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ENVIRONMENTAL FLOWS AND THE HEALTH AND VALUE OF THE BERG RIVER ESTUARY

Potential trade-offs between estuary value and regional water supply under a changing climate.

February 2021

ENVIRONMENTAL FLOWS AND THE HEALTH AND VALUE OF THE BERG RIVER ESTUARY:

Potential trade-offs between estuary value and regional water supply under a changing climate

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GLOSSARY

Anaerobic -- Living, active, occurring, or existing in the absence of free oxygen, as opposed to aerobic which means living, active, or occurring only in the presence of oxygen.

Anthropogenic -- Relating to, or resulting from the influence of human activity – generally eliciting environmental pollution.

Aquatroll -- A water quality data logger that records temperature and salinity at regular intervals throughout the day.

Assurance of Supply – The amount of water that can be delivered with a specified level of guarantee, based on the % of the time that it would not be delivered.

Baseflow -- The portion of the streamflow that is sustained between precipitation events, fed to streams by delayed pathways.

Benthic microalgae -- Microscopic algae that live benthically on rock or sediments in the water body, also called microphytobenthos.

Benthic/benthos -- The ecological region at the lowest level of a body of water such as an ocean, lake, or stream, including the sediment surface and some sub-surface layers.

Biodiversity -- The variability among living organisms from all terrestrial, marine, and other aquatic ecosystems, and the ecological complexes of which they are part: this includes diversity within species, between species and of ecosystems.

Biogeochemistry -- The study of the chemical, physical, geological, and biological processes and reactions that govern the composition of the natural environment.

Biomass -- The total quantity or weight of organisms in a given area or volume.

Black tide -- An event of low oxygen water on the West Coast of South Africa that cause mass mortalities of marine species that results from the decay of large plankton blooms under calm conditions.

Catadromous – Species that mature in freshwater and return to the sea to spawn.

Chlorophyll a -- A green pigment, present in all green plants (including algae) and cyanobacteria, which is responsible for the absorption of light to provide energy for photosynthesis.

Coastal trapped waves -- A class of waves dependent on the presence of a region of shallow ocean between the coast and the deep ocean. These waves exist because the shelf is not of uniform depth but falls off gradually towards the deep ocean; in other words, a major ingredient of their dynamics is the presence of a sloping sea floor or a bottom gradient. The differ from a coastal Kelvin wave in that Kelvin waves depend on the existence of a coast against which they can "lean" but do not require the existence of a shelf region. Like Kelvin waves, coastal trapped waves can only propagate towards the equator on the west coast of the oceans and towards the poles on the east coast of the oceans. They have similar periods as coastal Kelvin waves (several days to 2-3 weeks) and similar wave lengths (of the order of 2000 km, determined by the atmospheric weather patterns) but a different wave profile (with a relative maximum of sea level oscillation over the shelf edge). If the stratification of the shelf waters is taken into account, their

shape and associated currents are modified further, and they can have strongest currents at mid-depth. Because the stratification in shallow water is strongly affected by the seasons, the effect of coastal trapped waves on shelf currents and sea level variations can vary with the seasons.

Community -- In ecology, a community is a group or association of populations of two or more different species occupying the same geographical area and in a particular time.

Community composition -- The number of species in that community and their relative numbers.

Diatoms -- A type of phytoplankton group that form a silica-based cell wall.

Dinoflagellate -- A type of flagellate phytoplankton. Some produce toxins that can accumulate in shellfish, resulting in poisoning when eaten.

Downwelling -- The process of accumulation and sinking of higher density material beneath lower density material, such as cold or saline water beneath warmer or fresher water or cold air beneath warm air. It is the sinking limb of a convection cell.

Ephemeral -- Lasting for a very short time.

Euphotic zone -- Also known as the photic or sunlight zone; the uppermost layer of the ocean that receives sunlight enabling it to perform photosynthesis. It undergoes a series of physical, chemical, and biological processes that supply nutrients into the upper water column.

Euryhaline -- Organisms with a broad range of salinity tolerance.

Flagellates -- A type of phytoplankton group with one or more whip-like appendages called flagella.

Floodplain -- Broad and relatively flat area on either side of a stream, river or estuary that are inundated by water during floods.

Fluvial -- River/freshwater.

Fluvial input/flow -- A term used to refer to the processes associated with rivers and streams and the deposits and landforms created by them.

Fluvially -dominated -- River-dominated.

Frontal systems -- A weather front is a boundary separating two masses of air of different densities and is the principal cause of meteorological phenomena outside the tropics.

Halophytic -- A plant adapted to growing in saline conditions, as in a salt marsh.

Head (of estuary) -- The upstream part of the system where freshwater enters.

Hyperbenthic -- Animals that live just above the sediment/benthic surface

Hyperbenthic/ Hyperbenthos -- Benthic organisms that live just above the sediment.

Hypoxia -- When a body of water or a region becomes oxygen depleted

Infauna -- Animals that live on or within the sediment/benthos

Intertidal -- Also known as the foreshore or seashore, is the area that is above water level at low tide and underwater at high tide.

Intertidal (areas/zone) -- Also known as the foreshore or seashore; the area that is above water level at low tide and underwater at high tide.

Invasive alien species are species who's natural range occurs outside of South Africa and which were transported to their current location by humans; where they are able to reproduce, spread and typically cause negative ecological impact.

Longitudinal density gradient -- Drives estuarine circulation or gravitational circulation. This is the density gradient created along the estuary created by the inflow of freshwater from the head, and saline water from the mouth.

Macroalgae -- Also known as seaweed; refers to several species of macroscopic, multicellular, marine algae. The term includes some types of Rhodophyta, Phaeophyta and Chlorophyta macroalgae.

Macrophyte -- An aquatic plant large enough to be seen by the naked eye.

Mean annual potential evaporation (MAPE) -- Potential evaporation is the amount of evaporation that would occur if a sufficient water source were available. This is averaged over a year to give MAPE.

Microalgae -- Microscopic algae, typically found in freshwater and marine systems, living in both the water column and sediment. They are unicellular species which exist individually, or in chains or group.

Neap tide -- A tide just after the first or third quarters of the moon when there is least difference between high and low water.

Oceanography -- The study of the physical and biological aspects of the sea.

Orographic rainfall -- Rain that is produced from the lifting of moist air over a mountain.

Perennial -- Lasting or existing for a long or apparently infinite time; enduring or continually recurring i.e. permanent.

Phyto-detritus -- Organic particulate matter resulting from phytoplankton and other organic material in surface waters falling to the seabed. This process takes place almost continuously as a "marine snow" of descending particles.

Phytoplankton -- Microscopic organisms that live in aquatic systems that are able to photosynthesize to feed themselves.

Planktivore -- An organism that feeds on planktonic food, including zooplankton and phytoplankton.

Plankton -- Organisms drifting in oceans, seas, and bodies of fresh water. The word zooplankton is derived from the Greek zoon, meaning "animal", and planktos, meaning "wanderer" or "drifter". Typically comprised of phytoplankton and zooplankton, as well as the eggs, larvae and juveniles of larger animals.

