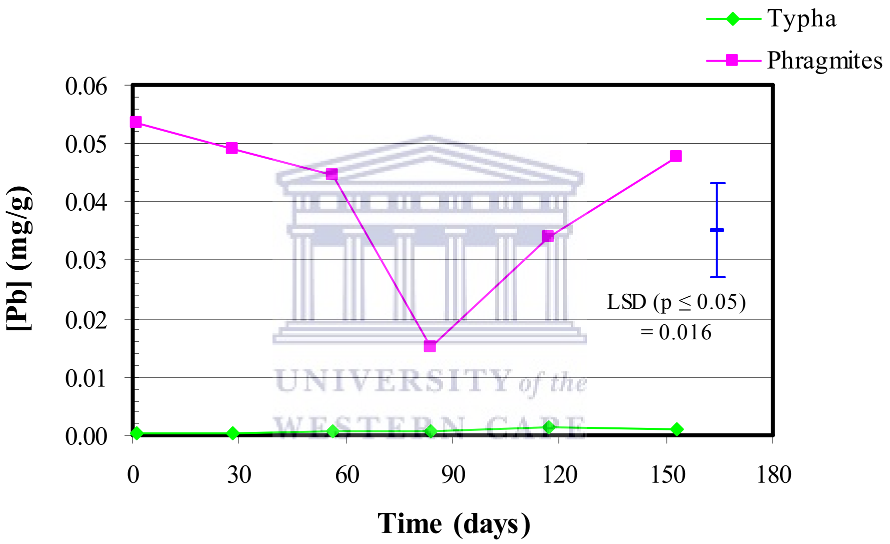

Figure 2.14 The variation in lead concentrations in *Typha capensis* and *Phragmits australis* from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.6 Manganese Concentrations

Manganese is an essential micronutrient for plants and animals. It is a functional component of nitrate assimilation and an essential catalyst of numerous enzyme systems in animals, plants and bacteria. The concentration of dissolved manganese is influenced by changes in redox potential, dissolved oxygen, pH and organic matter (Anon b, 1996).

The results showed no significant difference in the concentrations of manganese in the water and sediments during the study period. The mean manganese concentrations in water ranged from 0.006 to 0.021 mg/l. The mean concentrations of manganese in sediments were in the range of 0.012–0.331 mg/g. There were significant variations in the manganese concentrations in the plants over time. Manganese appears to decrease in the plants towards the wet season (days 56–153) (Figure 2.15). High concentrations of manganese were found in the sediments and plants and much less in the water. Manganese concentrations were higher in *T. capensis* than in *P. australis*, especially early in the sampling period (Figure 2.16).

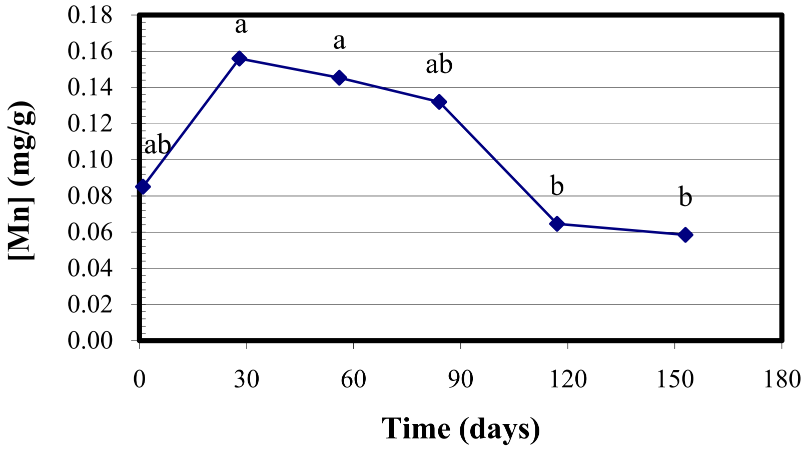

Figure 2.15 The variation in manganese concentrations in pooled plants from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

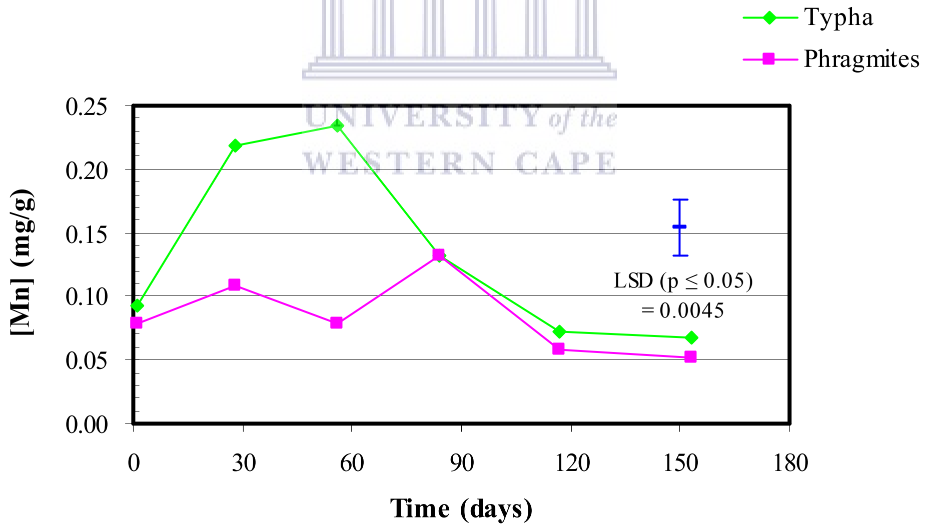

Figure 2.16 The variation in manganese concentrations in *Typha capensis* and *Phragmits australis* from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.7 Nickel Concentrations

Nickel is a natural element of the earth's crust. It occurs naturally in the environment at low levels in water and soil (Anon d, 2000).

During the period of investigation, significant increases in the nickel concentrations in both plants and water were observed (Figures 2.18 & 2.17). The comparatively high value of nickel in water was noted from April to July, which coincided with the occurrence of the comparatively high rainfall (Figure 2.2). This may be due to a number of reasons, such as runoff from the farms entering the river and increasing the nickel concentration in the river and/or nickel from the river sediment may have dissolved into the river water due to a decrease in the water pH over the rainy season (Figures 2.17 & 2.27). There was no significant variation in the nickel concentrations in the sediments over time. The mean concentrations of nickel in sediments were in the range of 0.0004–0.0028 mg/g. The concentrations of nickel were much higher in the plants than sediments and water. At the end, nickel concentrations were higher in *P. australis* than in *T. capensis*. This was, however not true overall (Figure 2.19).

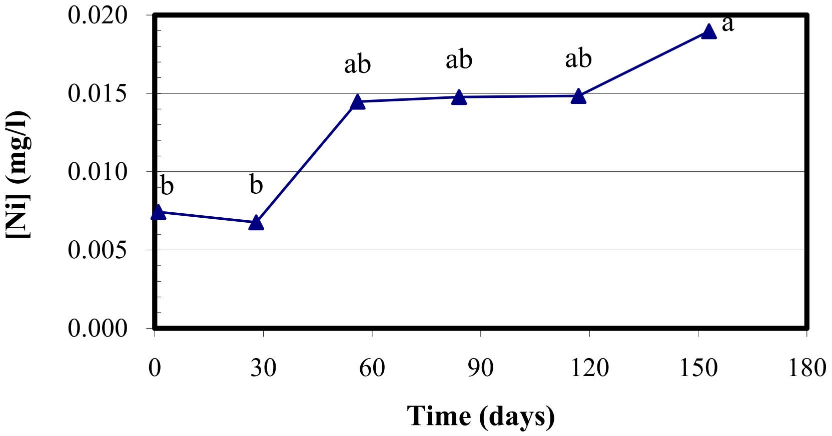

Figure 2.17 The variation in nickel concentrations in water from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

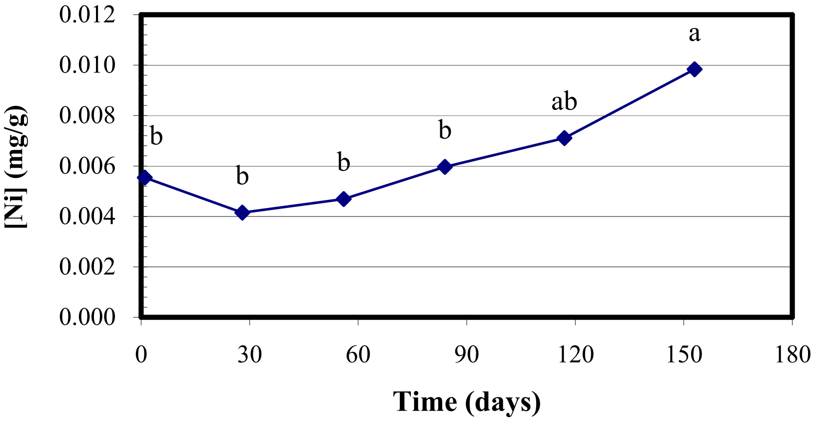

Figure 2.18 The variation in nickel concentrations in pooled plants from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

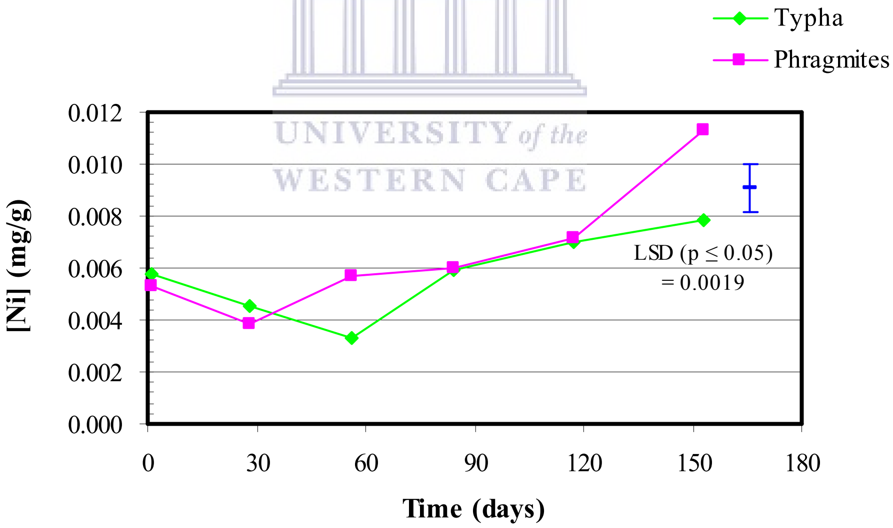

Figure 2.19 The variation in nickel concentrations in *Typha capensis* and *Phragmites australis* from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.8 Vanadium Concentrations

Vanadium is one of bioelements with biological assimilation and function. It plays an essential role in aquatic ecosystems. The behavior of vanadium relates to the variation of biological activity in the aquatic environment. The vanadium concentrations in natural waters are too low to be surveyed by the conventional analytical techniques (Masahito, 1989).

