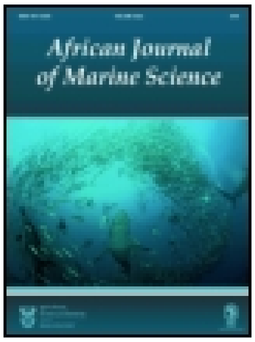

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Long-term physical, chemical and biological changes in a small, urban estuary

M Viskich[1], CL Griffiths[1]*, C Erasmus[2] and S Lamberth[2,3]

[1] Department of Biological Sciences and Centre for Invasion Biology, University of Cape Town, Cape Town, South Africa
[2] Branch: Fisheries Management, Department of Agriculture, Forestry and Fisheries, Cape Town, South Africa
[3] South African Institute for Aquatic Biodiversity, Grahamstown, South Africa
* Corresponding author, e-mail: charles.griffiths@uct.ac.za

The Diep River estuary, a small estuary in suburban Cape Town, South Africa, has been subject to disturbance for centuries. Several earlier studies have documented conditions in the system, providing baselines against which to measure more recent changes. This study: (i) describes major physical and hydrological changes that have occurred within this estuary; (ii) documents faunistic changes subsequent to earlier biological surveys; and (iii) provides an up-to-date faunal list. Salinity measurements and both invertebrate and fish samples were taken at five stations along the estuary in summer and winter 2014. A census of sandprawn *Callichirus kraussi* densities was also undertaken to compare with earlier surveys. Developments within the Diep River catchment and estuary have resulted in extensive changes in flow and salinity regimes, causing marked reductions in summer salinity levels, changes in frequency of mouth closure, and deteriorations in water quality. These have resulted in major changes in faunal composition and distribution, including an increase in numbers of non-indigenous species. Surveys in the early 1950s recorded 47 invertebrate species, whereas only 23 were found in 1974. A total of 23 species were again recorded in 2014, but these included several freshwater forms not previously reported, which had entered the system due to lowered salinity values, as well as new alien introductions. Only six of the 69 taxa recorded were reported by all three surveys. There have been substantial declines in sandprawn abundance, from 40 million in 1998 to just over 12 million in 2014. In all, 12 fish species were recorded in the 1950s, nine in 1974, but only five in 2014, including the newly detected invasive mosquito fish *Gambusia affinis* and the translocated tilapia *Tilapia sparrmanii*. Thus, only three of the original native fish species remain. Contrary to these losses, the present bird fauna appear to be more abundant and diverse than previously. Regular monitoring is recommended to obtain a clearer understanding of ongoing changes, and major management interventions will be needed if further degradation is to be prevented.

Keywords: alien introductions, *Callichirus kraussi*, Diep River estuary, *Gambusia affinis*, invasive species, Milnerton Lagoon, *Orchestia gammarella*

Introduction

Estuaries provide a wide range of ecologically important services, including as nursery areas for juvenile fish, as roosting and feeding sites for resident and migratory bird species, and as biological filters that break down waste and detoxify pollution (Jackson et al. 2011; van Niekerk and Turpie 2012; van Niekerk et al. 2013). They also constitute important recreational and aesthetic resources. However, estuaries are among the most extensively modified and threatened of aquatic environments (Blaber et al. 2000; McQuaid 2013). The threats they face include dredging operations, residential developments, bank and mouth stabilisation, water abstraction, organic and trace-metal pollution, exploitation of living resources, and introduction of non-native species (Blaber et al. 2000; Gutiérrez et al. 2012). Urban estuaries are particularly vulnerable to transformation, due to their usually long histories of development, exploitation and urbanisation, and hence it is challenging to restore them to a condition resembling their 'natural' state.

There are about 300 functional estuaries along the South African coast, but the great majority of these are on the warm-temperate South and subtropical East coasts, with only 16 being on the cool-temperate West Coast, west of Cape Point (van Niekerk et al. 2013, 2015). The Diep River estuary is one of these few West Coast systems and is an excellent example of a small, urban estuary that has been subjected to long-term modifications and disturbance (Clark 1998). The estuary lies approximately 5 km north of the centre of Cape Town (Western Cape province, South Africa) (33°54′ S, 18°28′ E) and within its suburbs. The catchment area is approximately 1 125 km^2 and consists mainly of industrial, agricultural and residential (formal and informal) areas (Taljaard et al. 1992; Clark 1998; Paulse et al. 2009).

The estuary extends from the Diep River mouth up to Blaauwberg Road bridge and covers an area of approximately 900 ha (Retief 2011), but consists of two very distinct sections, Rietvlei and Milnerton Lagoon (Figure 1a). Rietvlei is a large, roughly triangular wetland about 2 km × 1.5 km, situated at the head of the system. It is generally shallow, apart from two dredged lakes in the north-west corner, and is flooded during winter, but largely dries out during the dry summer months (Clark 1998). It is one of the

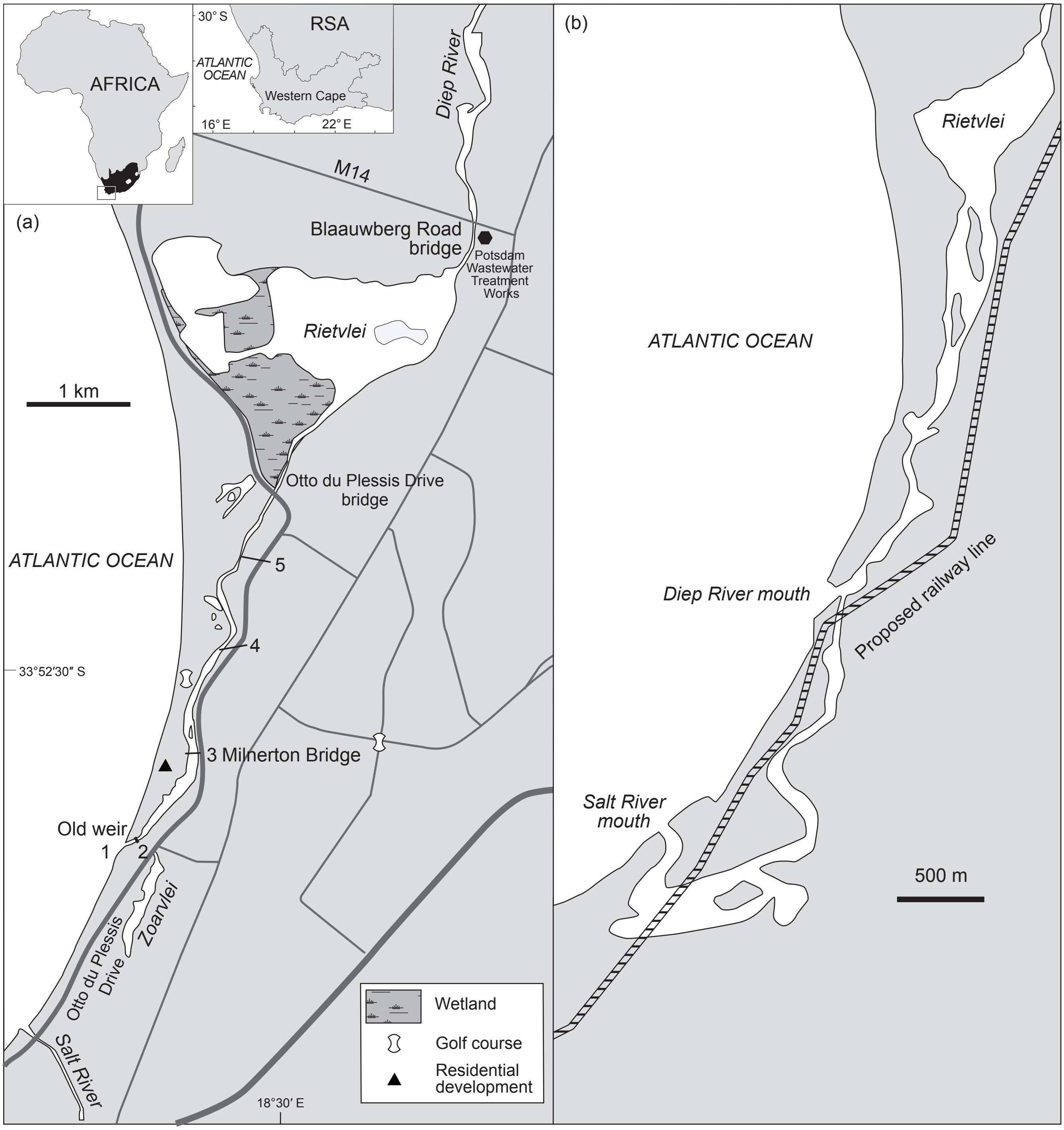

Figure 1: (a) Map of the Diep River estuary, showing the five observation stations (numbered 1–5) and other locations and features mentioned in the text (map adapted from Clark [1998] and Google Earth ['Milnerton Lagoon'; 33°89′ S, 18°48′ E]) and (b) historical map of the estuary (1846–1893), adapted from survey plans drawn by the Cape Town Municipality in 1893 (Western Cape Archives and Records) and by Capt. Sir Edward Belcher in 1846 (Grindley and Dudley 1988)

most important areas for waterbirds in the south-western Cape and has been ranked within the top 10% of all coastal wetlands in this region, in terms of numbers of birds present (Jackson et al. 2008; Retief 2011).