Primary producers -- An organism that converts an abiotic source of energy into energy stored in organic compounds, which can be used by other organisms i.e. plants and algae.

Red tides -- A discoloration of seawater caused by a bloom of toxic red dinoflagellates. Many red tides produce toxic chemicals that can affect both marine organisms and humans. The red tide toxins can accumulate in molluscan filter-feeders such as oysters and clams, which can lead to neurotoxic shellfish poisoning in people who consume contaminated shellfish.

Remineralisation -- The breakdown or transformation of organic matter (those molecules derived from a biological source) into its simplest inorganic forms.

River Estuary Interface (REI) -- Productive zone within an estuary where marine and freshwater meets, salinity is less than that of sea water and organic deposition due to reduced current speeds and flocculation occurs and is retained.

Secondary consumers -- Organisms that eat primary consumers for energy. Primary consumers are always herbivores, or organisms that only eat autotrophic plants. However, secondary consumers can either be carnivores or omnivores.

Semidiurnal tidal -- A tidal cycle with two nearly equal high tides and low tides every lunar day.

Spring tide -- The tide with the greatest difference between high and low tide which occurs following a full or new moon – when the earth and the moon are aligned.

Stenohaline -- Organisms that either are unable to tolerate, or barely able to tolerate salinity changes; organisms with a narrow range of salinity tolerance.

Substratum -- An underlying layer or substance, in particular a layer of rock or soil beneath the surface of the ground.

Subtidal -- Applied to that portion of a tidal-flat environment which lies below the level of mean low water for spring tides. Normally it is covered by water at all states of the tide.

Supratidal -- The area above the spring high tide line, on coastlines and estuaries, that is regularly splashed, but not submerged by ocean water. Seawater penetrates these elevated areas only during storms with high tides

Thermocline -- A clear boundary layer separating warm and cool water.

Tidal inundation -- The total water level that occurs on normally dry ground as a result of the tide and is expressed in terms of height of water above ground level.

Turbidity -- The quality of being cloudy, opaque, or thick with suspended matter.

Turbulent flow -- A type of flow in which a fluid undergoes irregular fluctuations, or mixing, in contrast to laminar flow, in which the fluid moves in smooth paths or layers. In turbulent flow the speed of the fluid at a point is continuously undergoing changes in both magnitude and direction.

Upwelling -- A process in which deep, cold water rises toward the surface. Water that rises to the surface as a result of upwelling is typically colder and is rich in nutrients. These nutrients "fertilise" surface waters, meaning that these surface waters often have high biological productivity.

Upwelling cell -- A specific area of intense upwelling.

Urbanisation -- The process of making an area more urban – as areas and populations grow and develop into cities.

Water residence time -- the flow rate that would fill the estuary in three spring tidal cycles or 42 days.

Wind stress -- In physical oceanography and fluid dynamics: the shear stress exerted by the wind on the surface of large bodies of water such as oceans, seas, estuaries and lakes. It is the force component parallel to the surface, per unit area, as applied by the wind on the water surface.

Xeric -- (of an environment or habitat) containing little moisture; very dry

Yield – The yield of a dam is the maximum annual volume of water that can be abstracted from the dam at a specific annual assurance of supply or recurrence interval of failure of supply, for example 98% assurance or 1:50 year recurrence interval. The annual volume abstracted is typically distributed over the twelve calendar months according to a site-specific seasonal distribution.

Zooplanktivorous -- An organism that consumes zooplankton

Zooplankton -- Heterotrophic plankton (i.e. plankton that cannot manufacture its own food by carbon fixation and therefore derives its intake of nutrition from other sources of organic carbon, mainly plant or animal matter. In the food chain, heterotrophs are secondary and tertiary consumers).

ABBREVIATIONS

Anchor	Anchor Research and Monitoring (Pty) Ltd
AOA	Annual operating analysis
BRIP	Berg River Improvement Plan
Chl a	Chlorophyll a
CSIR	Council for Scientific and Industrial Research
CWAC	Coordinated Waterbird Counts
CWDP	Coastal waters discharge permit
DEA&DP	Department of Environmental Affairs and Development Planning
DIN	Dissolved Inorganic Nitrogen
DIP	Dissolved Inorganic Phosphate
DO	Dissolved oxygen
DWS	Department of Water & Sanitation
EA	Environmental Authorization
EGSA	Ecosystem goods, services and attributes
EHI	Estuary Health Index
EIA	Environmental Impact Assessment
EWR	Environmental Water Requirement
GI	Gastrointestinal illness
HFY	Historical firm yield
IAP	Invasive alien plant
km	Kilometres
LBIB	Lower Berg Irrigation Board
MAPE	Mean annual potential evaporation
MAR	Mean annual runoff
PAH	Polycyclic aromatic hydrocarbons
POMA	Port Owen Marina Association
PSU	Practical Salinity Unit
RDM	Resource Directed Measures
REI	River-Estuary Interface
Rm	Millions of Rands
RQIS	Resource Quality Information Services
RQO	Resource Quality Objective
WCDM	West Coast District Municipality
WCWSS	Western Cape Water Supply System
WRYM	Water Resources Yield Model

EXECUTIVE SUMMARY

Introduction

The Berg River Estuary is one of the largest estuaries in South Africa, covering an area of 61 km^2, and encompassing about 60% of the estuarine habitat on the West Coast. It is one of the most important estuaries in the country in terms of its biodiversity, and is a valued nursery area for angling and commercially important fish species.

The delivery of ecosystem goods and services from the Berg River Estuary is contingent on two broad sets of policy decisions: (1) adequate quantities and quality of freshwater inflows in relation to seasonal requirements, and (2) management of activities in and around the estuary itself. The freshwater flow requirements are gazetted as Resource Quality Objectives following the Classification of the system. In 2008, a Resource Directed Measures (RDM) study put the estuary in a C-class of health. The Classification study of 2018 determined that the estuary should be retained in a C-class, but flow requirements were only gazetted for the dry season. The *in situ* management of the system is guided by the Berg River Estuary Management Plan (EMP), updated in 2019, but this has not yet been fully implemented.

There are concerns that the Berg River Estuary continues to face significant threats to its biodiversity, sense of place and value. Critically, the gazetted freshwater requirements were set on the basis of historical climate conditions and may not achieve the intended C-class under future climate change. Furthermore, they were based on an outdated body of work, with little research having been carried out over the last 15 years, which includes the most intense drought ever recorded.

This study provides an updated understanding of the ecological functioning, and the intrinsic, cultural and socio-economic value of the Berg River Estuary, and the implications of these for management of the estuary and its catchment water supplies. It incorporates new data collected since 2005 on freshwater inflow, water quality, birds and fish, and provides an updated assessment of its health status. It then examines a range of possible future scenarios to evaluate the trade-offs between allocating flows to maintain or restore estuary function, versus competing demands for water such as urban, agricultural or industrial uses, while considering the reality of climate change.