There was no significant difference in the vanadium concentrations in the water and sediments during the period of study. The mean concentrations of vanadium in water ranged from 0.004 to 0.007 mg/l. The mean concentrations of vanadium in sediments were in the range of 0.005–0.033 mg/g. There was significant variation in the vanadium concentrations in the plants over time. The mean concentrations of vanadium in plants were in the range of 0.001–0.003 mg/g. The peak in vanadium concentrations in plants appears to parallel the flush with the first rains (Figure 2.20). The vanadium concentration levels in sediments were higher than those in the plants, and were much greater than those in the water. While the concentrations in *T. capensis* and *P. australis* did not differ overall, *T. capensis* had higher concentrations initially (Figure 2.21).

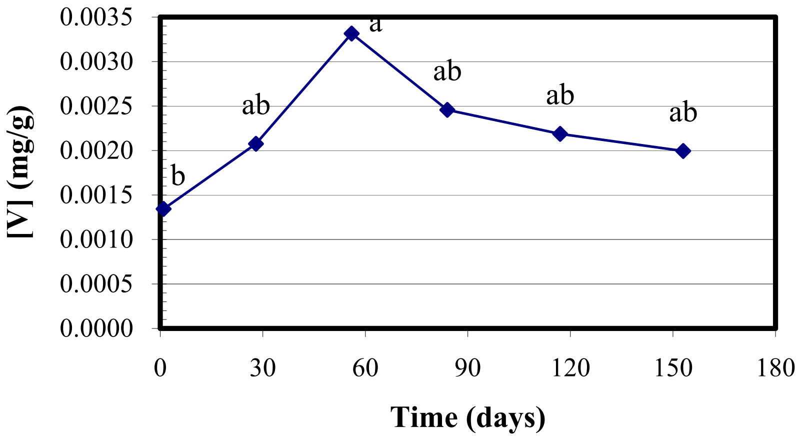

Figure 2.20 The variation in vanadium concentrations in pooled plants from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

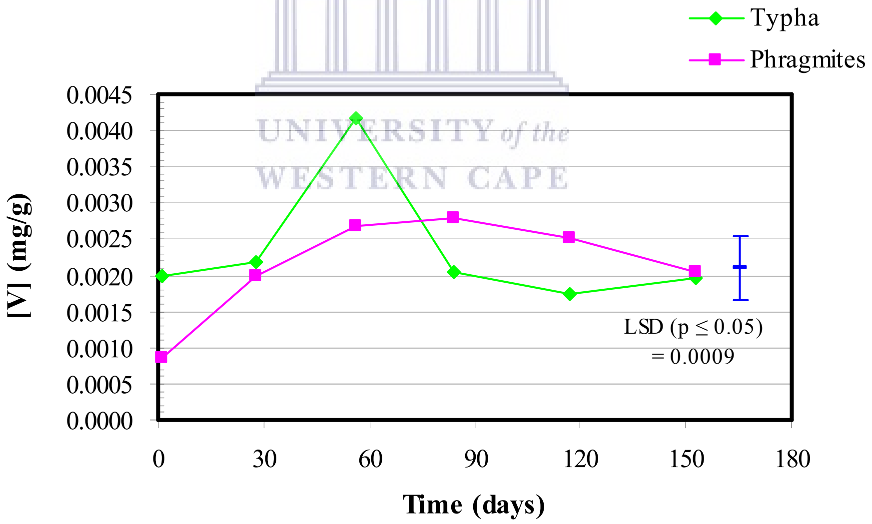

Figure 2.21 The variation in vanadium concentrations in *Typha capensis* and *Phragmites australis* from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.9 Zinc Concentrations

Zinc is an essential micronutrient for all organisms as it forms part of the active site in various metalloenzymes. In most waters, the zinc (II) as oxidation state of zinc is toxic to aquatic organisms at relatively low concentrations. The greatest dissolved zinc concentration occurs in low pH, low alkalinity and high ionic strength solutions. Adsorption of zinc by hydrous metal oxides, clay minerals and organic material is an important process in aquatic ecosystems since it affects the bioavailability and toxicity of zinc (Anon b, 1996). Zinc concentrations in plants above 0.005–0.02 mg/g are regarded as poisonous (Allen, 1989).

During the period of investigation, there was no significant difference in the zinc concentrations in the water and sediments. The mean concentrations of zinc in water ranged from 0.016 to 0.036 mg/l. The zinc concentrations in sediments were in the range of 0.009–0.045 mg/g. There were significant differences in the zinc concentrations in the plants over time. The mean concentrations of zinc in plants were in the range of 0.060–0.234 mg/g and an abrupt increase was seen at the end of the study period (Figure 2.22). The greatest value of zinc in plants was noted in July, which coincided with the lowest pH value in the water (Figure 2.27). Higher levels of zinc were found in *T. capensis* than in *P. australis* (Figure 2.23). The zinc levels were much higher in the plants than in the sediments which were in turn higher than these in the water. The abrupt increase in zinc levels in plants (Figure 2.22) was due to an increase in the concentrations in *T. capensis* (Figure 2.23).

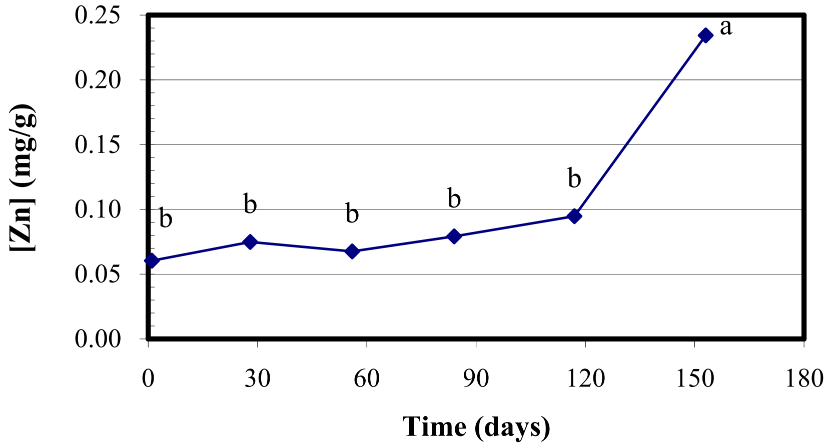

Figure 2.22 The variation in zinc concentrations in pooled plants from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

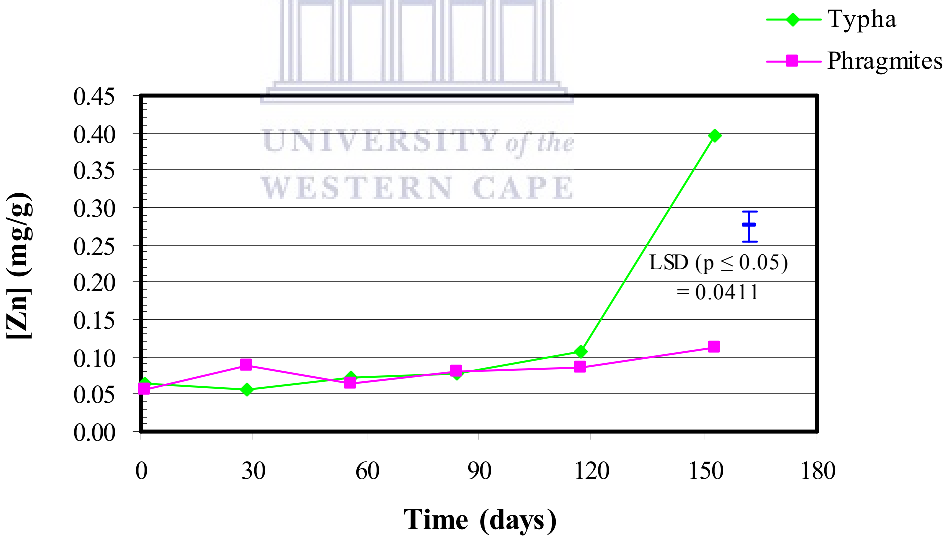

Figure 2.23 The variation in zinc concentrations in *Typha capensis* and *Phragmites australis* from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.10 Nitrogen Concentrations

The inorganic nitrogen includes all the major inorganic nitrogen components present in water, such as ammonia, ammonium, nitrite and nitrate. The concentration of inorganic nitrogen in water is affected by changes to water temperature and pH. Surface runoff from the surrounding catchment area, the discharge of effluent streams, agricultural fertilizers and organic industrial wastes are the major sources of inorganic nitrogen entering aquatic systems (Anon b, 1996). Ammonia (NH_3) and ammonium (NH_4^+) are reduced forms of inorganic nitrogen and their relative proportions are controlled by water temperature and pH. Most aquatic organisms are sensitive to the toxic effects of ammonia. The ammonium ion generally has little or no toxicity to aquatic biota, but contributes to eutrophication. Nitrite (NO_2^-) is the inorganic intermediate of the inter–conversion of nitrate and ammonia. Nitrate (NO_3^-) is the end product of the oxidation of organic nitrogen or ammonia and does not cause direct toxic effects. In the aquatic ecosystem, nitrite is converted to nitrate rapidly under oxidizing conditions. Therefore, nitrate is more stable than nitrite and is usually far more abundant in the aquatic environment (Anon b, 1996).

The sampling was conducted between February and July in 2005. Total nitrogen concentrations in plant samples from the Bottelary River were in the range of 0.48–4.79 mg/g. The mean concentrations of total nitrogen varied between 1.84–2.15 mg/g. There was no significant difference in the total nitrogen concentration of the plants over time.