The lower section of the estuary, termed the Milnerton Lagoon, extends from the mouth upstream to Otto du Plessis Drive; it is small (maximum width 150 m) and shallow (maximum depth 3 m), and bordered by a major road to the east and a golf course and the Woodbridge Island residential development to the west (Jackson et al. 2008) (Figure 1a). Earlier studies report that the mouth closed seasonally during the dry summer months

(November–March) and opened during the wet winter period (June–August) (Millard and Scott 1954; JG Weil, University of Cape Town, unpublished data[1]; Grindley and Dudley 1988). This resulted in hypersaline conditions and a reverse salinity gradient developing during summer, when certain parts of the estuary resembled salt pans (Millard and Scott 1954; Botes and Le Roux 2004). Notably, the salt content in the estuary was derived not only from seawater intrusion via the mouth, but also from the river water, which is relatively alkaline and high in salt, derived from the Malmesbury shales of the catchment (Millard and Scott 1954; Retief 2011).

Freshwater flow into Milnerton Lagoon comes from a variety of sources. The original freshwater source is the Diep River itself, which flows into the north-eastern corner of the Rietvlei wetlands (Retief 2011) and then onwards into the lagoon. However, most of this water has now been lost to agricultural abstraction upstream and has been replaced by treated effluent from the Potsdam Wastewater Treatment Works, which flows into Rietvlei (Retief 2011). Additional water sources include flow from stormwater drains along the eastern bank of the Rietvlei wetlands and a natural channel draining surface runoff into the western side of Rietvlei (Retief 2011). The lagoon region of the estuary represents some 10% of the nursery area for fish on the West Coast (Jackson et al. 2008) and several fish species, such as Cape stumpnose *Rhabdosargus holubi* and white steenbras *Lithognathus lithognathus*, depend on this and other regional estuaries as nursery sites (Jackson et al. 2008).

In 1984, the Diep River estuary was recognised officially as a nature reserve (Jackson et al. 2008), after which the Rietvlei Wetland Reserve was established in 1993 (Jackson et al. 2008). Since then, there have been several plans developed to guide management of the estuary (Jackson et al. 2008, 2011; Retief 2011). Despite these initiatives and the recognised ecological importance of the lagoon, new developments continue to encroach within the boundaries of the estuary (Jackson et al. 2008, 2011). The hydrodynamics of the system have also been radically altered, water quality is increasingly becoming a public health concern and several non-native fish and plant species have invaded the area (Jackson et al. 2008).

Several earlier studies have documented the fauna of the Diep River estuary. Millard and Scott (1954) provided the first biological survey of the system, while Scott (1954) published a short descriptive account of the bird fauna. Since then, Weil (unpublished data) carried out an ecological survey of the lagoon section only in 1974, duplicating the methods of Millard and Scott (1954). Du Toit (1982) also conducted an ecological survey of the estuary. A comprehensive review of all information available on the system at the time was produced by Grindley and Dudley (1988). Bait-collecting activities, including a population estimate for the exploited sandprawn *Callichirus* (then *Callianassa*) *kraussi*, were reported by Clark (1998), and that survey was repeated by Retief (2003). Hutchings and Clark (2010) also measured trace-metal concentrations, both in the sediments and in representative biota. Other recent reports have discussed management plans for the system, but without collecting appreciable additional data (Jackson et al. 2008, 2011; Retief 2011).

The aims of this study were to carry out a new survey of the aquatic fauna and to document major physical, hydrological and faunistic changes that have occurred within and around the Deep River estuary to date, based on comparisons with earlier survey information provided by Millard and Scott (1954), Weil (unpublished data) and Clark (1998). We also provide an updated list of fauna inhabiting the estuary and tabulate this in comparison with data extracted from earlier surveys, and comment on the current state and future management of the system.

Material and methods

Physical and structural changes

The main historical physical and structural changes that have occurred within and around the estuary are presented in the form of a table derived from earlier publications, particularly Grindley and Dudley (1988). Historical maps were obtained from the Western Cape Archives and Records Service.

Changes in salinity

Salinity measurements were taken using a portable salinity meter during late summer (April) and at the end of winter (August) 2014. Measurements were taken around the time of high tide at the same observation stations used by Millard and Scott (1954) and Weil (unpublished data) (Figure 1). Millard and Scott (1954) expressed salt content in terms of chlorinity. We therefore converted their average surface chlorinities (g l^{-1}) taken from 1948 to 1953 to salinity using the equation: salinity = 1.806 × chlorinity (Lewis 1980). We realise that this provides only an approximation of total salinity, due to the varying proportion of different salts in the system, but consider this adequate for the purposes of the study.

Faunistic changes

The faunistic surveys involved three separate components.

Infauna, epifauna and nekton

Methods of specimen collection replicated those used by Millard and Scott (1954) and Weil (unpublished data) as closely as possible. Line transects were undertaken in summer (April) and winter (August). The five observation stations used by Weil (unpublished data) were again selected. The first four stations used by Millard and Scott (1954) corresponded with our stations 2–5 (Figure 1), although these authors also recorded fauna found within the mouth section of the lagoon. At each station we collected four sediment samples, each of approximate area 30 cm × 30 cm and depth 30 cm, located along a transect starting on the lagoon bank and ending in the centre of the lagoon. A handnet was used to collect organisms in the water column at each station. All samples were passed through a 1-mm sieve before examination in a sorting tray. Species were identified to the lowest possible taxonomic level and then preserved in 70% alcohol. Fully terrestrial species collected along the lagoon margins, or that had fallen into the water, were excluded.

[1] Weil JG. 1974. Ecology of the Milnerton Lagoon. Honours project, Zoology Department, University of Cape Town, South Africa

Fish samples
Fish samples were collected at various points along the lagoon during the winter and summer of 2014. A 30 m × 2 m seine-net was used, with wings of 15 mm stretched mesh and the codend, and 5 m to either side of it, of 10 mm stretched mesh. Mean area swept by each sample was 180 m^2. Only one haul was taken at each site, because there was limited unfished space for replicates. We used these data to make comparisons with the fish species lists of Millard and Scott (1954) and Weil (unpublished data).

Prawn counts
The abundance of sandprawns *Callichirus kraussi* in the lagoon was estimated following the methods used by Clark (1998). The lagoon was subdivided into blocks 100 m long, extending from the mouth to Otto du Plessis Drive bridge (Figure 2). Sampling was conducted in late summer (April) 2014. Within each 100-m block, we randomly cast a 0.5 m × 0.5 m quadrat 20 times, while moving in a zigzag fashion across the width of the lagoon. Within each quadrat we counted the number of burrows made by sandprawns. Burrow numbers were then converted to prawn numbers by multiplying the number of burrow entrances by 1.05. This conversion factor was used for consistency with earlier surveys, and is based on the assumption that the ratio between the number of animals buried below the sediment and the number of burrow entrances is between 1.0 and 1.1 adult sandprawns per burrow entrance (Wynberg and Branch 1991; Clark 1998). However, subsequent studies have shown that the burrow:prawn ratio declines with decreases in salinity and water quality, because the animals burrow deeper to reach their preferred salinity (>16) (van Niekerk et al. 2010). Successive 100-m blocks were surveyed along the length of the lagoon, moving away from the mouth, until no burrow openings were recorded in three successive blocks. The total number of sandprawns per 100-m block was then calculated by measuring the width of the lagoon channel with a tape measure (from the upper limit of burrow occurrence on each bank, as these did not extend into the upper tidal level). Total prawns per 100-m section were then calculated as number per quadrat × 4 (i.e. multiplying up to 1 m^2) × width of the lagoon × 100.