The scenario analysis considers the implications of (a) ignoring, (b) meeting or (c) exceeding the gazetted flow requirements under (i) present day conditions (demand, water supply infrastructure), (ii) future demand and water supply infrastructure) and (iii) future demand and water supply infrastructure plus climate change. The future scenarios are set at 2040. The implications are considered in terms of water yields, estuary health and estuary value, and the trade-offs among these factors is analysed.

Study approach

The project commenced with a compilation of available data, information and literature on the Berg River Estuary and related areas, processes and sectors. The catchment hydrology and functioning of the estuary system were described, with a focus on estimating actual inflows to the estuary as accurately as possible and extending current understanding to include the recent drought. To this end, estuary data collected since 2005 (water quality and birds) were analysed

along with an extended rainfall and hydrology time series. This was used to update the scoring and overall health assessment of the estuary. Understanding of the socio-economic value of the estuary was also updated.

The study then examined options for the future through a scenario analysis. This aimed to articulate the trade-offs between allocating available water and/or augmenting the system to maintain or restore the estuary health, versus for use in urban, industrial, or agricultural use, taking the reality of climate change into account. The scenarios are summarised in Table I.

Table I. Hydrological scenarios assessed in the study. P0 is the assumed status quo. The naming of the scenarios is linked to development and climate context: present context (P), future hypothetical context without climate change (F) and future predicted context with climate change (C); and to the environmental water requirements (EWRs): ignoring EWRs (0), meeting gazetted EWRs (1); or meeting flow requirements for a C-class (2; this flow is higher than gazetted).

		No EWRs.	Gazetted low flow EWRs (0.6 m^3/sec)	Meeting requirements* for a C-class
Present-day WCWSS Infrastructure (PDI)	Present-day	**P0**	**P1**	**P2**
Future Infrastructure (FI) as planned	Without climate change	**F0**	**F1**	**F2**
Future Infrastructure (FI) as planned	With climate change	**C0**	**C1**	**C2**

* As updated in this study following reassessment of hydrology.

The present status and the alternative scenarios were scored in terms of the Estuary Health Index, a weighted sum of scores for a range of physical (or abiotic) and biotic parameters. This involves (a) estimating what the estuary was like in its natural condition (the Reference condition) (b) scoring the present condition of each component relative to this estimated Reference as a score out of 100, and (c) aggregating the overall score and converting the score to its Present Ecological Status category using a simple scale of A to F, where A represents the best/most pristine option.

The benefits derived from the estuary were estimated using an ecosystem services framework. The value of provisioning, regulating and cultural services were estimated based on existing studies as well as information collected during this study, particularly on property values.

Finally, we estimated the costs of supplying more water to the estuary in terms of the costs of achieving greater sectoral efficiency through demand management, and the costs of augmenting water supply. These measures are more feasible than water reallocation in a stressed catchment.

Hydrology

The 160 km long Berg River has its headwaters in the Jonkershoek and Franschhoek mountains of the Western Cape and flows in a north-westerly direction through Paarl and Wellington to Misverstand, continuing north through Porterville and Moorreesburg, before eventually discharging into the sea at Laaiplek on the West coast. The lower reaches are extremely flat, resulting in sea water intrusion nearly 100 km from the river mouth under high tide conditions.

Factors affecting flows to the estuary

No one has ever known exactly how much water flows into the Berg River Estuary. The nearest flow gauging station is at the Misverstand Dam, 100 km upstream of the head of the 70 km-long estuary. Below that, a number of irrigation farmers extract water directly from the river during the dry season. Their water usage was estimated from spatial data on the crops grown along the river banks. Compared with the natural ("reference") situation (before any water supply infrastructure and human settlement in the catchment), the estuary now receives only 50% of historical freshwater inflows. Furthermore, the timing of flows has changed. While the high flows during the winter rainfall season have been moderately affected, the low flows during the dry season have been greatly diminished to 36% of natural levels.

Where does the rest of the water go? Water use in the Berg River Catchment is primarily for irrigation of vineyards and orchards. There are also demands from forestry plantations in the higher-lying areas as well as alien vegetation infestations. There are some municipal abstractions to Paarl and Wellington, Tulbagh and Saron. A number of large dams have been constructed in the catchment (Berg River, Wemmershoek, Voëlvlei and Misverstand Dams), as well as many small farm dams. These large dams are managed as part of the Western Cape Water Supply System (WCWSS), which supplies water to the City of Cape Town (CoCT) and a number of smaller municipalities. Major water transfers between the Berg River Catchment and dams on the Steenbras, Riviersonderend and Breede Rivers also occur. Thus, water flowing down the Berg River and into its estuary is strongly influenced by the management of this broader WCWSS area.

Inflows into the estuary are more directly linked to management of the Lower Berg River sub-system of the WCWSS, which is made up of the Voëlvlei Government Water Scheme (GWS) and Misverstand Dam. The Voëlvlei GWS comprises the off-channel Voëlvlei Dam and canal diversions from three smaller rivers into the dam. This dam provides water to the CoCT and the West Coast District Municipality (WCDM). Water is also released from Voëlvlei Dam into the Berg River to supply irrigators during the summer months and to supply the Misverstand Dam. The main purpose of Misverstand Dam is to divert water to the Withoogte Water Treatment Works and thence to the Vredenburg/Saldanha area. The current allocations for urban, industrial and agricultural use in the Lower Berg Catchment are given in Table II.

Table II. Water Use Allocations for the Lower Berg Catchment as provided by DWS (January 2020)

Agricultural, Domestic & Industrial	Summer (Nov-Apr) allocation (Mm^3/a)	Winter (May-Oct) allocation (Mm^3/a)
Irrigation boards	51.29	32.41
Other agriculture	12.02	
Withoogte Water Treatment Plant (Saldanha, Swart & Berg)	23.44	
Piketberg	0.7	
Industry	1.39	
TOTAL	**88.84**	**32.41**

However, not all water that is consumed before reaching the estuary is used productively. A considerable amount is **lost to invasive alien plants** (IAPs). IAPs currently reduce system yield by some 27 Mm^3/a, and without action, their unimpeded spread could increase this figure to 95 Mm^3/a or 17% of total system yield by 2045.

The water situation for the estuary continues to deteriorate at an increasing rate. Not only are the demands for water from the WCWSS growing with population and economic growth, but flows are also being negatively impacted by **climate change**. Under climate change, mean annual runoff (MAR) could be reduced by around 15%, and historical firm yield by around 60 Mm^3 or 8% by 2050.

Recognising some of these pressures, the National Water Act (Act No. 36, 1998) provides for safeguarding the health of estuaries in terms of their inflow requirements through Classification and the determination of **Resource Quality Objectives** (RQOs). The RQOs for the Berg River Estuary include (a) the requirement for a minimum of 0.6 m^3/s of flow entering the estuary to prevent unnatural hyper-saline conditions from developing, and (b) a certain proportion of the natural monthly flow requirements during the winter months for periodic scouring of the estuary and habitat maintenance. These provisions are known as **environmental water requirements (EWRs)**.