There was no significant variation in the ammonium ion concentrations in water. The ammonium ion concentrations in water ranged from 0 to 3 mg/l. The average ammonium ion concentrations were in the range of 0.17–1.5 mg/l over the study period.

During the study period, there were significant variations in the nitrite concentrations in water, which ranged between 0.07 and 4.67 mg/l. The nitrite concentrations in water increased until day 84 (May), then decreased to the end (Figure 2.24), perhaps due to the fact that there was an increase in the rainfall (Figure 2.2).

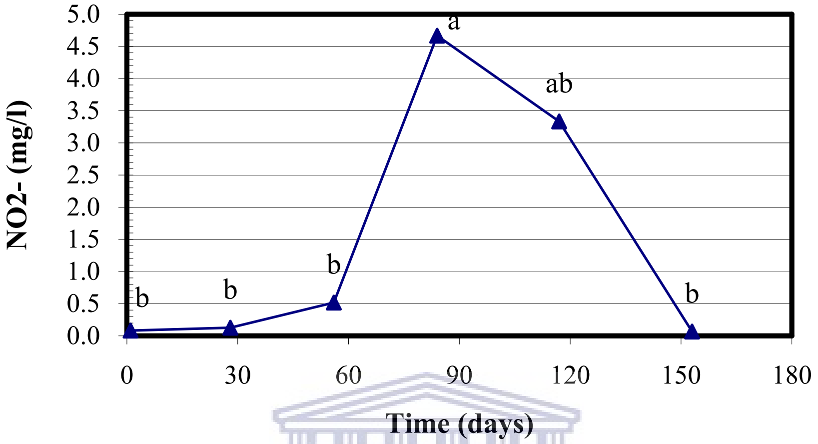

Figure 2.24 The variation in nitrite concentrations in water from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

Nitrate concentrations ranged from 0 to 200 mg/l in the water samples from the Bottelary River and the De Novo tributary. There were significant variations in the nitrate concentrations in water, which ranged from 41.67 to 150 mg/l and decreased during the wet season (Figure 2.25). This may be due to dilution from the rainwater (Figure 2.2).

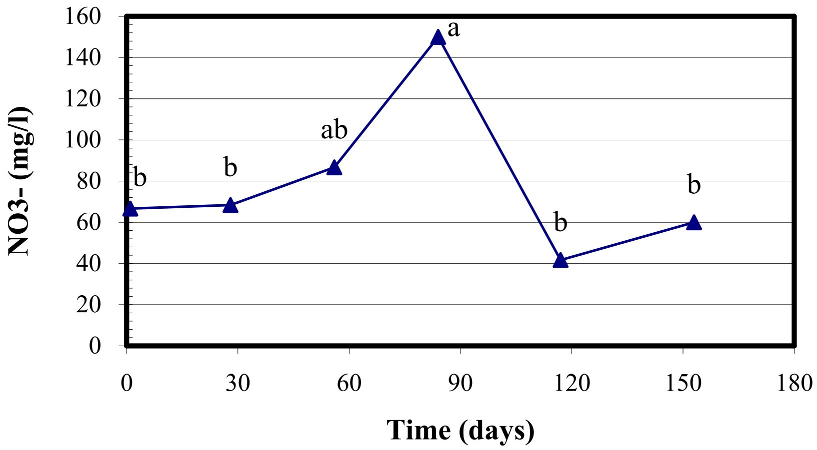

Figure 2.25 The variation in nitrate concentrations in water from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.11 Phosphorus Concentrations

Phosphorus is considered to be the principle nutrient controlling the degree of eutrophication in aquatic ecosystems. It is accumulated by a variety of living organisms as an essential macronutrient. The addition of large quantities of phosphorus accelerate algal and macrophyte growth in natural waters, enhancing eutrophication (Anon b, 1996).

The results showed no significant difference in the phosphorus concentrations in plant species, sediments and water over time. The mean concentrations of phosphorus in water were in the ranged of 14.10–18.72 mg/l. The mean concentrations of phosphorus in sediments ranged from 0.13 to 2.05 mg/g. The mean concentrations of phosphorus in plants were in the range of 2.55–4.53 mg/g. The higher levels of phosphorus were found in the plants than in the sediments and water. In general, *T. capensis* had higher phosphorus concentrations than *P. australis* (Figure 2.26).

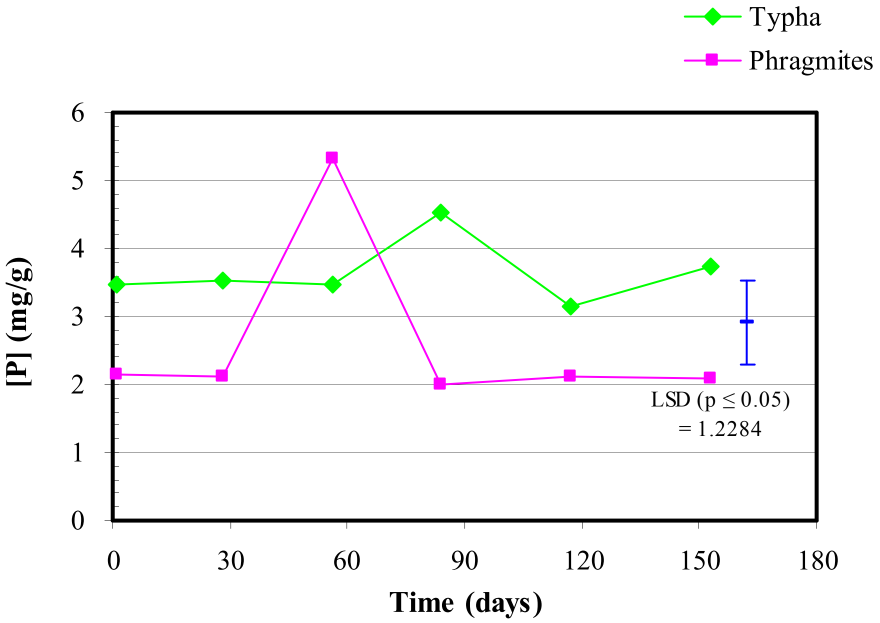

Figure 2.26 The variation in phosphorus concentrations in *Typha capensis* and *Phragmites australis* from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.12 pH Measurements

pH is used to indicate the alkalinity or acidity of a substance as ranked on a scale from 1.0 to 14.0. And it determines whether the water or soil is acidic, basic or neutral. pH affects many chemical and biological processes in the water. For example, low pH can allow toxic elements and compounds to become mobile and available for uptake by aquatic plants (Anon e, 2003). A pH reading below 6.5 generally considered as being acidic may cause problems of heavy metal toxicity (Anon a, 1993). A reading of 6.5 to 7.5 is considered neutral, suitable for general plant growth (Parkpain *et al.*, 2000), whereas above 7.5 is regarded as basic.

The range of pH in water samples from the Bottelary River generally varied between 6.90 and 8.43. There were significant differences in the pH of water over the course of the research period. The results indicated that the average pH of water was 7.5 and considered to be neutral (Figure 2.27). Measurements of pH in water deceased with time. Heavy metal availability is increased under acidic conditions (Lakatos, 1983). If the environmental conditions remain unchanged, the plant tissue lead concentration would decrease as the pH of the environment increases (Lepp, 1981). My results have shown that lower pH values may be the reasons for the increase in lead absorption (Figures 2.27 & 2.14).

Since the solubility of metal ions is pH dependent (Sparks, 1995), the pH level of waters in Bottelary River would be important in determining metal availability. Organic matter in the sewage–sludge and fertilizers are probably critical in metal immobilization for future years or decades (McBride, 1995).

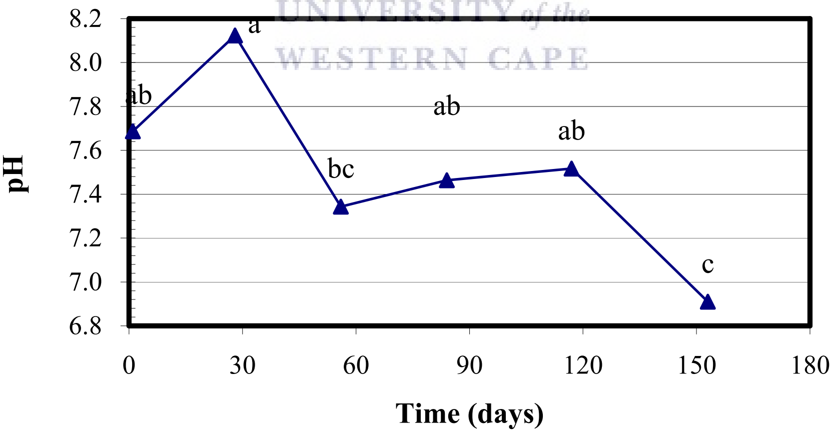

Figure 2.27 The variation in measurements of pH in water from the Bottelary River study area over the sampling period in 2005. Values marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.13 Conductivity Measurements

Conductivity is an indication of the amount of salts dissolved in water. It is also defined as the amount of ions (positive and negative) in water, and the water's ability to pass an electrical current (Anon e, 2003). Electrical conductivity is a useful indicator of the salinity or total salt content in a water sample (Anon b, 1996).

During the period of investigation, there was a significant decrease in electrical conductivity in the water samples (Figure 2.28). The values of electrical conductivity varied between 0.37 and 5.75 mS/cm. The average values of electrical conductivity were in the range of 1.14–2.48 mS/cm. Thus, the electrical conductivity was low and the salinity effects were negligible. The decrease may be due to the winter rains.