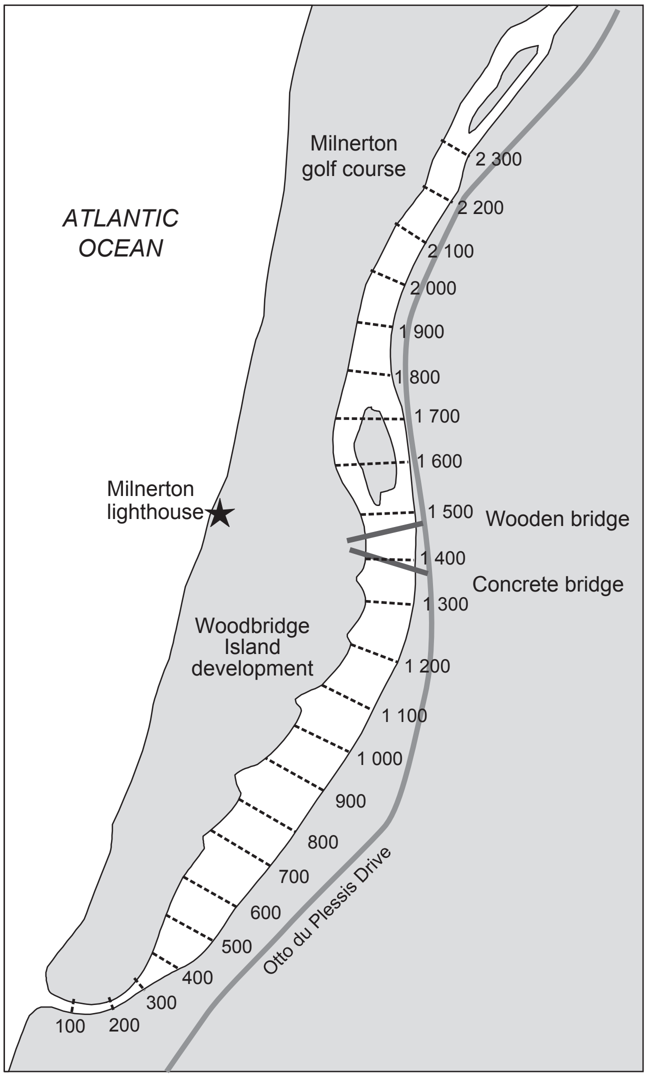

Figure 2: Map of lower part of Diep River estuary showing approximate positions of the 100-m-long blocks in which counts of sandprawn *Callichirus kraussi* were made (map taken from Clark 1998)

Results

Physical and chemical changes
The physical characteristics of the Diep River estuary have been modified considerably over the past 230 years. The first known map of the system is that of DM Barbier in 1786, as reproduced by Brown and Magoba (2009). Both that map and the more detailed version by Captain Sir Edward Belcher in 1846 (as reproduced in Figure 1b), show the Diep River having an extensive estuary system formed from a confluence of the present river with the Salt River Lagoon, which carries water from the Black and Liesbeek rivers. Belcher's map also showed the system entering the sea through two mouths – the present mouth, at that time situated slightly north of its current position, and another farther south, where it merged with the present Salt/Black rivers (Figure 1b). These two systems have now been separated artificially, with the Salt/Black River entering the sea though a canal and the Diep River through the former northern mouth, the position of which is now constrained by hardened banks and surrounding roads and buildings (Figure 1a).

The removal of riparian vegetation, combined with poor agricultural practices in the catchment, as well as sand entering the lower reaches from the sea, have resulted in the system becoming much shallower than in earlier times (Brown and Magoba 2009). Unsubstantiated reports state that ships (probably shallow barges, or ships' boats) were once able to navigate as far upstream as Vissershoek, approximately 13 km from the mouth (Grindley and Dudley 1988), a feat impossible in recent times. The substratum of most of Rietvlei and the lagoon is now muddy (Jackson et al. 2011). The first direct evidence of large-scale siltation in the lagoon dates back to 1905, when parts were deepened for rowing regattas (Brown and Magoba 2009). By 1920, a sandbar had developed, closing off the lagoon mouth from

Table 1: Timeline of major events that have occurred in and around the Diep River estuary

Date	Description
1899–1904	Wooden bridge built across Milnerton Lagoon
1904	Railway line built
1905	Sections of Milnerton Lagoon dredged
1928	Weir built across the mouth of Diep River
1929	Zonnekus: first house built on Woodbridge Island
1942	Weir demolished
1960	Potsdam Wastewater Treatment Works built. Effluent discharged into Diep River
1961	Otto du Plessis Drive bridge built across Diep River, separating Rietvlei and Milnerton Lagoon
1973	Rietvlei dredged to a depth of 9 m to provide fill for construction in Cape Town Harbour
1974	Lagoon again dredged for the Woodbridge Island development
1985	New concrete bridge built across lagoon, replacing old wooden bridge. Area below wooden bridge dredged to provide fill for Woodbridge Island housing development
1991/1992	Treated effluent discharged from Potsdam Wastewater Treatment Works channelled along the eastern boundary of Rietvlei
2004	Upgrade and expansion of Potsdam Wastewater Treatment Works

the sea. To counteract these changes, a weir was built across the mouth in 1928, in order to raise water levels upstream. The weir was subsequently destroyed by heavy floods in 1942 (Jackson et al. 2008; Brown and Magoba 2009).

Historical records suggest that the Diep River mouth remained almost permanently open to the sea (Grindley and Dudley 1988; Retief 2011). However, studies carried out in the 1950s and 1970s reported that the mouth closed periodically during the dry summer months (Millard and Scott 1954; Weil unpublished data; Jackson et al. 2011). In 1960, the Potsdam Wastewater Treatment Works was built and then in 1991/1992 a channel was constructed along the eastern boundary of the Rietvlei wetlands. Treated effluent from the treatment works is now discharged year-round from this channel into the lagoon. Since the excavation of this channel and the dredging activities described below, with the spoil being used to raise the ground level of what is now Woodbridge Island, the mouth has remained open virtually year-round (Jackson et al. 2008; Brown and Magoba 2009; Retief 2011), although in recent years it has closed in the summer (Brown and Magoba 2009). The discharge of sewage effluent into the system has various additional consequences, such as a massive decrease in salinity (detailed below) (Jackson et al. 2011).

After the founding of Milnerton Estates Limited in 1897 and the construction of a railway line in 1904, development and urbanisation began to accelerate (Grindley and Dudley 1988; Retief 2011). In 1904, a wooden bridge was built across the lagoon to enable the British to gain access to the coastline near the suburb of Milnerton to defend it against potential invaders (Brown and Magoba 2009) (Table 1). In 1985, when the Woodbridge Island housing development was established, a new concrete bridge was built next to the old wooden bridge (Brown and Magoba 2009). Two other bridges have also been built over the Diep River estuary; Blaauwberg Road bridge demarcates the upper limit of the estuary and Otto du Plessis Drive bridge, built in 1961, separates Rietvlei from the lagoon section of the estuary (Clark 1998; Brown and Magoba 2009). In the early 1970s the north-west part of Rietvlei was dredged to a depth of 9 m in order to provide fill for the construction of the new container berth in Cape Town Harbour, and in 1974 and again in 1985 the lagoon was dredged to provide sand for the Woodbridge Island development (Jackson et al. 2008) (Table 1).

Urbanisation of the areas surrounding Rietvlei and the Milnerton Lagoon has continued in conjunction with these developments. The land adjacent to the Diep River estuary is heavily developed with industrial and residential areas. The western bank is bordered by a golf course and the Woodbridge Island development, while the eastern bank is bordered by a sportsground, a fertiliser factory, an oil refinery and the wastewater treatment works (Clark 1998; Retief 2011). Stormwater from residential areas enters the estuary via a number of drains (Clark 1998; Retief 2011). The lower lagoon section of the estuary as it exists today is thus restricted to a confined channel, stabilised by road embankments and various bridges, but it is still a very important ecological system within the Western Cape province, representing one of the last functioning wetlands in the south-western Cape (Clark 1998). The estuary is also recognised as an important recreational site, supporting fishing, bait collecting and (at least until recent health concerns: Hutchings and Clark 2010; Jackson et al. 2011) a range of watersport activities (Clark 1998; Retief 2011).