However, **so far, these safeguards have not been applied**, and not enough water actually reaches the estuary. In fact, there are not even any gauges to ensure compliance. It has been proposed that these EWRs come into force with the development of future augmentation options for the WCWSS, which include the Voëlvlei Augmentation Scheme (VAS).

How estuary inflows and floods compare with reference conditions

We estimated the pattern of flows to the estuary by analysing observed flows at the gauging station (G1H031) below Misverstand Dam and adjusting for estimated agricultural water use and losses to IAPs between the dam and the estuary. Flow gauge G1H031 has a daily record from 1 June 1974. Analysis of these data highlight the extreme nature of the recent drought, where the average daily flow rate for the winter months in 2017 was only 1.2 m^3/s compared to the average winter flow rate of 43.8 m^3/s. A comparison of the simulated natural (from the WRYM) and observed Mean Annual Runoff (MAR) at Misverstand Weir reveals a gradual downward trend in both observed MAR and simulated natural flows. This may be due to increasing upstream demands as well as climate change.

In the next step we estimated the irrigation demands between Misverstand Dam and the estuary. Two approaches were used: (1) based on estimated areas and net water requirements of different crop types, and (2) based on the allocations downstream of Misverstand weir, from the Department of Water and Sanitation (DWS).

The full allocation from DWS is 7000 m^3/ha. This is split between summer and winter allocations. The summer allocation is split into 3000 m^3/ha provided from Voëlvlei and Misverstand dams and 4000 m^3/ha provided from run-of-river. For the winter portion, the total allocation of 7000 m^3/ha comes from run-of-river. The 18 million m^3/a allocation used in the Water Resources Yield Model (WRYM) is based on the 3000 m^3/ha for the Lower Berg Irrigation Board (LBIB) users situated upstream **and** downstream of the Misverstand weir. The total allocation for the LBIB users *downstream* of Misverstand weir is 9.4 million m^3/a in the summer months and 7.6 million m^3/a in the winter months.

Based on land cover data, there has been an increase in irrigated areas of approximately 24% from 2000 to 2018. This suggests that there is a much greater summer irrigation demand than is currently allowed for under the DWS allocations i.e. 16 million m^3/a compared to 9.4 million m^3. There has been a significant increase in irrigation demand downstream of Misverstand (Figure I).

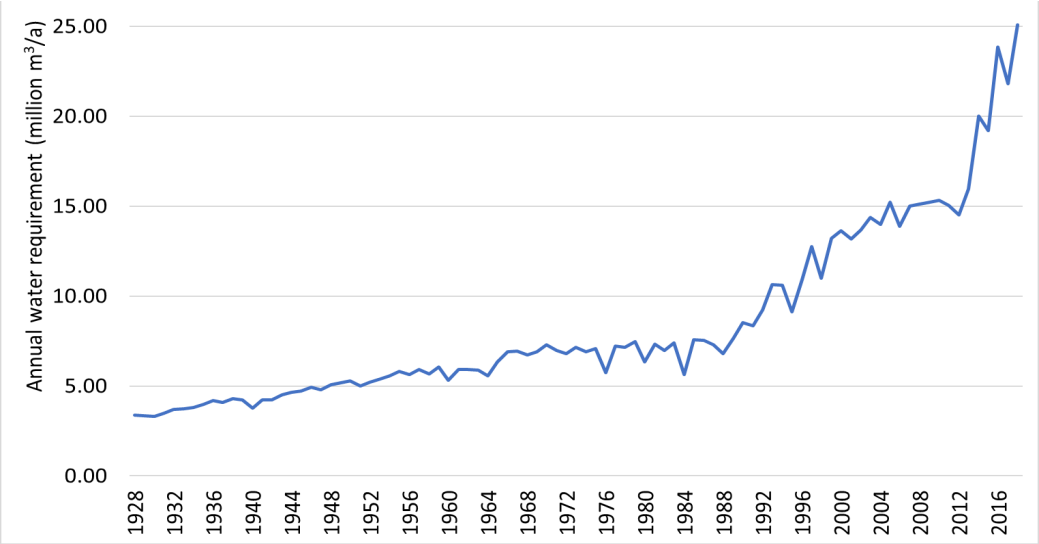


Figure I. Estimated annual historical irrigation demand downstream of Misverstand Dam

The mean monthly historical inflows to the estuary are estimated to drop below 0.6 m³/s from January to March. The historical data show that summer flows only reached this level less than 15% of the time from 1974 to 2019. The data also indicate a noticeable decrease in summer flows associated with increased irrigation demand from the early 1990s to the present day. Under present conditions, **there is typically no flow reaching the estuary in the summer months**, as it is all being used to meet irrigation demands downstream of Misverstand. Floods in the Berg River Estuary occur mostly during the winter months of June, July and August. There was no clear evidence of climate change impacts on annual maximum flood peaks up to now. However, the Berg River Dam has had an impact on flood peaks since its construction in 2007.

What could happen under alternative future scenarios?

At present, the estuary is estimated to receive **less than half** (47.4%) of the freshwater inflows that would have occurred under Reference condition (Table III). These flows are less than the gazetted requirements for the estuary. If the latter were met, it would receive just over half (51.3%) of natural flows. Scenario analysis of future changes relative to present day are presented in for historical firm yield (HFY) and the MAR of inflow to the estuary. These results suggest that while the historical firm yield does not seem to be affected by providing the estuary low flow EWRs of 0.6 m³/s, the mean annual runoff to the estuary is slightly affected. This is likely due to the fact that the minimum low flow requirement of 0.6 m³/s in the summer months does not represent a significantly large portion of the MAR. It is also important to note that despite allocations being gazetted, the minimum flows meant to be reaching the estuary are not actually getting there. In future it will be necessary to ensure compliance with these allocations to meet the minimum estuary flows. The impact of climate change on the historical firm yield and particularly the volume reaching the estuary is much more substantial.

When the estuary low flow EWRs are applied in Scenarios P1 and F1, the average minimum flows in summer meet the requirement of 0.6 m³/s. However, there are still some months when this may not be the case. When no provision is made for the minimum flow, the estuary flows drop below 0.6 m³/s **more than 50% of the time**. Also, of concern is how climate change reduces the frequency of high winter flows which are essential to the functioning of the estuary. During the summer months it is clear that under the present day scenario, the minimum low flow EWRs **are**

not being met. Under the future scenario EWR releases will need to be made during the summer months, and effort will need to be invested to ensure that these reach the estuary.

Table III. Results of hydrological scenario analysis.