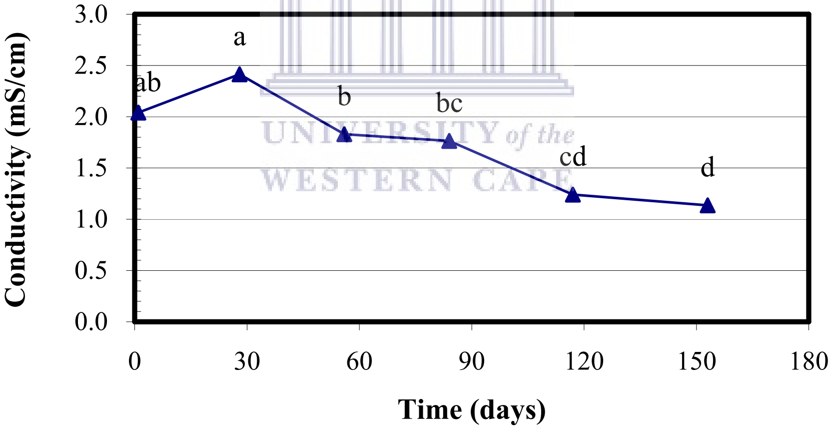

Figure 2.28 The variation in measurements of electrical conductivity in water from the Bottelary River study area over the sampling period in 2005. Values marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.14 Dissolved oxygen

Oxygen is measured as dissolved oxygen (Anon e, 2003). It is the volume of oxygen contained in water. Oxygen enters the water through photosynthesis in aquatic plants or from the transfer of oxygen between the air and water (waves, turbulence, currents, etc). Fast–moving water, lower temperature and lower salinity all result in the availability of more dissolved oxygen (Anon f, 2005).

WHO (1993) recommended a concentration of dissolved oxygen of 5 mg/l or above (Ngoye & Machiwa, 2004). Dissolved oxygen levels below 5 mg/l could cause adverse affects to aquatic life (Fatoki *et al*., 2003). From the results, about 6.7 percent (6.7%) of the dissolved oxygen concentrations in the water samples from the Bottelary River were less than 5 mg/l (Appendix 4.1).

There were significant differences in the oxygen content of water during the course of the six months. The dissolved oxygen concentrations in water samples from the Bottelary River ranged between 2.92 and 9.11 mg/l, indicating an oxygen saturation range of 32.0–90.4%. Figure 2.29 shows that the mean values of dissolved oxygen ranged from 5.47 to 8.37 mg/l, correspondingly, oxygen saturation range of 61.33–82%. The highest mean concentration of dissolved oxygen was noted in May, probably due to the low water temperature (Figure 2.30).

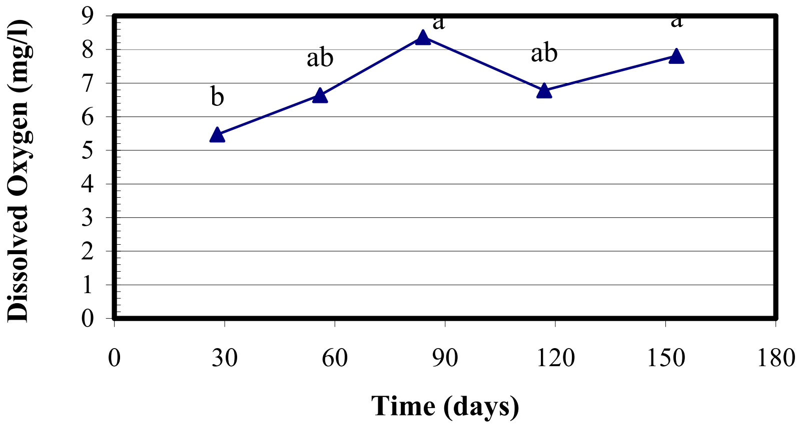

Figure 2.29 The variation in dissolved oxygen measurements in water from the Bottelary River study area over the sampling period in 2005. Concentrations marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.15 Water Temperature

Temperature plays an important role in water by affecting the rates of chemical reactions and the metabolic rates of organisms, as well as controlling the distribution of aquatic organisms. The temperatures determine the solubility of dissolved oxygen in water, its concentration and thus its availability to aquatic organisms (Anon b, 1996).

There were significant differences in the temperature of water over the study period. The water temperature generally ranged from 12.9 to 22.5 ºC. Figure 2.30 shows that the mean water temperature varied between 13.3 and 20.8 ºC and constantly decreased over the sampling period towards winter.

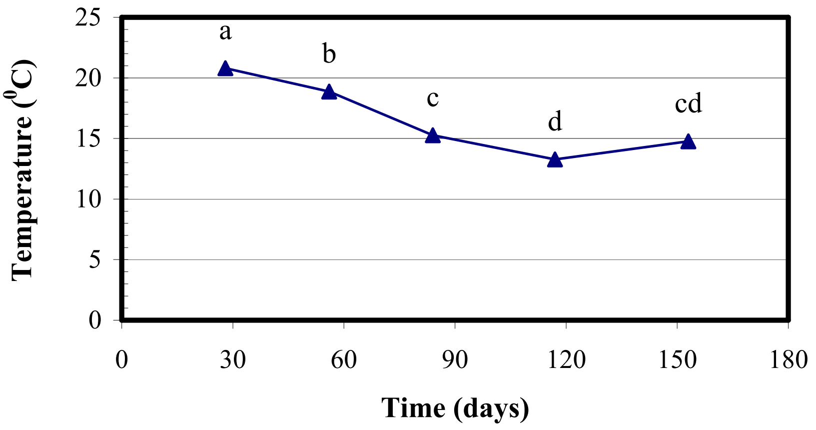

Figure 2.30 The variation in temperature measurements in water from the Bottelary River study area over the sampling period in 2005. Points marked with the same letter on the graph do not differ significantly ($p \leq 0.05$)

2.5.4.16 Pearson Correlation Coefficients

There were a series of significant correlations found between plant organs, sediments and water. A correlation found was that the dissolved oxygen in the water was negatively correlated with lead concentrations in the leaves and stems (Table 2.5). The iron concentrations in water were positively correlated with zinc concentrations in the leaves and stems. The nickel concentrations in stems, nitrogen and vanadium concentrations in leaves were positively correlated with iron concentrations in the water (Table 2.5). In addition, the water temperature was positively correlated with manganese concentrations in rhizomes and roots (Table 2.5).

Table 2.5 Significant Pearson Correlation Coefficients ($p \leq 0.05$) for plant organs in relation to water and sediments collected in Bottelary River area during the study period

Element	Plant Organs	Water	Sediments	N	Correlation Coefficient
Cadmium (Cd)	Rhizomes	Cr	-	6	0.82
	Roots	Temperature	-	5	–0.962
	Roots	Zn	-	6	–0.886
Chromium (Cr)	Leaves	-	Cd	6	0.868
Copper (Cu)	Rhizomes	Temperature	-	5	0.956
	Roots	DOP	-	5	–0.945
Iron (Fe)	Roots	Cu	-	6	0.855
	Stems	EC	-	6	–0.819
Manganese (Mn)	Leaves	NO_3^-	-	6	0.914
	Leaves	Cd	-	6	0.969
	Leaves	-	Pb	6	0.85
	Rhizomes	Temperature	-	5	0.938
	Rhizomes	Cr	-	6	–0.854
	Roots	Temperature	-	5	0.917
	Roots	Zn	-	6	0.935
	Stems	V	-	6	0.819
	Stems	Cr	-	6	0.898
	Stems	-	Cr	6	0.931
Nickel (Ni)	Stems	Fe	-	6	0.859
Lead (Pb)	Leaves	P	-	6	0.844
	Leaves	D.O	-	5	–0.989
	Roots	pH	-	6	–0.814
	Rhizomes	Cr	-	6	0.998
	Stems	Temperature	-	5	0.907
	Stems	DO	-	5	–0.915
	Stems	Ni	-	6	–0.910
Vanadium (V)	Leaves	Fe	-	6	0.835
Zinc (Zn)	Leaves	Fe	-	6	0.827
	Stems	Ni	-	6	0.832
	Stems	Fe	-	6	0.859
Phosphorus (P)	Leaves	-	Cd	6	0.816
	Roots	Cu	-	6	0.974
Total Nitrogen (TN)	Leaves	Fe	-	6	0.857
	Rhizomes	DOP	-	5	–0.895
	Rhizomes	NO_3^-	-	6	–0.823
	Rhizomes	-	V	6	–0.851

DOP = % dissolved oxygen; DO = mg oxygen per liter; EC = electrical conductivity

2.5.4.17 Quality of Water and Sediments

According to South Africa guideline for drinking water (SA), World Health Organization guideline for drinking water (WHO) and World Average of trace elements in unpolluted rivers (WA) (Meybeck & Helmer, 1989; Schiller & Boyle, 1987), the concentration ranges of the metals (Cd, Cu, Pb and Zn) were below the international guidelines and acceptable concentrations for drinking water. When compared to the world average of trace elements for unpolluted rivers, the river considered was polluted by cadmium (Table 2.6).

Table 2.6 Mean heavy metal concentration (mg/l) in the Bottelary River in Western Cape compared to international guidelines for drinking water

Metal	**Standards**			**This Study**
	SA[1]	**WHO**[2]	**WA**[3]	
Cd	0.005	0.003	0.001	0.0015
Cu	1	2	1.4	0.019
Pb	0.01	0.01	0.004	***
Zn	2	–	0.2	0.03

*** below detection limits

1. SA – South Africa guideline for drinking water
2. WHO – World Health Organization guideline for drinking water
3. WA – World Average of trace elements in unpolluted rivers

According to the South African Water Quality Guidelines for Ecosystems (Anon b, 1996), the results showed high concentrations of copper that were well above the target water quality range, chronic effect value and the acute effect value. The mean concentrations of zinc and cadmium were above the target water quality range and the chronic effect value but below the acute effect value. The mean and maximum concentration of manganese and lead from this study were below the target water quality range, chronic effect value and the acute effect value (Table 2.7).

Table 2.7 Mean heavy metal concentration (mg/l) found in the water and South Africa Water Guidelines for Ecosystem

This Study				**Standards**		
Water	**n**	**Mean**	**Max**	**TWQR**[1]	**CEV**[2]	**AEV**[3]
Cd	18	0.00155	0.0067	0.00015	0.0003	0.003
Cu	18	0.01934	0.0339	0.0003	0.00053	0.0016
Zn	18	0.02571	0.0579	0.0002	0.0036	0.036
Pb	18	***	***	0.0002	0.0005	0.0004
Mn	18	0.01116	0.0373	0.18	0.37	1.3

1) Target Water Quality Range (Anon b, 1996)
2) Chronic Effect Value (Anon b, 1996)
3) Acute Effect Value (Anon b, 1996)

*** below detection limits

According to the guidelines for the permissible utilization and disposal of sewage sludge (Anon c, 1997), the mean and the maximum concentrations of copper, lead and zinc in the sediments were within the maximum permissible concentrations allowed in soils and sludge respectively (Table 2.8).