Changes in salinity and flow regime

Since the establishment of the effluent channel along the east bank of Rietvlei in 1991/1992, the estuary has become increasingly freshwater-dominated, which is clearly shown by comparing the summer salinities at Otto du Plessis Drive bridge in 1954 (204), 1974 (80) and again in 2014 (0) (Figure 3). Historically, the mouth would close periodically during the summer, with the river effectively drying up. Combined with high evaporation rates, a reverse salinity gradient would then be established, with hypersaline conditions experienced at the vlei area of the estuary, near Otto du Plessis Drive bridge (Millard and Scott 1954; Weil unpublished data; Jackson et al. 2011). This phenomenon is clearly illustrated by the 1954 and 1974 data shown in Figure 3. More recently, the discharge of sewage effluent into the estuary has maintained relatively high flow levels,

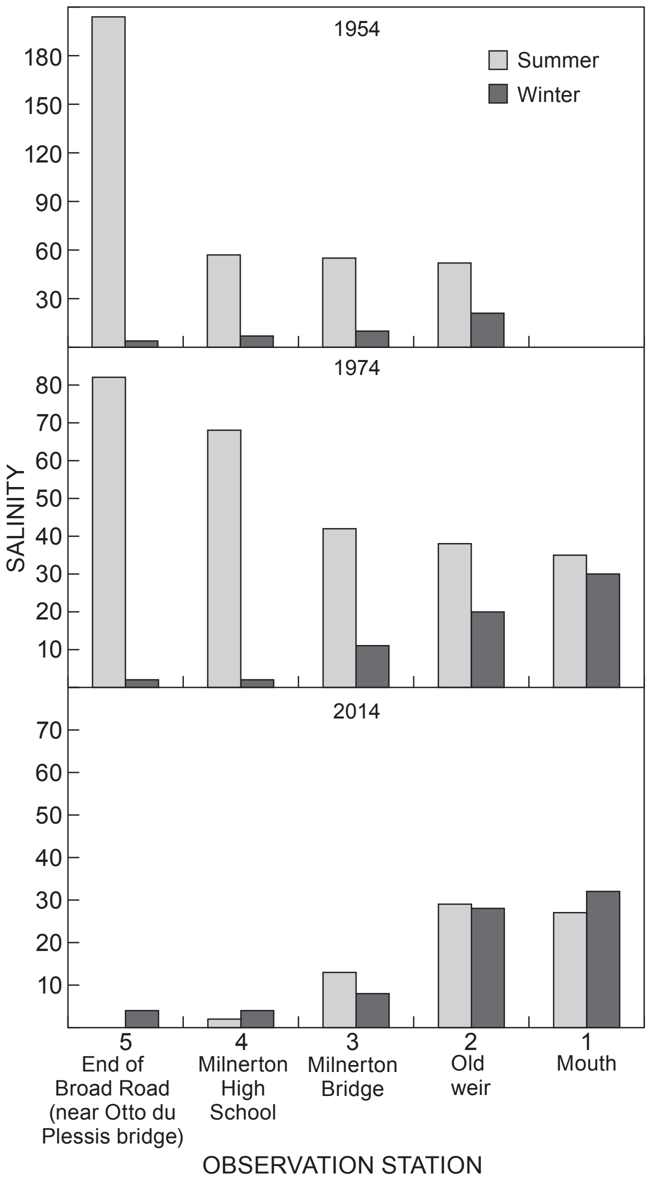

Figure 3: Salinity measurements taken at observation stations (names and numbers shown) in Milnerton Lagoon during winter and summer, in 1954 (Millard and Scott 1954), 1974 (Weil unpublished data) and 2014 (this study). The 1954 bar graph shows the average salinity taken in each season during the period 1948–1953

which have resulted in the mouth staying open almost permanently (Jackson et al. 2011). This has also resulted in a major drop in salinity levels, compared to those measured in 1954 and 1974 at the lagoon (Figure 3), with roughly the same salinity values being found at each of the stations for both summer and winter (Figure 3). In summer the flow consists predominantly of effluent from the treatment works (Jackson et al. 2011).

Faunistic changes

Infauna, epifauna

The species reported at each sample station in both summer and winter during the present study, as well as in earlier surveys by Millard and Scott (1954) and Weil (unpublished data), are compared in Table 2. There has been a dramatic decline in species richness of the invertebrate fauna since the earliest survey by Millard and Scott (1954), and there have been marked changes in species composition across the three surveys.

Millard and Scott (1954) recorded 47 species in the lower reaches of the system, but this declined by half to 23 reported by Weil (unpublished data), and remained at 23 in the present study. Between 1954 and 1974, the lists show 36 species as being ‘lost’, while 12 new ones were gained, giving a net loss of 24 species. From 1974 to 2014, 16 species were lost and 16 gained, giving a net gain of zero. Remarkably, only six of the 69 taxa listed overall were thus recorded by all three surveys.

The most obvious losses from the system have been the complete disappearance of all 10 bivalve, gastropod and bryozoan species reported in 1954 from both subsequent surveys (Table 2). Three of the four isopod species recorded were also not reported again after 1954, and four of the nine polychaete worms. The gains were largely among the insects (which are primarily freshwater species), which increased from about one-third of all species in both 1954 and 1974 to more than half in 2014. Two alien invertebrates not previously reported from the system were also recorded in 2014.

There have also been numerous more-subtle changes in community composition. Care needs to be taken when interpreting these, however, as individual species may have been missed in a given survey due to differences in sampling intensity and technique, identification skills, etc. For example, the absence of nematodes from our survey is unlikely to indicate their absence from the system, but is probably because we sieved all samples though a 1-mm mesh. Similarly, only Weil (unpublished data) reported Cladocera, Ostracoda and Copepoda, similarly ubiquitous groups that were almost certainly always present, but were just too small to be noted by the other surveys. Other changes, however, are certainly real. The disappearance of the two bivalve, two bryozoan, and all six gastropod species reported in 1954 could not be the result of sampling or identification issues because they are large, or easy to recognise, species or taxonomic groups. One of the most abundant species recorded in 2014 was the polychaete *Ceratonereis erythraeensis*, which had not previously been found in the system. We are confident that this species would not have been missed, particularly by Millard and Scott, because their polychaete specimens were identified by a leading polychaete taxonomist, Prof. JD Day.

Seasonal differences are evident in all three datasets. Species richness was considerably higher during summer than winter surveys. This was largely, but not entirely, due to the summer abundance of taxa such as insects. There were no marked trends in species richness along the length of the lagoon, although of course many individual species are restricted to either the mouth or more-upstream sections of the system (as detailed in Table 2).

Fish survey

There have been clear declines in fish species richness since the original study carried out by Millard and Scott

Table 2: Complete species list of all infauna and epifauna recorded at Milnerton Lagoon in the summer and winter periods of 2014 (this study), 1974 (Weil unpublished data) and 1954 (Millard and Scott 1954). The total number of species recorded at each station (refer to Figure 1) in winter and summer and the total number of species collected in each study are also presented (√ = present, AB = abundant). Cryptogenic and alien invasive species are highlighted in bold

Taxonomic group		2014										1974										1954									
		Summer					Winter					Summer					Winter					Summer					Winter				
		1	2	3	4	5	1	2	3	4	5	1	2	3	4	5	1	2	3	4	5	1	2	3	4	5	1	2	3	4	5
Platyhelminthes													√	√	√	√										√					√
Nematoda													√	√	√	√		√					√	√	√	√					
Bryozoa	*Lophopodella capensis*																									√					√
	Membranipora sp.																						√								
Mollusca	*Choromytilus meridionalis*																					√					√				
	Ostrea sp.																					√									
	Helcion pruinosa																					√									
	Argobuccinum argus																										√	√	√		
	Assiminea sp.																										√				
	Tomichia ventricosa																									√					
	Bullia digitalis																										√				
	Aplysiopsis sinusmensalis																						√								
	Bulinus tropicus																									√					
Cnidaria	*Anthothoe stimpsoni*												√	√																	
Nemertea	*Lineus olivaceus*												√	√																	
Polychaeta	***Capitella capitata***		√	√	√			√	√				AB	AB	AB	√		√	√				√	√				√	√		
	Ceratonereis erythraeensis			AB	√				AB	AB	√																				
	Ficopomatus enigmaticus			√	√								AB	AB	√	√	AB	AB	√	√	√			√	√				√	√	
	Scolepsis squamata	√											√									√					√				
	Prionospio sexoculata																						√	√	√			√	√	√	
	Lumbrinereis tetraura																					√									
	Nephthys capensis																						√					√			
	Oridia parvula																							√					√		
Oligochaeta								√			√													√					√		
Crustacea	Branchiopoda												√		√					√	√										
	Ostracoda												√		√					√	√										
	Copepoda												√		√					√	√										
	Balanus algicola																					√									
	Amphibalanus sp.						√																								
	Eurydice longicornis	√	√																				√	√	√			√	√	√	
	Exosphaeroma hylecoetes																								√						
	Exosphaeroma varicolor																					√	√								
	Paridotea ungulata																						√								
	Orchestia gammarella			√	√	√			√		√																				
	Melita zeylanica			√	√								AB	AB	AB			√					√	√	√						
	Afrochiltonia capensis													AB	AB	√			√				√	√	√	√		√	√	√	
	Callichirus kraussi						√						AB	√	√	√	√	√	√				AB	√				AB	√		
	Hymenosoma orbiculare		√		AB								√	√	√								√	√	√			√	√	√	
	Palaemon pacificus												AB	√	√								√	√	√			√	√	√	
	Cyclograpsus punctatus																						√	√				√	√		

Table 2: (cont.)