Scenario	Scenario	HFY (Mm^3/a)	Percentage change from present day HFY (%)	Average Annual inflow to Estuary (Mm^3/a)	Percentage change from present day MAR (%)
Natural	Reference	n/a	n/a	912.4	98%
Present day, no EWR	P0	507.8	0%	459.2	0%
Present day, min EWR	P1	507.8	0%	468.6	2%
Future, no EWR	F0	528.1	4%	432.8	-6%
Future, min EWR	F1	528.1	4%	438.4	-5%
Future CC, no EWR	C0	479.9	-5%	303.2	-34%
Future CC, min EWR	C1	470.2	-7%	312.3	-32%

Changes in hydrological health under Present Day and the Future Scenarios

The hydrological health is calculated on the basis of (a) general inflow patterns, and (b) the frequency and magnitude of flood events relative to the Reference state. Results of this assessment suggest that the hydrological health is in a very poor state (PD 36.3% - i.e. seriously modified, Table IV). This is much worse than the score in 2010 (DWA 2010, 72%). The change in score is partly due to reductions in flow since 2010, but largely due to the fact that an effort was made in this study to calculate the actual volumes of water reaching the estuary, rather than assuming that all flow released from Misverstand Weir reaches the estuary. Incorporating the effects of climate change suggests a severe impact on the hydrological health of the Berg River Estuary, with scores dropping to 22.9% without any protection of the low flow EWRs (C0), with slight improvement (28.7%) if low flow EWRs are respected (C1).

Table IV. Change in hydrology functioning score.

	Wt	Hist. (2010)	PD (2020)	P1	F0	F1	C0	C1
				PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
1a. Present MAR as % of Ref. MAR	60	68	50.3	51.4	47.4	48.0	33.2	34.2
1b. Present Low flows as % of Ref. Low flows			17.3%	22.1%	17.3%	35.9%	9.5%	28.0%
2. % Similarity in mean annual frequency of floods	40	79	40.0%	40.0%	40.2%	39.1%	25.1%	25.1%
Health score = weighted mean of 1 and 2		72	36.3	38.0	35.5	40.8	22.9	28.7

Hydrodynamics & sediments

Hydrodynamics

The hydrodynamics of the Berg River Estuary are complex. Key drivers influencing its hydrodynamic functioning are weather, the volume and characteristics of freshwater inflows, the characteristics of seawater entering the system, sea level, and channel morphology and floodplain dynamics. The hydrological regime of the Berg River Estuary is driven by strong seasonality in rainfall, air temperature and evaporation. More than 80% of the rain falls in winter, bringing floods that carry silt from the catchment and submerging the floodplains adjacent to the estuary. Freshwater inflows flush out sea water which penetrates into the lower reaches of the estuary due to tidal influences. Freshwater spates exceeding 140 m^3/s are sufficient to fully flush the estuary of saline water (Schumann 2009). Flooding of the area alongside the estuary is of crucial importance to the ecology of the system, and to restrict sediment build-up. The flooding regime is influenced by upstream water levels at the start of a flooding sequence. Drought conditions and uptake of water upstream therefore result in a smaller area of floodplain being inundated, with impacts on the ecological functioning of the system.

There has been significant modification to the Berg River Estuary in the last 50 years, most notably at the mouth. In 1969, the estuary mouth was artificially shifted one kilometre to the northeast to connect the estuary directly to St Helena Bay. This new canalised mouth allows a relatively unrestricted exchange of water between the estuary and ocean. Other anthropogenic influences that have changed the channel morphology of the estuary include bridge construction and development alongside the estuary.

Despite the seemingly large nature of these anthropogenic changes, impacts on the hydrodynamic functioning and sediment transport in the Berg River Estuary as a whole have remained relatively limited, mostly due to its large size. Very little development has taken place on the floodplains in the upper estuary, and this portion of the estuary has retained much the same historical size and course. Effects of the anthropogenic developments described above are seen mostly in lower reaches of the estuary, particularly at the mouth.

Sediment dynamics

Typically, estuaries contain a mixture of river and marine sediments, the balance of which is determined by the amount of water moving in and out of the estuary during a tidal cycle, and the amount entering through riverine base flows and floods.

Under low flow conditions, erosion and deposition are confined to the main channel, and the overall volume of sediment transported is very small. Under flood conditions, erosion and deposition patterns are much more extensive. Floods carry a large amount of silt from the catchment, which is deposited when floodwaters slow down. Floods also flush out sediments that have built up in the system. Sedimentation in the lower reaches of the estuary may have increased due to the new canalised mouth, and the resultant decline in tidal flushing of the system.

The sediment structure of the channels and floodplains are directly linked to the erosion issues of the lower Berg River Estuary. Erosion of channel banks is potentially threatening valuable habitat. Areas where bank erosion are a concern are at Admiral Island, Carinus Bridge, Cerebos Saltworks, Kuifkopvisvange and Kliphoek.

How physical habitats, sediment processes and hydrodynamics compare with reference conditions

Under reference conditions, periodic floods restricted the build-up of riverine sediment in the estuary, and the entry of marine sediment into the system. However, intertidal areas have been affected by anthropogenic activities over the years. Moving and permanently fixing the mouth has disrupted the natural movement and deposition of sediments and has created a deeper channel, greater tidal flux and allowed more marine sediment intrusion into the estuary. A reduction in the number and size of floods has also had an impact on sediment processes due to less flushing, reduced sediment transport and reduced scouring capacity. Bridges and embankments have caused siltation of the channel and contributed to bank erosion and habitat degradation. The system would have been significantly more constricted in the reference state during the summer months. Closure would have occurred during the dry summer, but no longer occurs today due to the canalised and permanently open mouth.

Under present day conditions, lower freshwater inputs mean higher residence times during summer, with only 47-50% of the natural flow reaching the estuary. Even during winter months, less freshwater from the catchment is reaching the system, increasing residence times relative to reference conditions. Water levels would naturally have shown some variability between seasons due to consistent flooding during the winter months. However, with average flows lower under present day conditions and with fewer "wet" months, the water levels under present day conditions are less variable between seasons.

Changes in hydrodynamic health under Present Day and the Future Scenarios

At present, the physical habitats and sediment processes associated with the Berg River Estuary are relatively stable, with the biggest change from the reference state occurring in the lower section of the estuary where human activities have altered the natural functioning of the system. Water retention time and water levels are the two indicators that have changed the most relative to the Reference condition.

The 2010 RDM study attributed much of the changes in physical habitat score to development around the estuary. Since 2010 the changes in physical habitat have remained relatively stable with the scores remaining the same for two of the three indicators (Table V). The present condition of physial habitats has declined further, largely due to changes in the hydrological regime which have resulted in less sediment flushing, a longer time for fluvial sediments to pass through upper reaches and reduced coarse sediment deposition due to dam trapping. Under Scenarios P1, F0 and F1, where changes in flow are not significant, the scores remain the same as present day. However, the scores change quite dramatically for Scenarios C0 and C1 where future flows into the estuary are reduced significantly under climate change, and sediment processes are negatively affected. For both the climate change scenarios the overall health score declines to 59%, from the present value of 66%.

The overall hydrodynamic health of the estuary is scored based on five indicators – mouth condition, abiotic state, stratification, retention time, and water level. Overall, the hydrodynamics score of the Berg River Estuary is lower under present day (2020) than that assessed in 2010, though only one of the four hydrodynamic parameters were scored in 2010 (Table VI).