Table 2.8 Mean concentrations (mg/g) found in the sediments and maximum permissible metal content in soil and sludge

This Study				**Max Standards**	
Sediments	**n**	**Mean**	**Max**	**Soil**	**Sludge**
Cu	18	0.0041	0.0201	0.0066	0.0505
Zn	18	0.0221	0.1135	0.0465	0.3535
Pb	18	0.0021	0.0074	0.0066	0.0505

2.5.4.18 Elements in Plants

According to the typical ranges of, and requirement for mineral nutrients found in plants (Larcher, 2003), the total nitrogen and phosphorus in both plants were within the required range for plant growth and the typical ranges, except for phosphorus in *T. capensis* were above the requirement. Iron and zinc in both plants were above the typical ranges and the requirement for plant growth. The nickel concentrations in both plants were above the typical ranges. The concentrations of manganese in both plants were above the required range. The lead concentrations in both plants were within the typical ranges (Table 2.9).

Table 2.9 Average concentrations of mineral nutrients in *Typha capensis* and *Phragmites australis* from this study compared with the typical ranges of, and requirement for mineral nutrients found in plants (Larcher, 2003)

Element	**Mean (mg/g)**		**Typical Ranges(mg/g)**	**Requirement (mg/g)**
	T. capensis	***P. australis***		
Cu	0.007	0.0061	0.004-0.02	0.005-0.01
Fe	1.8756	1.1887	0.002-0.7	ca. 0.1
Mn	0.1364	0.0847	0.003-1.0	0.03-0.05
N	1.94	2.03	12-75	15-25
Ni	0.00576	0.00655	to 0.005	–
P	3.64	2.63	0.1-10.0	1.5-3
Pb	0.00001	0.0004	to 0.02	–
Zn	0.129	0.0815	0.001-0.4	0.01-0.05

2.6 Conclusions

Concentrations in Water, Sediments and Plants:

The results show that most of element concentrations (Cd, Cr, Ni, P and Zn) in plants were higher than those in the sediments, and were much greater than those in the water. However for iron, lead, manganese and vanadium, the greatest concentration levels were found in the sediments. This suggests that most heavy metals were accumulated by plants from the sediments rather than from the water.

The mean and the maximum concentrations of copper, lead and zinc in the sediments were within the maximum permissible concentrations allowed in soils and sludge respectively (Anon c, 1997) (Table 2.8). Water samples had high concentrations of copper which were above the Acute Effect Value (AEV) and cadmium and zinc which were above the Chronic Effect Value (CEV) according to the South African Water Quality Guideline values for aquatic ecosystems (Anon b, 1996) (Table 2.7). The total nitrogen and phosphorus in both plants were within the required range for plant growth and the typical ranges, except for phosphorus in *T. capensis* was above the requirement. Iron and zinc in both plants were above the typical ranges and the requirement for plant growth. The nickel concentrations in both plants were above the typical ranges. The concentrations of manganese in both plants were above the required range. The lead concentrations in both plants were within the typical ranges (Table 2.9).

An abrupt increase of most heavy metal concentrations was seen between days 28 and 84. The increased concentrations of over the study period are thought to be primarily derived from the anthropogenic input of these constituents from agricultural and urban runoff because of increased rainfall. Due to increased runoff during the rainy season, the large amounts of water containing metal and nutrients compounds were transported via surface and subsurface runoff from the farmlands into the river. The increased concentrations of heavy metals in the river water could also have been due to increased suspension of sediments that entered the river through agricultural and urban runoff. The decrease in pH over time would also have increased the solubility of the metal ions.

High concentrations of nitrogen and phosphorus were also observed in the river, mainly coming from the De Novo Tributary. Eutrophication problems would be caused by these excessive nitrogen and phosphorus levels. There was no salinity problem in the river as revealed by the low levels of conductivity. These findings support the conclusions of Feng (2005).

Organs of Accumulation:

The concentrations (Cd, Cr, Cu, Fe, Mn, Ni, V and Zn) in the organs of plants confirm the ability of the roots to accumulate and attain the highest concentrations. But significantly more lead was accumulated in the stems than in the roots of *P. australis*. For these two plant species, most metal concentrations followed the descending order of roots, rhizomes, leaves and stems.

Role of Species:

The concentrations of chemicals varied in macrophytes according to the plant species and the particular element (Seidel, 1976). The tissues of *T. capensis* accumulate much more of iron, manganese, zinc than the tissues of *P. australis* (Table 2.2). Therefore, we can safely assume that *T. capensis* is proved more appropriate to be used as biological monitor for these elements. The lead levels in *P. australis* were found to be much higher than in *T. capensis* and therefore *P. australis* could be an appropriate biomonitor of lead.

Position Comparisons:

Higher mean concentrations of copper, manganese and vanadium were observed at site ‘Upstream’ on the farm ‘Vredehoek’ (Table 2.4). The high levels of copper could be attributed to agricultural runoff from the use of copper based pesticides on the farms. The reasons for the high manganese levels are at present not clear. The concentrations of chromium and nickel were highest in the De Novo Tributary on the farm ‘Willemsrus’. The high levels of these elements in the Bottelary River could be attributed to the effluent discharge from the Scottsdene WWTW (Nitrogen and Phsophorus – Feng, 2005) and the urban runoff from the Scottsdene canal. The mean cadmium value at site ‘Downstream’ was relatively high.

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Chapter 3

Summary and Recommendations: Monitoring of Heavy Metals in Bottelary River System by Macrophytes

3.1 Summary

The use of plants to determine the degree of heavy metal contamination in water and sediments was undertaken, in order to effectively monitor and provide possible recommendations to improve the water quality in the aquatic ecosystem of the Bottelary River.

Most of element concentrations (Cd, Cr, Ni, P and Zn) in plants were higher than those in the sediments, and were much greater than those in the water. However for iron, lead, manganese and vanadium, the greatest concentration levels were found in the sediments. This suggests that most heavy metals were accumulated by plants from the sediments. Over the study period, there were significant variations in the concentrations of the elements in the plants except for cadmium, copper, lead, nitrogen and phosphorus levels. Significant increases in the concentrations of chromium, nickel, vanadium and zinc in plants occurred.

The results from this study showed that roots consistently presented higher metal concentrations than either the rhizomes or above–ground organs, except for lead in the stems where concentrations were higher than in other organs. For these two plant species, most metal concentrations followed the descending order of roots, rhizomes, leaves and stems. Significant correlations were found between the plant organs, sediments and water. The dissolved oxygen in water was negatively correlated with lead concentrations in the leaves and stems. The iron concentrations in the water were positively correlated with the zinc concentrations in the leaves and stems. The nickel concentrations in stems, nitrogen and vanadium concentrations in leaves were positively correlated with iron concentrations in the water.

It would appear that *Typha capensis* would be useful for monitoring the heavy metals: iron, manganese and zinc. The lead levels in *Phragmites australis* were found to be much higher than in *T. capensis* and therefore *P. australis* could be a more appropriate biomonitor of lead. The accumulations of cadmium and copper were similar in both species.

High concentrations of copper, manganese and vanadium were observed at site ‘Upstream’ on the farm ‘Vredehoek’. The concentrations of chromium and nickel were highest in the De Novo Tributary from Scottsdene. The mean cadmium value at site ‘Downstream’ was relatively high. The main sources of pollutants in the Bottelary River were agricultural runoff, urban runoff from the Scottsdene Town and treated sewage effluent from the Scottsdene Wastewater Treatment Works (WWTW).

According to the South African Water Quality Guidelines for Ecosystems (Anon a, 1996) and Permissible utilization and disposal of sewage sludge (Anon b, 1997), the results showed too high levels of cadmium, copper and zinc in the water. These high levels of heavy metals in the water could cause toxicity to the aquatic organisms and terrestrial animals. The mean and the maximum concentrations of copper, lead and zinc in the sediments were within the maximum permissible concentrations allowed in soils and sludge respectively. Iron and zinc in both plants were above the required range for plant growth and the typical ranges (Larcher, 2003). Analyses of water, sediment and plant samples indicated that the Bottelary River was polluted with cadmium, copper and zinc.

3.2 Recommendations

In this study, the rooted accumulator *T. capensis* has been found to be better candidate than *P. australis* for heavy metal uptake in the river. The problem is the management of these plants because they are well known as aquatic weeds that grow very fast and easily block the river systems.

The pollution of the Bottelary River should be given more attention because of the high concentration of heavy metals (cadmium, copper and zinc) found during the full study period. The heavy metal contents in the Bottelary River and the De Novo tributary were affected by agricultural practices, urban pollution, the Scottsdene WWTW and climatic conditions in the catchment area. The main source of pollution for the Bottelary River is the De Novo tributary, which receives urban and agricultural runoff as well as effluents from the Scottsdene WWTW. Future studies should also look at the possible sources of heavy metals in the river.

In order to reduce the impact of urban runoff, we should consider if we need to plant a riparian buffer zone and create runoff containment zones, and possibly even expand the wetlands in the lower reaches of the De Novo tributary.

There is great need to gather long term data in order to evaluate heavy metal pollution in the Bottelary River and in other water bodies. Good biomonitoring tools as confirmed in this study will facilitate this.