Taxonomic group		2014										1974										1954									
		Summer					Winter					Summer					Winter					Summer					Winter				
		1	2	3	4	5	1	2	3	4	5	1	2	3	4	5	1	2	3	4	5	1	2	3	4	5	1	2	3	4	5
Insecta	Leptophlebiidae					√																									
	Eristalis sp.														√																
	Cloeon lacunosum																							√	√	√			√		
	Ishnura senegalensis				AB	√																			√						
	Euborellia annulipes			√					√																						
	Belostomatidae					√																									
	Micronecta scutellaris			√	√																					√					
	Sigara meridionalis					√	AB																		√	√					
	Notonectidae						√																								
	Enithares sp.						√																								
	Notonecta lactitans																									√					
	Plea piccanina																													√	
	Sigara sjostedti														√	√				√	√										
	Hydrophilidae					√																									
	Berosus spretus														√	√				√	√				√	√			√	√	
	Helophorus aethiops																													√	
	Helochares dilutus																								√						
	Hydrochus sp.																								√	√					
	Ochthebius extremus																									√				√	
	Rhantus peringueyi																								√						
	Bledius sp.		√				√								√	√															
	Pogonus lamprus														√	√															
	Berosus sp. (larvae)														√																
	Culicidae																			AB	AB										
	Fucellia capensis	√																													
	Tendipes sp.												AB	AB	AB	√	√	√													
	Chironomus sp.				√	AB	√																√	√	√	√		√	√		
	Ephydridae																							√	√	√			√	√	√
	Collembola																							√	√	√			√	√	√
Total number of species at each station		3	4	7	9	7	7	2	4	1	3	0	15	12	19	11	3	6	4	7	7	7	16	17	19	16	5	11	17	12	4
Total number of species in each season		18					12					22					13					43					27				
Total number of species in each study		23										23										47									

(1954), who recorded 12 species, compared to nine recorded by Weil (unpublished data) and only five in the present study. Millard and Scott (1954) reported the thin mullet, or harder, *Liza ramada* (now classified as *Liza richardsonii)*, white steenbras and white stumpnose as the most common fish found within the lagoon, whereas Weil (unpublished data) found that the estuarine round herring *Gilchristella aestuaria* had the highest abundance.

Of the five fish species recorded in 2014 (Table 3), by far the most abundant species was *L. richardsonii*, which is an opportunistic species that moves between optimum conditions in the estuarine and marine environments (Whitfield 1994; Lamberth et al. 2010). Other abundant species were *G. aestuaria* and the mosquito fish *Gambusia affinis* – an extremely invasive species found in relatively high abundance in upstream stations and a new record for the system. *Tilapia sparrmanii*, which, although endemic to southern Africa, is a translocated species from the Orange River catchment and not native to the study region, was also recorded for the first time (Table 3). Thus, only three of the original native species were still present in 2014.

Sandprawn density

Clark (1998) reported that the density of sandprawns increased progressively upstream from the mouth, peaking at the block between 1 100–1 200 m with an abundance of just over 8×10^6 prawns (Figure 4). Prawns extended 2.2 km from the mouth of the lagoon and the estimated total standing stock was approximately 40 million individuals (Clark 1998).

Our study showed the sandprawn population to have declined substantially and to have moved its centre of gravity downstream to the 800–900 m block, where just over 4×10^6 individuals were recorded. Also, sandprawns were only recorded up to 1.6 km from the mouth (Figure 4). The estimated total standing stock was 12.3 million individuals. Additionally, 93% of the total sandprawn population is now vulnerable to bait-collecting activities, because all bait collecting is confined to the region between the mouth and 1 000 m upstream, as noted by Clark (1998) and BM Clark (Anchor Environmental Consultants, pers. comm.).

Alien and invasive species

Three new introduced species and one new translocated species were recorded in the Diep River estuary in the 2014 surveys (marked in bold in Tables 2 and 3). These were the semiterrestrial amphipod *Orchestia gammarella* and earwig *Euborellia annulipes*, the introduced fish *G. affinis* and the translocated *T. sparrmanii.*

Three other known or suspected introduced species had been recorded previously in the system and their presence was confirmed. These were the abundant reef-forming polychaete *Ficopomatus enigmaticus*, which has now formed particularly large reefs under the bridges to Woodbridge Island, and two cryptogenic (suspected as introduced) species, the amphipod *Melita zeylanica* and the polychaete *Capitella capitella*. The latter species has recently been shown to represent a species complex, so this identification remains dubious (see Mead et al. 2011).

Discussion

Physical and hydrological changes

The Diep River estuary has been subjected to a long history of exploitation and perturbation, and is no longer capable of functioning in a completely natural fashion (Jackson et al. 2011).

Dredging

Dredging of sections of Milnerton Lagoon has occurred on separate occasions in 1905, 1974 and 1985, while in 1973, a very large section of Rietvlei was dredged to a depth of 9 m (Table 1). Dredging of shallow-water estuaries has a profound effect on benthic infauna and microalgae, and ultimately on fish species within the system (Ray 2005). Firstly, dredging involves the direct, physical removal of sediment, which can have major implications for the recruitment and establishment of benthic infauna (Ray 2005). Dernie et al. (2003), for example, showed that the removal of only 20 cm of sediment from a Welsh sandflat required 208 days for the infaunal community to re-establish. Additionally, increased depth created by dredging results in a reduction in ambient light reaching the bottom in the dredged area (Ray 2005). This impacts negatively on benthic microalgae that are critical components of estuarine primary production, supporting many benthic invertebrates and fish species (Ray 2005).

Bridges and other hard structures

Bridges create obstructions to the flow of watercourses, causing the riverbed over which the bridge is located to become eroded (Kingston 2006). Also, as hard structures within the dominantly soft environments of estuaries, bridges provide habitat for different species. This is particularly evident on the two bridges to Woodbridge Island, the pilings of which are focal points for colonisation by huge reefs of the invasive *F. enigmaticus*.

The construction of concrete walls along the banks of the Diep River estuary has also converted large areas of either soft sand and sediment or reed bed into steep, hardened surfaces. This has also resulted in extensive loss of intertidal habitat, especially in the lower reaches. The fauna previously occupying these lost habitats is largely undocumented, because it appears that the banks had already been hardened by the time of Millard and Scott's (1954) survey (see their Figure 8.2). McQuaid and Griffiths (2014) showed that hard surfaces greatly facilitated the recruitment and/or expansion of exotic and invasive species within another estuary nearby (Zandvlei Estuary), where large reefs of the invasive *F. enigmaticus* have grown to such a size that they dominate overall invertebrate biomass and even hamper navigation under bridges.

Water quality

With the increase in urbanisation around the Diep River estuary, significant changes in the volume and quality of water flowing into the system have occurred (Jackson et al. 2011). Several effluent discharges are released into Rietvlei (Taljaard et al. 1992). The most important are discharge from the wastewater treatment works, stormwater runoff from an oil refinery, and the large residential stormwater

Table 3: List of all the fish species recorded in the Milnerton Lagoon in the summer and winter periods of 2014 (this study), 1974 (Weil unpublished data) and 1954 (Millard and Scott 1954). For sampling stations see Figure 1. Species are grouped into their estuarine-dependence categories (adapted from Whitfield 1994). Counts were recorded in 2014 and 1974, but only presence (√) was recorded in 1954. Note: *Liza richardsonii* was previously reported as *Liza ramada*; introduced or translocated species are highlighted in bold

Species (common names) within estuarine-dependence categories			2014										1974										1954									
			Summer					Winter					Summer					Winter					Summer					Winter				
			1	2	3	4	5	1	2	3	4	5	1	2	3	4	5	1	2	3	4	5	1	2	3	4	5	1	2	3	4	5
Estuarine residents that breed mostly in estuaries, never in the marine environment																																
Clupeidae	*Gilchristella aestuaria*	(Estuarine round herring)	1	80					108	2	8	1		20									√			√		√			√	
Estuarine residents with marine and estuarine breeding populations																																
Atherinidae	*Atherina breviceps*	(Cape silverside)																			13	13	√			√		√				
Clinidae	*Clinus superciliosus*	(Super klipvis)																					√	√	√							
Gobiidae	*Caffrogobius nudiceps*	(Barehead goby)	13						2	2		5		4					1				√			√		√				
Gobiidae	*Psammogobius kysnaensis*	(Knysna sandgoby)												12					2				√			√		√			√	
Juveniles entirely dependent on estuaries as nursery areas																																
Mugilidae	*Mugil cephalus*	(Flathead mullet)												2									√			√		√			√	
Sparidae	*Lithognathus lithognathus*	(White steenbras)												6					1				√			√		√			√	
Sparidae	*Rhabdosargus tricuspidens*	(Cape stumpnose)																								√		√				
Juveniles occurring mainly in estuaries but also found at sea																																
Soleidae	*Heteromycteris capensis*	(Cape sole)																					√	√	√			√	√	√		
Juveniles occurring in estuaries but usually more abundant in the sea																																
Sparidae	*Rhabdosargus globiceps*	(White stumpnose)												4									√			√		√				
Pomatomidae	*Pomatomus saltator*	(Elf)																					√			√						
Mugilidae	*Liza richardsonii*	(Harder/mullet)	190	290	610	125			2 610	2 100	11	96		15					3				√			√		√			√	
Freshwater species, sometimes found in estuaries																																
Galaxiidae	*Galaxias punctifer*	(Dwarf galaxias)																			8	8										
Introduced or translocated freshwater species, sometimes found in estuaries																																
Poeciliidae	***Gambusia affinis***	(Mosquito fish)				30	18																									
Cichlidae	***Tilapia sparrmanii***	(Banded tilapia)			1							1																				
Total number of species			5										9										12									