Table V. Change in physical habitat and processes score.

	Ref	2010	P0 2020	P1	F0	F1	C0	C1
				PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
a. % Similarity in intertidal area exposed	100.0	61.0	61.0	61.0	61.0	61.0	60.0	60.0
b. % Similarity in sand fraction relative to total sand and mud	100.0	75.0	75.0	75.0	75.0	75.0	55.0	55.0
c. Resemblance of subtidal estuary to Reference condition: depth, bed or channel morphology	100.0	74.0	64.0	64.0	64.0	64.0	60.0	60.0
Health score = weighted mean of a and b and c	100.0	71.0	66.0	66.0	66.0	66.0	59.0	59.0

Scenarios that include climate change (C0 and C1) show a further reduction in hydrodynamic health, especially due to declines in retention time and water level health scores (Table VI). This score does not change much for scenarios P1, F0 and F1, indicating little impact from EWR flows. Scenarios that include climate change (C0 and C1) show a further reduction in hydrodynamic health (46.4% and 46.8% respectively), especially due to declines in retention time and water level health scores (Table VI).

Table VI. Hydrodynamic scoring of the system under historical (2010), present day (2020) and for all considered scenarios.

	Wt	2010	P0 2020	P1	F0	F1	C0	C1
				PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
Mouth	50	90.0	90.0	90.0	90.0	90.0	90.0	90.0
Abiotic state		—	69.3	68.4	66.3	73.4	58.1	64.2
Stratification	10	—	90	90.0	90.0	90.0	80.0	80.0
Retention time	20	—	42.6	42.6	42.6	42.6	21.3	21.3
Water level	20	—	45.5	45.5	41.2	40.7	20.0	20.0
Health score		90.0	60.8	60.8	58.9	61.3	46.4	46.8

Water Quality

Factors affecting water quality in the estuary

The water quality characteristics of an estuary include physical properties such as temperature, salinity and dissolved oxygen. Water quality is also determined by dissolved substances, such as nutrients and trace metals, and suspended matter in the water, like sediments and microorganisms. These are affected by various sources, including runoff from the river catchment area, the atmosphere and tidal inputs from the sea.

There has been little change in the conditions in St Helena Bay over the last 70 years (Hughes *et al.* 1991, Mather *et al.* 2009). The characteristics of the estuary mouth have changed markedly, most importantly through diversion and canalisation, but also through sedimentation and dredging. However, the biggest change to the system has been to the conditions in the Berg River Catchment and resultant impacts on the quantity and quality of freshwater reaching the estuary. This is mostly due to transformation of land in the catchment for agriculture, urban and industrial development, and changes in rainfall patterns (Taljaard *et al.* 2010, Cullis *et al.* 2019). Direct inputs from activities on and along the margins of the estuary have also had a significant impact on water quality in the estuary.

Salinity

Salinity changes with depth and distance upriver during varying tides and seasons of a year. During summer (low flow), the saline water from St Helena Bay penetrates between 43 km to 45 km up the estuary. During winter months with higher incoming freshwater flow, the salinity intrusion is greatly reduced to 3 km to 15 km. Under normal summer conditions salinity within the Berg River Estuary declines from the mouth to the upper reaches. Alarmingly, data collected for the DEA&DP reveals that during the recent drought (2015-2018), the lower to middle reaches of the estuary experienced a reverse salinity gradient, where the salinity actually increased from the mouth to the station 23 km upstream. This resulted from the lack of freshwater reaching the estuary from the catchment.

In the lower reaches of the estuary salinity remains high throughout the summer months, when river flow is lowest. In winter salinity is much more variable, as seawater is periodically flushed out by freshwater inflows. During the recent drought, variation in salinity in the lower reaches of the estuary during winter was greatly reduced. As a result, the estuary remained highly saline (>17) for a period of almost 18 months, while salinity did not drop below 5 for more than 18 months in the middle reaches. This has important implications for other water quality parameters, dissolved oxygen and nutrients in particular. It can also have profound impacts on estuarine biota, which are adapted to and generally require seasonal reductions in salinities.

Reduced freshwater inflow during the drought also had a marked impact on maximum salinity levels attained in the estuary. In the middle reaches, maximum levels attained each year increased from ~32 in 2016, to ~35 in 2017, and reached ~39 in 2018, which is substantially higher than normal sea water (34.5). At Jantjiesfontein, 52 km from the estuary mouth, salinity values as high as 4 were recorded during the summer of 2016/2017. This is concerning as these levels exceed the gazetted RQOs, which state that salinity across the estuary should not exceed 35 , and salinity in the areas above 40 km upstream of the mouth should not exceed 1 .

Dissolved Oxygen

Natural levels of dissolved oxygen are governed by temperature and salinity, as well as the organic content of the water. Oxygen is removed from the water column by the respiration of

biota and through decomposition of organic matter. Marine waters entering the mouth of the Berg River Estuary have a wide range of dissolved oxygen values (1-10 mg/l). Oxygen levels in the freshwater entering at the head of the estuary are mostly high (8-12 mg/l).

In winter, highly oxygenated freshwater spates fill the estuary and flush any oxygen depleted water plugs out of the system. However, a reduction in winter river inflows limits or prevents the expulsion of these oxygen depleted plugs, which can negatively impact species that cannot tolerate low levels of dissolved oxygen. If the oxygen concentration within these oxygen depleted plugs drops below 4 mg/l, juvenile fish species may start to be negatively affected, while mass mortalities of fish can occur at concentrations below 2 mg/l.

Nutrients

Nutrient cycling drives biological production in estuaries. Human activites may disrupt the natural balance of nutrient cycling, leading to changes in the system's trophic structure. The introduction of nitrogen and phosphorus through land clearing, application of fertilizer, discharge of human wastes, animal production, and urban runoff is known as **eutrophication**. Increased nutrient levels stimulate undesirable plant growth, such as phytoplankton. Phytoplankton growth reduces the amount of light reaching other plant species, thereby inhibiting their growth. Other undesirable effects include oxygen depletion from the decomposition of phytoplankton and plant biomass.

Dissolved Inorganic Nitrogen (DIN) increased slighlty with increasing distance from the mouth, suggesting that it is mainly being introduced by freshwater inflow. Levels of DIN in winter river inflow have almost doubled in the last decade (Table VII). This is probably linked to increased agricultural activities and expansion of human settlements in the catchment. The DIN levels in seawater entering at the mouth of the estuary have remained stable.

Dissolved total ammonia decreased with distance upstream. However, values for ammonia within St Helena Bay are significantly lower than those recorded in the lower estuary. This suggests that ammonia is being introduced into the estuary through discharge from the fish factory near the mouth. Ammonia concentrations have increased markedly in the middle reaches of the estuary during low flow periods (Table VII). This suggests that nutrients entering in the mouth area are being pushed further upstream. Levels of total ammonia entering the estuary during the high flow season from the catchment have also increased markedly in the last decade.