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Appendix 4.1 Rainfall recorded at the Cape Town International Airport during the study period

Total Daily Rainfall (millimeters) 2005						
Date	**Feb.**	**Mar.**	**Apr.**	**May**	**Jun.**	**Jul.**
1					0.4	
2	0.8				9.2	
3	0.6				5.4	
4						
5		0.6		6		
6		0.2		0.2		
7					15.8	
8					4	
9		3.8				
10			44.4			
11			8			
12				20.9	14.8	
13				0.8	12.7	
14					4.8	
15						
16						
17		0.9		13.8		2.2
18			11.7		8.7	16.6
19			10.6		0.1	32
20			8.4	0.4	2.4	2.8
21		2.2				10.6
22			6.8	3.9		
23			2.7	0.2		
24	0.6	1	0.2			
25					10.5	0.2
26				7.7	1.4	0.2
27				2.4		
28						
29	***			3.8		
30	***		2.5	1		
31	***		***	16.6	***	
Total	**2**	**8.7**	**95.3**	**77.7**	**90.2**	**64.6**

Daily rainfall (in mm) – only rainfall ≥ 0.1 mm is reflected on this report (blank) indicates that no rain fell on that day

*** indicates that data is missing or not yet available in the current month

Appendix 4.2 Analytical parameters in the Bottelary River: raw data

Date	Days	Position	Water																	
			T^0C	DO	DO	EC	pH	NH_4^+	NO_2^-	NO_3^-	P	Cd	Cr	Cu	Fe	Mn	Ni	Pb	V	Zn
				%	mg/l	mS/cm		mg/l	mg/l	mg/l	mg/l	µg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
2005.2.17	1	Upstream	.	.	.	2.43	7.79	0.5	0.00	60	10.734	4.299	0.0006	0.0115	0.2124	0.016	0.000	0	0.0069	0.0428
2005.2.17	1	Tributary	.	.	.	1.71	7.65	0.5	0.00	80	30.084	0.266	0.0226	0.0141	0.076	0.000	0.016	0	0.0075	0.0106
2005.2.17	1	Downstream	.	.	.	1.98	7.62	0.5	0.25	60	9.167	0.139	0.0008	0.0257	0.6121	0.009	0.006	0	0.0054	0.0155
2005.3.17	28	Upstream	20.1	77.9	7.08	2.32	7.87	0.5	0.00	85	9.271	3.642	0.0015	0.0062	0.1466	0.000	0.000	0	0.0038	0.0284
2005.3.17	28	Tributary	19.8	32.0	2.92	2.44	7.65	0.0	0.03	50	27.881	1.454	0.0224	0.0139	0.2022	0.000	0.008	0	0.0044	0.0579
2005.3.17	28	Downstream	22.5	74.1	6.42	2.48	8.85	0.0	0.35	70	18.997	0.170	0.0010	0.0235	0.3826	0.019	0.012	0	0.0043	0.0225
2005.4.14	56	Upstream	17.1	83.9	8.07	2.44	7.66	0.0	0.05	100	8.646	4.896	0.0004	0.0328	0.3063	0.025	0.011	0	0.0075	0.051
2005.4.14	56	Tributary	20.5	53.0	5.80	1.34	7.14	2.0	0.75	70	25.089	0.257	0.0118	0.0339	0.1284	0.000	0.015	0	0.0045	0.0141
2005.4.14	56	Downstream	19.0	64.0	6.05	1.71	7.23	2.0	0.75	90	14.217	0.136	0.0009	0.026	0.5475	0.004	0.017	0	0.0038	0.0186
2005.5.12	84	Upstream	14.4	85.1	8.80	1.95	7.62	0.5	0.00	200	14.074	6.685	0.0038	0.0314	0.1937	0.026	0.018	0	0.0085	0.0565
2005.5.12	84	Tributary	16.5	70.5	7.20	1.52	7.33	1.0	10.00	50	23.586	1.662	0.0379	0.0065	0.1327	0.000	0.013	0	0.0072	0.013
2005.5.12	84	Downstream	14.9	90.4	9.11	1.82	7.44	0.5	4.00	200	8.746	0.561	0.0016	0.0293	1.064	0.037	0.013	0	0.0045	0.0189
2005.6.14	117	Upstream	12.9	71.6	7.61	1.95	7.49	0.5	0.00	25	9.796	0.541	0.0007	0.0139	0.1257	0.000	0.016	0	0.0041	0.0291
2005.6.14	117	Tributary	13.6	65.5	6.70	0.37	7.27	3.0	5.00	50	24.659	0.883	0.0411	0.0047	0.2113	0.029	0.010	0	0.0040	0.0137
2005.6.14	117	Downstream	13.3	58.4	6.04	1.40	7.79	1.0	5.00	50	18.711	0.252	0.0219	0.028	0.3125	0.009	0.018	0	0.0046	0.0219
2005.7.20	153	Upstream	14.5	78.0	7.82	1.46	6.91	0.0	0.00	60	7.410	1.184	0.0005	0.0153	0.2318	0.008	0.016	0	0.0091	0.0176
2005.7.20	153	Tributary	15.1	79.0	7.95	0.79	6.90	3.0	0.10	60	24.421	0.293	0.0432	0.0069	0.1792	0.000	0.022	0	0.0054	0.011
2005.7.20	153	Downstream	14.7	78.5	7.66	1.16	6.92	1.0	0.10	60	10.464	0.439	0.0013	0.0245	1.4775	0.018	0.019	0	0.0001	0.0197

Date	Days	Position	Sediments									
			P	Cd	Cr	Cu	Fe	Mn	Ni	Pb	V	Zn
			mg/g	µg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g
2005.2.17	1	Upstream	1.6740	0.0043	0.0117	0.0077	14.74	0.0825	0.0029	0.0005	0.0209	0.0214
2005.2.17	1	Tributary	0.1405	0.0058	0.0008	0.0005	0.87	0.0032	0.0002	0.0018	0.0017	0.0039
2005.2.17	1	Downstream	0.8255	0.0185	0.0023	0.0027	2.44	0.0213	0.0005	0.0045	0.0029	0.0135
2005.3.17	28	Upstream	0.3051	0.0020	0.0037	0.0017	3.62	0.0181	0.0006	0.0022	0.0101	0.0098
2005.3.17	28	Tributary	0.1279	0.0068	0.0010	0.0005	0.98	0.0040	0.0003	0.0016	0.0020	0.0069
2005.3.17	28	Downstream	0.3247	0.0128	0.0017	0.0019	1.81	0.0148	0.0003	0.0035	0.0016	0.0109
2005.4.14	56	Upstream	0.0484	0.0027	0.0044	0.0118	21.89	0.2946	0.0078	0.0004	0.0488	0.0648
2005.4.14	56	Tributary	0.1458	0.0058	0.0016	0.0006	1.60	0.0062	0.0004	0.0017	0.0032	0.0073
2005.4.14	56	Downstream	0.2191	0.0047	0.0032	0.0011	3.11	0.0162	0.0003	0.0036	0.0053	0.0084
2005.5.12	84	Upstream	5.6980	0.0343	0.0101	0.0201	31.20	0.9782	0.0072	0.0002	0.0908	0.1135
2005.5.12	84	Tributary	0.1588	0.0052	0.0017	0.0009	1.20	0.0047	0.0003	0.0021	0.0022	0.0066
2005.5.12	84	Downstream	0.3224	0.0229	0.0008	0.0032	2.91	0.0104	0.0008	0.0074	0.0046	0.0159
2005.6.14	117	Upstream	0.8643	0.0069	0.0052	0.0046	11.95	0.2063	0.0028	0.0004	0.0301	0.0220
2005.6.14	117	Tributary	0.3716	0.0115	0.0040	0.0024	3.50	0.0107	0.0004	0.0039	0.0028	0.0168
2005.6.14	117	Downstream	0.5292	0.0229	0.0007	0.0054	7.98	0.0719	0.0021	0.0008	0.0060	0.0318
2005.7.20	153	Upstream	0.4444	0.0082	0.0091	0.0047	8.41	0.1205	0.0020	0.0007	0.0260	0.0177
2005.7.20	153	Tributary	0.1920	0.0072	0.0011	0.0009	1.08	0.0082	0.0002	0.0022	0.0014	0.0071
2005.7.20	153	Downstream	0.5026	0.0142	0.0002	0.0037	5.68	0.0578	0.0014	0.0010	0.0140	0.0195

Date	Days	Position	Phagmites										
			Organs	TN	P	Cd	Cr	Cu	Fe	Mn	Ni	Pb	V	Zn
				mg/g	mg/g	µg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g
2005.2.17	1	Upstream	Roots	.	2.8455	0.2964	0.0065	0.0136	2.07	0.2247	0.0076	0.0000	0.0045	0.1341
2005.2.17	1	Upstream	Rhizomes	1.164	0.7931	0.0287	0.0185	0.0087	0.37	0.0383	0.0038	0.0002	0.0004	0.0419
2005.2.17	1	Upstream	Stems	1.322	1.5256	0.0347	0.0112	0.0044	0.12	0.1530	0.0035	0.0016	0.0002	0.0278
2005.2.17	1	Upstream	Leaves	3.090	1.8260	0.0203	0.0078	0.0024	0.29	0.2074	0.0041	0.0000	0.0002	0.0218
2005.2.17	1	Tributary	Roots	.	1.8054	0.1243	0.0828	0.0066	1.35	0.0650	0.0114	0.0005	0.0017	0.1606
2005.2.17	1	Tributary	Rhizomes	2.232	2.5595	0.0546	0.0166	0.0055	0.40	0.0172	0.0046	0.0010	0.0004	0.0732
2005.2.17	1	Tributary	Stems	0.989	0.8854	0.0463	0.0032	0.0012	0.14	0.0219	0.0028	0.0001	0.0001	0.0325
2005.2.17	1	Tributary	Leaves	3.143	1.7626	0.0094	0.0168	0.0015	0.21	0.0891	0.0039	0.0000	0.0003	0.0251
2005.2.17	1	Downstream	Roots	4.789	3.7750	0.6478	0.0563	0.0061	2.18	0.0808	0.0127	0.0000	0.0021	0.0920
2005.2.17	1	Downstream	Rhizomes	1.882	2.3270	0.0269	0.0051	0.0013	0.24	0.0102	0.0031	0.0002	0.0002	0.0232
2005.2.17	1	Downstream	Stems	1.532	2.7455	0.0246	0.0067	0.0034	0.18	0.0069	0.0031	0.0015	0.0001	0.0164
2005.2.17	1	Downstream	Leaves	.	2.8950	0.0309	0.0047	0.0088	0.27	0.0317	0.0034	0.0011	0.0002	0.0381
2005.3.17	28	Upstream	Roots	.	0.8182	0.1093	0.0072	0.0196	3.54	0.5224	0.0006	0.0000	0.0073	0.1261
2005.3.17	28	Upstream	Rhizomes	1.042	1.5229	0.0150	0.0092	0.0053	0.30	0.0263	0.0063	0.0004	0.0003	0.0290
2005.3.17	28	Upstream	Stems	1.269	1.3580	0.0137	0.0061	0.0036	0.14	0.0200	0.0022	0.0007	0.0001	0.0210
2005.3.17	28	Upstream	Leaves	.	2.9912	0.0262	0.0089	0.0044	0.39	0.0732	0.0056	0.0012	0.0003	0.0392
2005.3.17	28	Tributary	Roots	.	4.0798	0.1896	0.0152	0.0099	3.02	0.1152	0.0048	0.0007	0.0031	0.3683
2005.3.17	28	Tributary	Rhizomes	1.952	1.9901	0.0460	0.0115	0.0025	0.24	0.0233	0.0036	0.0003	0.0002	0.0646
2005.3.17	28	Tributary	Stems	2.005	2.5587	0.0216	0.0076	0.0025	0.19	0.0117	0.0028	0.0011	0.0001	0.0417
2005.3.17	28	Tributary	Leaves	3.493	2.2334	0.1193	0.0097	0.0065	0.28	0.0356	0.0029	0.0004	0.0002	0.0432
2005.3.17	28	Downstream	Roots	.	3.2421	0.1969	0.0099	0.0116	6.63	0.4214	0.0042	0.0002	0.0099	0.2666
2005.3.17	28	Downstream	Rhizomes	2.425	1.5290	0.0209	0.0116	0.0021	0.89	0.0211	0.0057	0.0000	0.0015	0.0230
2005.3.17	28	Downstream	Stems	1.024	1.8581	0.0147	0.0056	0.0054	0.19	0.0062	0.0034	0.0009	0.0002	0.0173
2005.3.17	28	Downstream	Leaves	3.003	1.0795	0.0578	0.0108	0.0029	0.45	0.0346	0.0039	0.0000	0.0007	0.0294