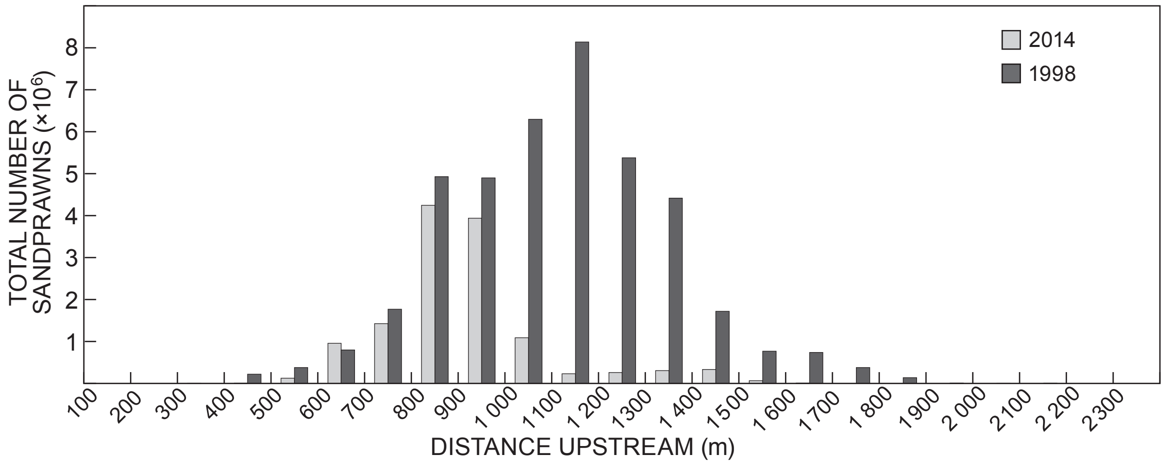

Figure 4: Estimated number of sandprawns *Callichirus kraussi* per 100-m block in the Diep River estuary (data for 1998 from Clark 1998)

discharge in the north-western section of Rietvlei (Taljaard et al. 1992; Jackson et al. 2008; Brown and Magoba 2009; Retief 2011). Taljaard et al. (1992) noted bacterial pollution in the lagoon as being minor, and trace-metal concentrations were found to pose no toxicological threat to the biota within the system and within maximum limits for water quality criteria (Taljaard et al. 1992). Since that study, the treatment works has been upgraded and expanded, and, in addition, a new sewage channel along the eastern boundary of the Rietvlei wetlands was constructed in 1991/1992 (Brown and Magoba 2009; Jackson et al. 2008; Retief 2011). Various water quality studies have since been conducted in the estuary. Paulse et al. (2009) investigated microbial contamination at three sites along the Diep River and found that, at all three sites, the microbial counts exceeded stipulated limits in terms of water quality guidelines. Jackson et al. (2009), Hutchings and Clark (2010) and Ayeni et al. (2010) determined the extent of metal contamination along the lower Diep River. All three studies found the area to be polluted with a variety of metals, with concentrations of some exceeding established guidelines. Such metal contamination can have a detrimental effect on plants, microorganisms, human health and overall ecosystem health (Ayeni et al. 2010). Concentrations of trace metals (arsenic, cadmium, mercury and lead) in fish and invertebrates from the Rietvlei–Diep system also showed a similar pattern to the sediments and exceeded most South African and international limits for foodstuffs (Hutchings and Clark 2010). Jackson et al. (2011) thus suggest that the water in Milnerton Lagoon has been unsuitable for recreational contact since at least 2001.

Changes in salinity

Historically, the mouth of the system closed periodically during the summer, and high evaporation rates led to hypersaline conditions and a reverse salinity gradient (Retief 2011). Millard and Scott (1954) described Rietvlei 'as a series of extensive saline seasonal pans' whereby in late summer and autumn, Rietvlei was completely dried up. Millard and Scott (1954) and Weil (unpublished data) both stressed the importance of this seasonal salinity gradient in structuring the fauna in the upper (Rietvlei) reaches of the Diep River estuary and the head of Milnerton Lagoon. However, since the establishment of the channel along the east bank of Rietvlei in 1991/1992, the estuary has become increasingly freshwater-dominated (Jackson et al. 2008; Brown and Magoba 2009; Retief 2011). This has had major implications for both the horizontal and vertical distribution of the fauna within the system. Weaker salinity gradients along the estuary may also have negatively impacted larval and juvenile fish recruitment, but this effect will have been reversed almost entirely by increased freshwater flow and olfactory cues entering the sea (Whitfield 2005).

Faunistic changes

Invertebrates

Our results show dramatic differences in diversity and species structure of the invertebrate communities reported in the 1954, 1974 and 2014 surveys. Care needs to be taken when comparing such data though, due to differences in sampling effort, sampling method and taxonomic expertise available to the authors. The authors of both earlier studies were not explicit about the exact numbers and sizes of samples taken, making it difficult to replicate their surveys exactly. Those authors probably also had different levels of taxonomic expertise; Millard and Scott (1954) utilised the expertise of a team of specialists, whereas JG Weil was a student and so probably identified specimens himself. We are confident, however, that differences in effort or technique cannot account for the more marked changes observed, such as the disappearance of all eight of the mollusc species reported in 1954. The overall trend thus appears to be a dramatic decline in species, with increasing dominance of freshwater forms, particularly those found in the water column, such as the Hemiptera.

All three studies reported a noticeable blackening of the subsurface sediment over the upper reaches of the estuary, accompanied by the smell of hydrogen sulphide and almost complete absence of burrowing organisms. In fact, most of the invertebrates collected during the present study were found living in sediments along the banks of the lagoon, or among the aquatic vegetation, with the benthic infauna being depauperate, especially above the Milnerton Bridge. Weil (unpublished data) found similar results and suggested that oxygen concentration may be the limiting factor controlling faunal distribution along the length of the lagoon and from the centre to the lagoon banks.

Fish

There has been a progressive decline in fish diversity within the system, from 12 species reported in 1954 to nine in 1974 and only five, two of which are newly reported introductions, in 2014.

Millard and Scott (1954) reported *L. richardsonii*, *R. holubi* and *L. lithognathus* as the most commonly found fish within the lagoon, whereas Weil (unpublished data) found that *G. aestuaria* had the highest abundance. Currently, *L. richardsonii*, *G. aestuaria*, and the introduced *G. affinis* are the most abundant fish species within the lagoon. Linefish (i.e. angling) species, once abundant, were not recorded in our samples. Retief (2011) and Jackson et al. (2008) suggest that the low water quality, combined with high ammonia concentrations experienced in the lagoon through malfunctions in the wastewater treatment works, have caused important benthic organisms, such as *C. kraussi*, and other invertebrate species to undergo massive declines, reducing food resources available for fish and contributing to the major decline of important linefish species within the estuary. Even in the lower reaches where these invertebrates still occur, elevated freshwater flow would have led to the establishment of deeper burrows and the unavailability of the invertebrates as prey. Although many estuarine-associated species are adapted to hypoxia, it is unlikely that any benthic fish would have persisted where anoxic conditions occur (Lamberth et al. 2010). Anoxia aside, an overall crash in the availability of invertebrate burrows would have resulted in a demise of the populations of Gobiidae and other commensal or mutualistic fish that find refuge in association with them (Karplus and Thompson 2011).

Sandprawns

Both the distribution and the estimated standing stock of sandprawns in Milnerton Lagoon have contracted substantially since the study of Clark (1998). Clark (1998) concluded that the level of prawn harvesting at that time was sustainable; however, he recommended that bait collecting be confined to the region between the mouth and 1 000 m upstream (Clark 1998), to ensure that only half of the prawn population would be affected by exploitation. Clearly, this is no longer the case, as the population is now both much smaller and largely confined to the exploited area.

Paradoxically, the current sandprawn population may in fact represent a partial recovery, since Retief (2003) found the population to be completely absent, perhaps as a result of chemical pollution from the wastewater treatment works. Further surveys are recommended to determine whether the current population is expanding or declining.