Table VII. Mean concentration of dissolved inorganic nutrients (dissolved inorganic nitrogen, ammonia and inorganic phosphate) in freshwater (<1) flowing into the Berg River Estuary during high and low flow seasons between 1989 and 2019.

	Year	NOx-N ($\mu g.l^{-1}$)	NH4-N ($\mu g.l^{-1}$)	PO4-P ($\mu g.l^{-1}$)
Low flow (Summer)	1990	70.0	57.0	41.0
	1996	36.6	32.7	13.7
	2005	37.0	21.1	6.4
	2013-2019	650.0	25.0	60
High flow (Winter)	1989	644.1	73.6	27.2
	1995	1193.2	56.3	44.5
	2005	880.0	48.1	38.6
	2013-2019	1245.0	123.9	68.4

Dissolved Inorganic Phosphate (DIP) showed similar patterns across all periods with levels not varying greatly from the mouth to middle reaches of the estuary. The summer dissolved inorganic phosphate was higher than that of winter values, when river flow was high. This indicates the introduction of phosphates from the ocean by tidal exchange. Historic data show that there has been a marked increase in dissolved inorganic phosphate concentrations in the lower portions of the estuary during low flow summer periods (Table VII). It is assumed that this is linked to a reduction in freshwater inflows and a simultaneous increase in volumes of seawater in the estuary. Levels of dissolved inorganic phosphate during winter have also increased in the last decade, consistent with increased levels recorded at sites further upstream in the catchment.

Escherichia coli *(E. coli),* an indicator of pathogenic micro-organisms, was also monitored at high risk sites. Data collected from the lower and middle reaches of the estuary from 2013 to present suggest that average *E. coli* levels fluctuate throughout the year, with a peak count occurring in both the high and low flow periods. Averaging the data at each station shows overwhelmingly that the major source of *E. coli* contamination is at the mouth of the estuary at station 10, which is located next to the Laaiplek Fishing Harbour and Amawandle Pelagic. *E. coli* levels at this site exceeded the prescribed limit above which a water body is considered "poor" or "unfit" for recreational purposes, indicating that the fish factory may pose a risk to the tourism industry.

Heavy metals

In 2010, measured trace metal concentrations at some stations along the estuary exceeded South African and international sediment quality guidelines for certain metals including Arsenic, Cadmium, Chromium, Copper, Mercury, Nickel and Lead.

Changes in water quality under Present Day and Future Scenarios

In characterising the hydrodynamic and water quality characteristics within each abiotic state, the estuary was sub-divided into four zones (Figure II). These zones were largely derived from typical salinity distributions and channel bathymetry.

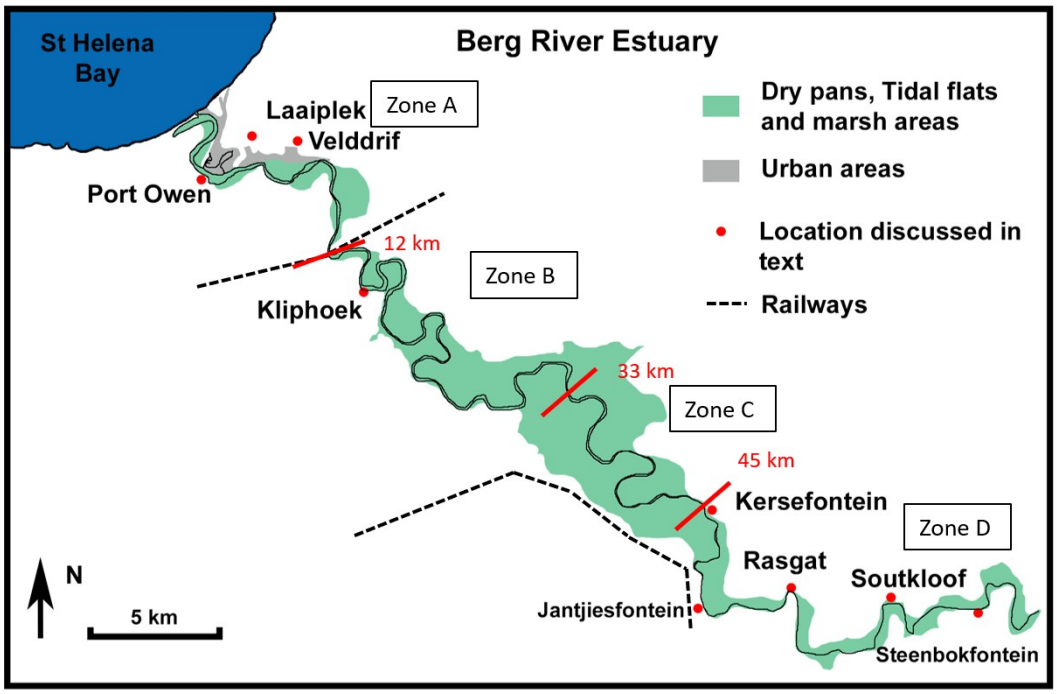


Figure II. Map showing Zones A to D referred to in the text (map adapted from Schumann, 2007).

Salinity is often used to describe the abiotic state within estuaries. In the Berg River Estuary freshwater flow is very seasonal. Winter floods flush out sea water which penetrates into the lower to mid-reaches of the estuary during the summer. The Berg River Estuary has been characterised into five typical abiotic states, summarised below.

The average salinity within the estuary has increased in the present day relative to the reference condition. When compared to the RDM study conducted in 2010 the salinities for Zone A and D are unchanged. However, Zone B and Zone C have higher average salinities relative to 2010. This is likely due to the fact that decreased winter freshwater flows prevent the complete flushing of these zones, allowing the formation of a body of saline water which moves back and forth within the system. Salinity values for scenarios P1, F0 and F1 do not differ greatly from present as flows were not dramatically altered in these scenarios. Scenarios in which EWRs are included show slightly lower average salinities than those without EWRs.

Under climate change conditions, salinities across all zones will increase as a function of two factors. Firstly, sea level rise will push the extent of the salinity intrusion further upstream. Secondly, the reduction of freshwater flows and the reduced flushing of the system overall will increase the likelihood of salinity plugs occurring, as well as the occurrence of reverse salinity gradients. The presence of the reverse salinity gradient and values that exceed 35 (and therefore the gazetted RQO for salinity within the estuary) during the recent drought, raise concerns that RQOs will be more frequently or more severely exceeded under the climate change scenarios.

Table VIII Typical abiotic states of the Berg River Estuary, from Taljaard *et al.* 2010.