Date	Days	Position	Phagmites											
			Organs	TN	P	Cd	Cr	Cu	Fe	Mn	Ni	Pb	V	Zn
				mg/g	mg/g	µg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g
2005.4.14	56	Upstream	Roots	.	38.6299	0.1596	0.0037	0.0120	11.41	0.4247	0.0000	0.0010	0.0066	0.0966
2005.4.14	56	Upstream	Rhizomes	.	1.5440	0.0435	0.0068	0.0045	0.31	0.0420	0.0023	0.0002	0.0002	0.0334
2005.4.14	56	Upstream	Stems	.	0.6588	0.0054	0.0037	0.0026	0.17	0.0282	0.0034	0.0003	0.0001	0.0194
2005.4.14	56	Upstream	Leaves	3.195	2.6178	0.0081	0.0053	0.0002	0.32	0.0540	0.0039	0.0010	0.0001	0.0346
2005.4.14	56	Tributary	Roots	.	5.0302	0.2077	0.0095	0.0089	5.72	0.1126	0.0019	0.0012	0.0046	0.1434
2005.4.14	56	Tributary	Rhizomes	.	4.0685	0.1141	0.0796	0.0119	2.03	0.0537	0.0322	0.0000	0.0017	0.1218
2005.4.14	56	Tributary	Stems	1.409	1.9264	0.0198	0.0038	0.0018	0.21	0.0198	0.0035	0.0011	0.0001	0.0941
2005.4.14	56	Tributary	Leaves	.	0.9223	0.0042	0.0015	0.0020	0.20	0.0544	0.0016	0.0001	0.0002	0.0234
2005.4.14	56	Downstream	Roots	.	2.0586	0.1451	0.0068	0.0120	2.40	0.0598	0.0041	0.0000	0.0175	0.1159
2005.4.14	56	Downstream	Rhizomes	.	2.7843	0.0201	0.0239	0.0238	0.61	0.0107	0.0094	0.0005	0.0006	0.0446
2005.4.14	56	Downstream	Stems	.	1.5922	0.0178	0.0061	0.0038	0.13	0.0218	0.0028	0.0000	0.0002	0.0195
2005.4.14	56	Downstream	Leaves	2.956	2.0574	0.0104	0.0083	0.0021	0.23	0.0596	0.0033	0.0000	0.0002	0.0219
2005.5.12	84	Upstream	Roots	.	4.5934	0.2582	0.0147	0.0206	4.61	0.4912	0.0063	0.0002	0.0145	0.2139
2005.5.12	84	Upstream	Rhizomes	.	2.1460	0.0163	0.0145	0.0069	0.57	0.0647	0.0038	0.0000	0.0012	0.0352
2005.5.12	84	Upstream	Stems	.	0.4046	0.0398	0.0033	0.0035	0.14	0.0260	0.0021	0.0000	0.0001	0.0355
2005.5.12	84	Upstream	Leaves	2.075	1.6244	0.0190	0.0129	0.0015	0.53	0.3707	0.0061	0.0000	0.0004	0.0441
2005.5.12	84	Tributary	Roots	.	2.6281	0.1179	0.0176	0.0056	2.58	0.1842	0.0122	0.0002	0.0064	0.1113
2005.5.12	84	Tributary	Rhizomes	.	1.1927	0.0229	0.0055	0.0013	0.43	0.0197	0.0037	0.0000	0.0003	0.0461
2005.5.12	84	Tributary	Stems	1.515	2.3243	0.0086	0.0095	0.0011	0.25	0.0793	0.0042	0.0000	0.0001	0.0958
2005.5.12	84	Tributary	Leaves	.	2.0592	0.0409	0.0120	0.0031	0.36	0.2743	0.0074	0.0000	0.0004	0.0991
2005.5.12	84	Downstream	Roots	.	2.1999	0.4423	0.0279	0.0037	3.59	0.0482	0.0107	0.0003	0.0088	0.1505
2005.5.12	84	Downstream	Rhizomes	.	1.5888	0.0847	0.0153	0.0150	0.33	0.0163	0.0079	0.0007	0.0006	0.0680
2005.5.12	84	Downstream	Stems	1.287	1.4841	0.0241	0.0030	0.0091	0.20	0.0024	0.0059	0.0003	0.0001	0.0409
2005.5.12	84	Downstream	Leaves	2.793	1.8536	0.0208	0.0030	0.0036	0.17	0.0111	0.0016	0.0000	0.0004	0.0299

Date	Days	Position	Phagmites											
			Organs	TN	P	Cd	Cr	Cu	Fe	Mn	Ni	Pb	V	Zn
				mg/g	mg/g	µg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g
2005.6.14	117	Upstream	Roots	.	0.7509	0.0767	0.0363	0.0127	2.94	0.1842	1.3590	0.0001	0.0106	0.0944
2005.6.14	117	Upstream	Rhizomes	.	1.1257	0.0213	0.0118	0.0067	0.52	0.0350	0.6345	0.0000	0.0013	0.1007
2005.6.14	117	Upstream	Stems	.	0.2268	0.0220	0.0086	0.0032	0.26	0.0326	0.4876	0.0000	0.0005	0.0418
2005.6.14	117	Upstream	Leaves	.	1.0476	0.0174	0.0057	0.0066	0.32	0.0332	0.2440	0.0000	0.0001	0.0365
2005.6.14	117	Tributary	Roots	.	3.6287	0.2856	0.0133	0.0045	1.64	0.0498	0.8356	0.0012	0.0080	0.1385
2005.6.14	117	Tributary	Rhizomes	.	2.9259	0.0515	0.0212	0.0079	0.49	0.0297	0.8509	0.0003	0.0008	0.0953
2005.6.14	117	Tributary	Stems	.	2.5564	0.0427	0.0079	0.0040	0.38	0.0318	0.4656	0.0007	0.0002	0.0798
2005.6.14	117	Tributary	Leaves	.	2.1239	0.0302	0.0133	0.0048	0.30	0.1256	0.4964	0.0000	0.0003	0.0910
2005.6.14	117	Downstream	Roots	.	2.9422	0.4091	0.0171	0.0105	2.95	0.0929	1.5425	0.0005	0.0076	0.1600
2005.6.14	117	Downstream	Rhizomes	2.372	3.1920	0.0276	0.0083	0.0043	0.68	0.0187	1.1363	0.0004	0.0004	0.0617
2005.6.14	117	Downstream	Stems	.	1.6222	0.0447	0.0045	0.0024	0.19	0.0086	0.2555	0.0000	0.0003	0.0458
2005.6.14	117	Downstream	Leaves	.	3.2147	0.0412	0.0037	0.0014	0.20	0.0597	0.2817	0.0008	0.0001	0.0715
2005.7.20	153	Upstream	Roots	.	1.0992	0.0997	0.0317	0.0100	1.80	0.0828	1.6140	0.0000	0.0074	0.1674
2005.7.20	153	Upstream	Rhizomes	1.094	1.7889	0.0008	0.0072	0.0016	0.13	0.0100	0.0356	0.0000	0.0002	0.0058
2005.7.20	153	Upstream	Stems	0.709	0.7936	0.0116	0.0066	0.0023	0.19	0.0229	0.2956	0.0000	0.0001	0.2284
2005.7.20	153	Upstream	Leaves	.	2.1972	0.0450	0.0158	0.0008	0.57	0.1517	0.9640	0.0000	0.0008	0.0695
2005.7.20	153	Tributary	Roots	.	3.2206	0.0732	0.1235	0.0112	7.22	0.1230	5.1135	0.0042	0.0093	0.1510
2005.7.20	153	Tributary	Rhizomes	1.182	2.3617	0.0429	0.0134	0.0031	0.61	0.0240	0.7161	0.0008	0.0007	0.1925
2005.7.20	153	Tributary	Stems	.	2.0238	0.0297	0.0138	0.0016	0.22	0.0202	0.6229	0.0001	0.0002	0.0950
2005.7.20	153	Tributary	Leaves	.	2.7248	0.0174	0.0061	0.0033	0.22	0.1011	0.6007	0.0001	0.0001	0.0490
2005.7.20	153	Downstream	Roots	.	1.7362	0.3001	0.0214	0.0080	1.38	0.0318	0.7980	0.0000	0.0044	0.2245
2005.7.20	153	Downstream	Rhizomes	.	1.8398	0.0457	0.0181	0.0108	0.45	0.0231	1.1034	0.0000	0.0007	0.0724
2005.7.20	153	Downstream	Stems	.	4.4693	0.0249	0.0098	0.0109	0.28	0.0227	1.5640	0.0004	0.0003	0.0966
2005.7.20	153	Downstream	Leaves	.	0.7310	0.0030	0.0058	0.0045	0.08	0.0000	1.3267	0.0001	1.8445	0.0000