As salinity and water quality decline, sandprawns burrow deeper to reach their preferred salinity (>16) at which they can breed (van Niekerk et al. 2010). Hence it becomes energetically expensive to maintain an opening to the surface once the prawns are deep in the sediment and so they 'share' burrow entrances, which results in a reported decline in burrow:prawn ratios. Similarly, bait collectors, fish and wading birds find it more difficult to catch prawns and other burrowing invertebrates during high freshwater flow (van Niekerk et al. 2010). This behaviour may explain why the prawn population appears to recover quickly after toxic events in the estuary, possibly without the need for new recruitment from the sea.

Sandprawns are 'keystone species' and important ecosystem engineers in marine soft sediments, and their burrowing activities facilitate the colonisation of the sediment by a wide variety of other species (Pillay and Branch 2011). These species, together with the prawns themselves, constitute important food sources for many bird and fish species (Clark 1998; Pillay and Branch 2011). Specifically, their burrowing activities turn over very large quantities of sediment, which has been found to reduce bacterial colonisation on the sediment surface and to oxygenate sediments to depths that may exceed 1 m (Clark 1998; Pillay and Branch 2011). The sandprawn thus clearly plays a very important role within the Diep River estuary and its decline has probably had multiple effects on the abundance and types of other infauna and epifauna present.

Alien invasive species

The most obvious and important invasive invertebrate in the estuary is the reef-building polychaete *F. enigmaticus*, which is confined to hard substrata such as rocks, reeds, submerged wood, bottles and other debris. It is a fast-growing, highly invasive species that originates from Australia and acts as an ecosystem engineer, capable of modifying its habitat physically, chemically and biologically (Schwindt et al. 2001; Heiman et al. 2008; McQuaid and Griffiths 2014). All previous surveys in the estuary reported the presence of this species and Weil (unpublished data) noted it as highly abundant at Stations 1 and 2, with its occurrence extending upstream to Station 5. The development of concrete walls along the banks of the estuary and the construction of bridges across the lagoon undoubtedly facilitated the recruitment and expansion of this species within the estuary, as shown by the recent work of McQuaid and Griffiths (2014) and McQuaid (2013) in the nearby Zandvlei Estuary. To date, no studies have quantified the spread of this highly invasive species within the Diep River estuary, but anecdotal evidence suggests that the colonies are larger than in the past.

The impacts that this species has on native communities and the surrounding environment are complex and can be both positive and negative to other components of the fauna (Schwindt et al. 2001; Heiman et al. 2008; McQuaid and Griffiths 2014). The reefs can be a nuisance to human users, because they foul boats, wharfs and bridges, impede boat traffic and their sharp surfaces can injure recreational users. *Ficopomatus enigmaticus* reefs also provide a new, structurally complex, heterogeneous habitat in muddy ecosystems (Schwindt et al. 2001; McQuaid and Griffiths 2014) and these enhance invertebrate recruitment and survival by providing shelter, food and substratum for both native and non-native species (Schwindt et al. 2001; Heiman et al. 2008; Heiman and Micheli 2010; McQuaid and Griffiths 2014). A study quantifying the association of other invertebrate species with the *F. enigmaticus* reefs would be required in order to define its impacts within this system. Given the small extent and limited habitat available to this species in Milnerton Lagoon, elimination would seem possible and would require mechanical removal of the reefs that are associated with the bridges and the removal of other hard, submerged foreign substrata (rocks, bottles, logs and other debris) from the lagoon.

The other introduced invertebrates reported are the earwig *E. annulipes* and the beach hopper *O. gammarella*, both of which were associated with drift weed deposited along the high water mark. Both are new records for the area, but neither have known ecological impacts that require management intervention.

Although only two introduced fish, *G. affinis* and *T. sparrmanii*, were recorded in the lower section of the estuary during our study, it is noteworthy that the equally invasive sharptooth catfish *Clarias gariepinus*, carp *Cyprinus carpio* and Mozambique tilapia *Oreochromis mossambicus* also occur in the upper reaches of the system (Brown and Magoba 2009).

The alien invasive *G. affinis* is one of the most widely introduced species on the planet, and is recognised as being among the most damaging invasive fish species worldwide (Howell et al. 2013). It was deliberately introduced into South Africa in 1936 for purposes of mosquito control and as fodder fish for introduced bass *Micropterus* spp. and is now well established in freshwater ecosystems across southern Africa (Howell et al. 2013). This fish is likely to have a variety of deleterious impacts, including competition with and predation on the eggs of native fish and frogs, alterations of invertebrate and vertebrate communities through habitat alteration, and introduction of fish diseases (Howell et al. 2013). Little work has been done on its impacts in South Africa, despite its dominance in many smaller waterbodies (Olds et al. 2011).

Tilapia sparrmanii is native to the Orange River Basin and to much of southern Africa to the north, but it has been translocated to many waterbodies outside its original range. It can compete with indigenous fishes and has been implicated in the decline of threatened native fish species. It also acts as a vector for at least 14 species of fish parasites (Picker and Griffiths 2011). Once introduced, both *G. affinis* and *T. sparrmanii* are difficult to eliminate.

Birds

It was not an aim of this study to survey the bird fauna, but as there are some data available from other sources, these are discussed here. A detailed analysis of long-term trends in the species composition and abundance of waterbirds in the Rietvlei section of the system over the period 1950–1997 is provided by Kaletja-Summers et al. (2001). The number of waterbirds in that section has increased some fourfold over time, with increases recorded in both the Palaearctic wader and resident components of the fauna. The reasons for this are complex, but include changes in the water regime (temporary waterbodies now becoming permanent), increased cover by emergent reeds – resulting partly from increased water and organic nutrient levels (Kaletja-Summers et al. 2001) – and decreased disturbance following the declaration of the area as a nature reserve.

Less information is available about the lagoon section that forms the focus of this study. A fairly anecdotal account of the composition of the bird fauna in the 1950s is provided by Scott (1954) and this can be compared with recent counts on the website http://cwac.adu.org.za (Animal Demography Unit Coordinated Waterbird Counts [CWAC] project). Although these data are difficult to compare statistically, given differences in frequency and intensity of observation, they do suggest an increase in bird numbers, as has been observed in Rietvlei. For example, Scott (1954) gave the following maximum bird densities for abundant resident species (each followed by recent maximum CWAC counts in brackets) – kelp gull: 'up to 50' (116); Hartlaub's gull: 'up to 70' (136); common tern: '40–50' (250); cormorants: '<12' (82, for summed species); Cape shoveller: 'a few' (20). Other species absent from Scott's (1954) account are also now common, most notably red-knobbed coot and African darter, which are predominantly freshwater species. All these data thus support the conclusion that the estuary now supports higher numbers and more species of birds than in the past. The reasons for this are likely as given above for Rietvlei (i.e. more fresh water, more nutrients, better protection).

Conclusion

In the last 200 years, the Diep River estuary has experienced extensive transformation in its physical structure, chemical composition and biota. Activities in the Diep River catchment, together with the intensive urban development in the areas adjacent to the estuary, have resulted in massive changes to biodiversity, altered flow and salinity regimes, deterioration in water quality, and an increase in non-indigenous species introductions. Earlier studies on this system report the Diep River estuary as being relatively unpolluted in terms of nutrients and trace metals, with a high abundance and diversity of fish and invertebrate species. Current evidence suggests the condition of the estuary has deteriorated strikingly since then, with high microbial levels being recorded in the lagoon and a marked decrease in the number and abundance of invertebrate species. This decrease is most clearly exemplified by the well-documented change in *C. kraussi* abundance and distribution in the lagoon, over a relatively short time-period of only 16 years. These changes and disturbances have been accompanied by the expansion and new introduction of exotic and invasive species, most significantly the highly invasive *G. affinis* and the reef-building *F. enigmaticus*. The only group of organisms that appear to have diversified and increased in density are the birds.

Decisive action addressing these problems needs to be implemented if this valuable system is to be saved from further stress and degradation. In particular, urgent consideration needs to be given to reducing some of the pollution sources. Additionally, a salinity regime that promotes the reestablishment of the keystone *C. kraussi* is recommended. Regular monitoring of the infaunal and epifaunal populations for this system should also be implemented, in order to obtain more accurate data on the ongoing faunistic changes occurring within this highly dynamic environment. Mechanical elimination of the exotic *F. enigmaticus* could also be considered.

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References

Ayeni OO, Ndakidemi PA, Snyman RG, Odendaal JP. 2010. Metal contamination of soils collected from four different sites along the lower Diep River, Cape Town, South Africa. *International Journal of the Physical Sciences* 5: 2045–2051.

Blaber SJM, Cyrus DP, Albaret JJ, Chong VC, Day JW, Elliott M et al. 2000. Effects of fishing on the structure and functioning of estuarine and nearshore ecosystems. *ICES Journal of Marine Science* 57: 590–602.

Botes WAM, Le Roux M. 2004. Milnerton Estuary: an assessment of the effects from reduced flows from the Potsdam WWTW. Consultancy report prepared for the City of Cape Town. WAM Technology, Stellenbosch.