Date	Days	Position	Typha											
			Organs	TN	P	Cd	Cr	Cu	Fe	Mn	Ni	Pb	V	Zn
				mg/g	mg/g	µg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g
2005.2.17	1	Upstream	Roots	0.481	3.8848	0.1010	0.0065	0.0139	3.54	0.1951	0.0007	0	0.0094	0.0671
2005.2.17	1	Upstream	Rhizomes	1.865	3.8135	0.0337	0.0071	0.0082	0.71	0.0379	0.0036	0	0.0006	0.0583
2005.2.17	1	Upstream	Leaves	2.320	2.8169	0.0161	0.0055	0.0043	0.29	0.0903	0.0020	0	0.0002	0.0191
2005.2.17	1	Tributary	Roots	1.945	2.8884	0.2314	0.0291	0.0042	8.28	0.1933	0.0156	0	0.0040	0.1899
2005.2.17	1	Tributary	Rhizomes	.	3.0503	0.0469	0.0536	0.0110	1.04	0.0214	0.0133	0	0.0005	0.0696
2005.2.17	1	Tributary	Leaves	1.847	2.5002	0.0188	0.0015	0.0041	0.26	0.1382	0.0000	0	0.0002	0.0230
2005.2.17	1	Downstream	Roots	1.050	5.8057	0.0801	0.0154	0.0052	3.58	0.0888	0.0067	0	0.0021	0.0770
2005.2.17	1	Downstream	Rhizomes	1.970	3.6497	0.1381	0.0159	0.0001	0.87	0.0203	0.0067	0	0.0006	0.0553
2005.2.17	1	Downstream	Leaves	2.407	2.9454	0.0215	0.0019	0.0033	0.27	0.0549	0.0037	0	0.0002	0.0221
2005.3.17	28	Upstream	Roots	.	4.1001	0.1235	0.0098	0.0063	4.98	0.7926	0.0043	0	0.0135	0.0794
2005.3.17	28	Upstream	Rhizomes	1.760	3.5831	0.0731	0.0168	0.0481	1.08	0.0957	0.0046	0	0.0017	0.0709
2005.3.17	28	Upstream	Leaves	2.180	2.3181	0.0132	0.0036	0.0052	0.36	0.3125	0.0011	0	0.0001	0.0111
2005.3.17	28	Tributary	Roots	.	2.9815	0.0632	0.0110	0.0088	2.55	0.0512	0.0058	0	0.0012	0.0845
2005.3.17	28	Tributary	Rhizomes	2.040	4.0287	0.0218	0.0121	0.0020	0.86	0.0253	0.0079	0	0.0007	0.1017
2005.3.17	28	Tributary	Leaves	1.882	3.1282	0.0146	0.0024	0.0043	0.23	0.0967	0.0000	0	0.0002	0.0229
2005.3.17	28	Downstream	Roots	.	4.6792	0.0693	0.0161	0.0005	5.01	0.3076	0.0082	0	0.0017	0.0614
2005.3.17	28	Downstream	Rhizomes	2.267	4.2197	0.0199	0.0070	0.0027	0.44	0.0616	0.0039	0	0.0003	0.0495
2005.3.17	28	Downstream	Leaves	1.655	2.6014	0.0146	0.0056	0.0022	0.34	0.2206	0.0054	0	0.0002	0.0213

Date	Days	Position	Typha											
			Organs	TN	P	Cd	Cr	Cu	Fe	Mn	Ni	Pb	V	Zn
				mg/g	mg/g	µg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g
2005.4.14	56	Upstream	Roots	.	4.5734	0.3149	0.0073	0.0217	6.72	1.2014	0.0007	0.0000	0.0256	0.1451
2005.4.14	56	Upstream	Rhizomes	1.917	5.1140	0.0278	0.0074	0.0159	0.54	0.0722	0.0054	0.0000	0.0008	0.1330
2005.4.14	56	Upstream	Leaves	2.583	2.9103	0.0112	0.0023	0.0077	0.25	0.2452	0.0013	0.0000	0.0001	0.0365
2005.4.14	56	Tributary	Roots	.	1.9180	0.1142	0.0099	0.0021	4.59	0.0483	0.0055	0.0000	0.0054	0.0857
2005.4.14	56	Tributary	Rhizomes	1.287	1.6254	0.0459	0.0070	0.0039	1.77	0.0315	0.0036	0.0000	0.0024	0.0394
2005.4.14	56	Tributary	Leaves	1.725	1.4909	0.0208	0.0033	0.0014	0.30	0.1157	0.0015	0.0000	0.0005	0.0311
2005.4.14	56	Downstream	Roots	.	5.9679	0.0186	0.0051	0.0002	4.84	0.0630	0.0045	0.0000	0.0020	0.1258
2005.4.14	56	Downstream	Rhizomes	.	5.3150	0.0282	0.0051	0.0024	0.66	0.0411	0.0029	0.0001	0.0006	0.0378
2005.4.14	56	Downstream	Leaves	.	2.2924	0.0252	0.0019	0.0048	0.36	0.2924	0.0046	0.0000	0.0001	0.0171
2005.5.12	84	Upstream	Roots	.	6.9301	0.1093	0.0090	0.0071	6.13	0.3354	0.0033	0.0000	0.0047	0.0667
2005.5.12	84	Upstream	Rhizomes	1.970	4.2753	0.0218	0.0026	0.0030	1.03	0.0441	0.0008	0.0000	0.0007	0.0342
2005.5.12	84	Upstream	Leaves	2.653	4.2172	0.0366	0.0046	0.0072	0.49	0.2686	0.0016	0.0000	0.0006	0.0650
2005.5.12	84	Tributary	Roots	.	6.3599	0.0928	0.0153	0.0022	6.81	0.2677	0.0125	0.0000	0.0063	0.1971
2005.5.12	84	Tributary	Rhizomes	0.622	1.9226	0.0159	0.0046	0.0025	1.31	0.0556	0.0052	0.0000	0.0017	0.0482
2005.5.12	84	Tributary	Leaves	.	3.4507	0.0486	0.0188	0.0051	0.44	0.0357	0.0069	0.0000	0.0005	0.0611
2005.5.12	84	Downstream	Roots	.	6.1956	0.1493	0.0063	0.0041	6.63	0.0849	0.0088	0.0000	0.0028	0.0855
2005.5.12	84	Downstream	Rhizomes	.	4.3283	0.0464	0.0035	0.0128	0.76	0.0253	0.0058	0.0000	0.0006	0.0849
2005.5.12	84	Downstream	Leaves	.	2.9862	0.0753	0.0031	0.0058	0.68	0.0660	0.0088	0.0000	0.0005	0.0488

Date	Days	Position	Typha											
			Organs	TN	P	Cd	Cr	Cu	Fe	Mn	Ni	Pb	V	Zn
				mg/g	mg/g	µg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g	mg/g
2005.6.14	117	Upstream	Roots	.	2.6033	0.1377	0.0158	0.0124	2.44	0.0443	0.0083	0.0000	0.0029	0.1758
2005.6.14	117	Upstream	Rhizomes	.	1.9053	0.0188	0.0142	0.0073	0.61	0.0398	0.0046	0.0000	0.0013	0.0280
2005.6.14	117	Upstream	Leaves	2.092	2.7686	0.0174	0.0057	0.0066	0.32	0.0332	0.0024	0.0000	0.0001	0.0365
2005.6.14	117	Tributary	Roots	.	4.2862	0.1249	0.0151	0.0164	4.13	0.2320	0.0146	0.0000	0.0057	0.3085
2005.6.14	117	Tributary	Rhizomes	.	5.0757	0.0356	0.0220	0.0079	1.03	0.0322	0.0065	0.0000	0.0017	0.1073
2005.6.14	117	Tributary	Leaves	.	4.6283	0.0564	0.0078	0.0098	0.68	0.1008	0.0048	0.0001	0.0009	0.0785
2005.6.14	117	Downstream	Roots	.	1.4568	0.3606	0.0248	0.0033	1.37	0.0239	0.0135	0.0000	0.0025	0.1172
2005.6.14	117	Downstream	Rhizomes	1.585	2.6212	0.0357	0.0046	0.0050	0.34	0.0402	0.0042	0.0000	0.0003	0.0562
2005.6.14	117	Downstream	Leaves	.	2.8484	0.0625	0.0037	0.0038	0.30	0.1080	0.0045	0.0000	0.0003	0.0646
2005.7.20	153	Upstream	Roots	.	2.7321	0.1641	0.0297	0.0096	1.59	0.0387	0.0114	0.0000	0.0033	0.4892
2005.7.20	153	Upstream	Rhizomes	.	3.0106	0.0273	0.0094	0.0085	0.71	0.0716	0.0057	0.0000	0.0009	0.0968
2005.7.20	153	Upstream	Leaves	.	2.9894	0.0169	0.0040	0.0066	0.39	0.1036	0.0025	0.0000	0.0001	0.3042
2005.7.20	153	Tributary	Roots	.	5.8116	0.1038	0.0238	0.0028	5.04	0.1314	0.0112	0.0001	0.0094	0.3500
2005.7.20	153	Tributary	Rhizomes	2.320	4.5144	0.0318	0.0219	0.0047	0.77	0.0384	0.0099	0.0000	0.0006	0.2600
2005.7.20	153	Tributary	Leaves	3.055	2.1427	0.0265	0.0063	0.0040	0.51	0.0957	0.0073	0.0000	0.0004	0.3487
2005.7.20	153	Downstream	Roots	.	3.8106	0.6808	0.0136	0.0067	1.21	0.0326	0.0105	0.0000	0.0021	1.2943
2005.7.20	153	Downstream	Rhizomes	.	5.9642	0.0716	0.0119	0.0101	0.51	0.0575	0.0082	0.0000	0.0005	0.1755
2005.7.20	153	Downstream	Leaves	3.090	2.5815	0.0270	0.0028	0.0052	0.31	0.0449	0.0041	0.0000	0.0002	0.2469

. indicates that data is missing or not yet available

T^0C: Temperature

DO: Dissolved oxygen

EC: Electrical conductivity

NH_4^+: Ammonium ion

NO_2^- : Nitrite

NO_3^- : Nitrate

TN: Total nitrogen

P: Phosphate

Cd: Cadmium

Cr: Chromium

Cu: Copper

Fe: Iron

Mn: Manganese

Ni: Nickel

Pb: Lead

V: Vanadium

Zn: Zinc