Brown C, Magoba R (eds). 2009. *Rivers and wetlands of Cape Town*. WRC Report No. TT 376/08. Pretoria: Water Research Commission.

Clark BM. 1998. Human utilisation of the Diep River Estuary with an assessment of the intensity and impact of bait collecting on the ecology of the estuary. Report prepared for the Blaauwberg Municipality, WWF-SA and Friends of Rietvlei. Anchor Environmental Consultants, Cape Town.

Dernie KM, Kaiser MJ, Richardson EA, Warwick RM. 2003. Recovery of soft sediment communities and habitats following physical disturbance. *Journal of Experimental Marine Biology and Ecology* 285–286: 415–434.

Du Toit JT. 1982. Ecological aspects of the Diep River between Milnerton estuary and Rietvlei. MSc thesis, University of Cape Town, South Africa.

Grindley JR, Dudley S[FJ]. 1988. *Estuaries of the Cape Part II: synopses of available information on individual systems. Report No. 28: Rietvlei (CW 24) and Diep (CW 25).* CSIR Research Report 427. Stellenbosch: Estuarine and Coastal Research Unit, Council for Scientific and Industrial Research.

Gutiérrez JL, Jones CG, Byers JE, Arkema KK, Berkenbusch K, Commito JA et al. 2012. Physical ecosystem engineers and the functioning of estuaries and coasts. In: Wolanski E, McLusky DS (eds), *Treatise on estuarine and coastal science*. Waltham: Academic Press. pp 53–81.

Heiman KW, Micheli F. 2010. Non-native ecosystem engineer alters estuarine communities. *Integrative and Comparative Biology* 50: 226–236.

Heiman KW, Vidargasi N, Micheli F. 2008. Non-native habitat as home for non-native species: comparison of communities associated with invasive tubeworm and native oyster reefs. *Aquatic Biology* 2: 47–56.

Howell DH, Woodford DH, Weyl OLF, Froneman W. 2013. Population dynamics of the invasive fish, *Gambusia affinis*, in irrigation impoundments in the Sundays River Valley, Eastern Cape, South Africa. *Water SA* 39: 485–490.

Hutchings K, Clark BM. 2010. Assessment of the effects of heavy metal accumulation from land based sources on the Diep estuary. Unpublished report prepared for Department of Environmental Affairs and Development Planning, Western Cape Government. Anchor Environmental Consultants, Cape Town.

Jackson L, Conrad J, Carstens M. 2011. Estuary management plan for the Diep estuary. Unpublished report prepared for the CAPE Estuaries Programme, January 2011. Coastal and Environmental Consulting, Cape Town. Available at https://www.capetown.gov.za/en/EnvironmentalResourceManagement/Documents/TableBay/Estuary_Mngt_Plan_for_Diep_Estuary_2011-01.pdf [accessed 30 October 2015].

Jackson L, Conrad J, De Meyer M, Cullinan C, Mahomedy L, Dehrmann A. 2008. Situation assessment for the Diep estuary. Unpublished report prepared for the CAPE Estuaries Programme, October 2008. Peak Practice, Hermanus.

Kaletja-Summers B, Allan DG, Longrigg TD. 2001. Long-term trends, seasonal abundance and energy consumption of waterbirds at Rietvlei, Western Cape, South Africa, 1950–1997. *Ostrich* 72: 63–79.

Karplus I, Thompson AR. 2011. The partnership between gobiid fishes and burrowing alpheid shrimps. In: Patzner RA, Van Tassell JL, Kovacic M, Kapoor BG (eds), *The biology of gobies*. Enfield, New Hampshire: Science Publishers, CRC Press. pp 559–607.

Kingston S. 2006. Backwater effects caused by bridges. Thesis, Universidad Politecnica de Cartagena, Spain.

Lamberth SJ, Branch GM, Clark BM. 2010. Estuarine refugia and fish responses to a large anoxic, hydrogen sulphide, “black tide” event in the adjacent marine environment. *Estuarine, Coastal and Shelf Science* 86: 203–215.

Lewis EL. 1980. The practical salinity scale 1978 and its antecedents. *IEEE Journal of Oceanic Engineering* OE-5: 3–8.

McQuaid K[A]. 2013. Long-term change in the invertebrates of Zandvlei Estuary, with focus on the invasive reef worm *Ficopomatus enigmaticus*. MSc thesis, University of Cape Town, South Africa.

McQuaid KA, Griffiths CL. 2014. Alien reef-building polychaete drives long-term changes in invertebrate biomass and diversity in a small, urban estuary. *Estuarine, Coastal and Shelf Science* 138: 101–106.

Mead A, Carlton JT, Griffiths CL, Rius M. 2011. Introduced and cryptogenic marine and estuarine species of South Africa. *Journal of Natural History* 45: 2463–2524.

Millard NAH, Scott KM. 1954. The ecology of South African estuaries. Part VI. Milnerton estuary and the Diep River, Cape. *Transactions of the Royal Society of South Africa* 34: 279–324.

Olds AA, Smith MKS, Weyl OLF, Russell IA. 2011. Alien invasive freshwater fishes in the Wilderness Lakes system, a wetland of international importance, Western Cape, South Africa. *African Zool*ogy 46: 179–184.

Paulse AA, Jackson VA, Khan W. 2009. Comparison of microbial contamination at various sites along the Plankenburg- and Diep Rivers, Western Cape, South Africa. *Water SA* 35: 469–478.

Picker M, Griffiths C. 2011. *Alien and invasive animals – a South African perspective.* Cape Town: Struik Nature.

Pillay D, Branch GM. 2011. Bioengineering effects of burrowing thalassinidean shrimps on marine soft-bottom ecosystems. *Oceanography and Marine Biology: an Annual Review* 49: 137–192.

Ray LG. 2005. Ecological functions of shallow, unvegetated estuarine habitats and potential dredging impacts (with emphasis on Chesapeake Bay). Report No. ERDC TN-WRAP-05-3, Wetlands Regulatory Assistance Program. Available at http://el.erdc.usace.army.mil/elpubs/pdf/tnwrap05-3.pdf [accessed 2 November 2015].

Retief JJ. 2003. A sand-prawn survey in the Diep River estuary, Milnerton. Unpublished report. Rietvlei Wetland Reserve, City of

Cape Town, Cape Town.
Retief JJ. 2011. Integrated reserve management plan, Table Bay Nature Reserve. Unpublished report. Biodiversity Management Branch, Environmental Resource Management Department, City of Cape Town, Cape Town.
Schwindt E, Bortolus A, Iribarne OO. 2001. Invasion of a reef-builder polychaete: direct and indirect impacts on the native benthic community structure. *Biological Invasions* 3: 137–149.
Scott KM. 1954. Notes on the birds of the Diep River and Rietvlei. *Ostrich* 25: 25–31.
Taljaard S, De Villiers S, Fricke AH, Kloppers WS. 1992. Water quality status of the Rietvlei/Milnerton Lagoon system (Diep River Estuary) – 1988–89. CSIR Data Report EMAS-D 92007. Council for Scientific and Industrial Research, Stellenbosch.
van Niekerk L, Adams JB, Bate GC, Forbes AT, Forbes NT, Huizinga P et al. 2013. Country-wide assessment of estuary health: an approach for integrating pressures and ecosystem response in a data limited environment. *Estuarine, Coastal and Shelf Science* 130: 239–251.
van Niekerk L, Taljaard S, Adams JB, Fundisi D, Huizinga P, Lamberth SJ et al. 2015. *Desktop provisional ecoclassification of the temperate estuaries of South Africa*. WRC Report No. K5/2187. Pretoria: Water Research Commission.
van Niekerk L, Taljaard S, Adams JB, Huizinga P, Lamberth SJ, Snow GC et al. 2010. Reserve determination studies for selected surface water, groundwater, estuaries and wetlands in the Western Cape: Ecological Water Requirements Study – Estuarine RDM Report: Palmiet Rapid Assessment. CSIR Report Number CSIR/NRE/CO/ER/2010/0077/A. Council for Scientific and Industrial Research, Stellenbosch.
van Niekerk L, Turpie JK (eds). 2012. South African National Biodiversity Assessment 2011: Technical Report. Volume 3: Estuary Component. CSIR Report Number CSIR/NRE/ECOS/ER/2011/0045/B. Council for Scientific and Industrial Research, Stellenbosch.
Whitfield AK. 1994. An estuary-association classification for the fishes of southern Africa. *South African Journal of Science* 90: 411–417.
Whitfield AK. 2005. Fishes and freshwater in southern African estuaries – a review. *Aquatic Living Resources* 18: 275–289.
Wynberg RP, Branch GM. 1991. An assessment of bait collecting for *Callianassa kraussi* Stebbing in Langebaan Lagoon, Western Cape, and of associated avian predation. *South African Journal of Marine Science* 11: 141–152.

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