STATE	Brief Description	Monthly Flow Range (m^3/s)
1	Severe marine-dominated - saline intrusion extends further than 45 km upstream of mouth (into Zone D, see Figure II)	<0.5
2	Marine-dominated - saline intrusion extends up to 45 km from mouth (downstream of Zone D, see Figure II)	0.5 - 1
3	Small to medium freshwater inflow – marine influence evident up to 33 km from mouth (i.e. downstream of Zone C), with strong freshwater influence in upper ~40 km (in Zones C and D)	1 - 5
4	Medium to high freshwater inflow – marine influence only evident up to 12 km from mouth (downstream of Zone B), with strong freshwater influence in upper ~60 km (in Zones B-D)	5 - 25
5	Freshwater-dominated – estuary is fresh throughout (Zones A-D), except during spring tides when seawater intrusion may extend up to 6 km from mouth into Zone A during high tides	>25

Increased levels of dissolved inorganic nitrogen and phosphateover the last 10 years contribute to the lower present day (2020) score for water quality relative to 2010 (Table IX). Nutrient scores under future development and climate change are expected to drop slightly relative to present day. While nutrient levels within the Berg River may increase, this is mitgated by expected reductions in the amount of nutrient-rich water entering the estuary. Similarly, water clarity within the estuary (2b in Figure II) has declined relative to 2010, as increased nutrients have caused an increase in algal blooms. No new data is available for toxic substances (trace metals, hydrocarbons or pesticides), however, these substances are linked to human activities. Therefore, while we have elected not to change the score for the present day (P0) and P1

scenarios relative to the 2010 RDM study, the scores under the future development (F0 and F1) and climate change (C0 and C1) scenarios are lower.

Overall, the water quality health score for the Berg River Estuary has decreased from 40.2% in 2010 to 31.1% for present day (2020, Table IX). Water quality health scores for scenario P1, F0 and F1 do not differ greatly from present as flows will not be dramatically altered in these scenarios. However, scenarios in which EWRs are included show slightly better scores than those without EWRs. Under future development scenario, the water quality health score drops further to 28.7% (F0) but is slightly better when low flow EWRs are supplied (F1, 30.8%). Scores drop further when the effects of climate change are included (21.5 and 22.1% for Scenarios C0 and C1, respectively). It is important to note that under present conditions the average high flow values for dissolved inorganic nitrogen, average high and low flow values for inorganic phosphates and periodically, the *E.coli* count within the lower reaches of the estuary all exceed the gazetted RQOs. This raises the concern that the occurrence of these exceedances and/or the extent to which the RQOs are exceeded may increase under future scenarios, especially under climate change scenarios C1 and C2.

Table IX. Expected changes in water quality health scores in the Berg River Estuary under the various future flow scenarios. *2010 health scores based on 2010 RDM study.

	Weight	2010*	P0 2020	P1	F0	F1	C0	C1
				PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
1. Salinity	40	63.0	50.8	55.9	48.9	53.0	31.6	33.0
2 - General water quality								
a. N and P concentrations		25.0	18.0	16.9	15.3	16.0	14.8	14.9
b. Water clarity (suspended solids/ turbidity/transparency)		85.0	65.1	63.4	64.9	62.7	57.5	56.1
c. Dissolved oxygen (mg/l) concentrations		85.0	77.3	79.1	72.1	74.0	62.7	64.4
d. Toxic substances		80.0	80.0	80.0	70.0	70.0	60.0	60.0
General water quality = min (a to d)	60	25.0	18.0	16.9	15.3	16.0	14.8	14.9
Water quality health score = weighted mean (1,2)		40.2	31.1	32.5	28.7	30.8	21.5	22.1

Biodiversity

This section provides a description of the various biotic groups of the Berg River Estuary. The focus is on improved understanding of how these biotic communities respond to physical drivers related to changes in freshwater flows, and predicting their responses under future flow regimes. We provide an assessment of the current status (Present Day 2020) of each taxonomic group, which is compared with the findings of the RDM study in 2010. The impacts of the hydrological

scenarios described above are also assessed for each of the taxonomic groups. All scores are relative to a natural "reference" condition.

Microalgae

Microalgae are the unicellular algae that either live suspended in the water column as phytoplankton or settled on the bottom or on plants. Microalgal productivity is the most important determinant of overall biomass of estuarine biota. The microalgae community composition reflects the physical conditions of the estuary, and the main physical drivers include salinity, water residence time and nutrients. Microalgae biomass is highest in the river estuary interface zone where salinity is 10-15 . This zone shifts seasonally.

Microalgae are more abundant in winter than summer, probably due to higher nutrient loads entering the estuary from the catchment during winter. During the 1990s and early 2000s, there was a dramatic increase in the phytoplankton biomass in the Berg River Estuary, probably due to increases in inorganic nutrient concentrations. This is likely to have inceased further since then. The health of the microalgae assemblages associated with the Berg River Estuary has clearly declined markedly since the system was last evaluated in 2010 (down from 75 to 57% similarity to natural) and is linked to observed reductions in water quality (increased nutrient levels and salinity).

Riparian vegetation

The Berg River Estuary has the largest and most diverse riparian vegetation and wetland habitat of any permanently open estuary in South Africa. Riparian vegetation is dependent on regular and consistent flooding, and is sensitive to changes in salinity.

The main vegetation communities associated with the Berg River Estuary are macroalgae, submerged macrophytes, salt marsh, and reeds and sedges. Their distribution is largely dependent on the longitudinal salinity profile of the system, with marine communities towards the mouth and freshwater communities further upstream.

Macroalgae are generally concentrated in the lower reaches of the estuary. Subtidal vegetation in the lower reaches is dominated by eelgrass *Zostera capensis*, replaced by pond weed *Potamogeton pectinatus* in the fresher upper reaches. Moving out of the channel and regularly-submerged intertidal areas, eelgrass gives way to intertidal salt marsh. Salt marsh composition is strongly zoned by the degree of tidal inundation. Above the intertidal area of the lower estuary, intertidal salt marsh gives way to supratidal salt marsh. These areas are typically flooded in winter.

In the fresher reaches further upstream, the narrow intertidal and floodplain areas are occupied by sedge marsh or taller reed marsh. The floodplain also contains numerous pans with scattered saltpan plants. At higher elevations, the floodplain is occupied by a dry floodplain community. These plants depend on annual floods to reduce salinity levels and deposit soils. In the uppermost reaches of the estuary, the riverbanks are lined by riparian woodland. Alien species such as *Eucalyptus, Acacia* and *Populus* are also common.

The vegetation of the Berg River Estuary has been shaped by a strongly seasonal flow and flood regime which maintains the floodplain systems. A prolonged decline in river input results in a decline in the frequency and intensity of winter floods in the system, and causes the dieback of floodplain vegetation dependent on periodic inundation. Decreased river input may also result in a shift in the salinity profile of the system, impacting species that survive only within specific salinity levels. Extended dry periods will therefore have consequences for the riparian vegetation

community structure, composition and function. This will in turn affect other biological components of the estuary, such as birds and fish. These impacts are compounded by historical and current anthropogenic pressures.

The Western Cape drought of 2015-2018 provided an opportunity to investigate the effects of extended freshwater starvation, when very little flow and no floods were experienced in the estuary. Remote sensing data showed a clear decrease in vegetation during the 2015-2018 drought (Figure III). Vegetation dieback occurred predominantly in the floodplains. There was also a decrease in vegetation along some parts of the main channel, indicating dieback of fringing reeds and sedges due to changes in salinity.

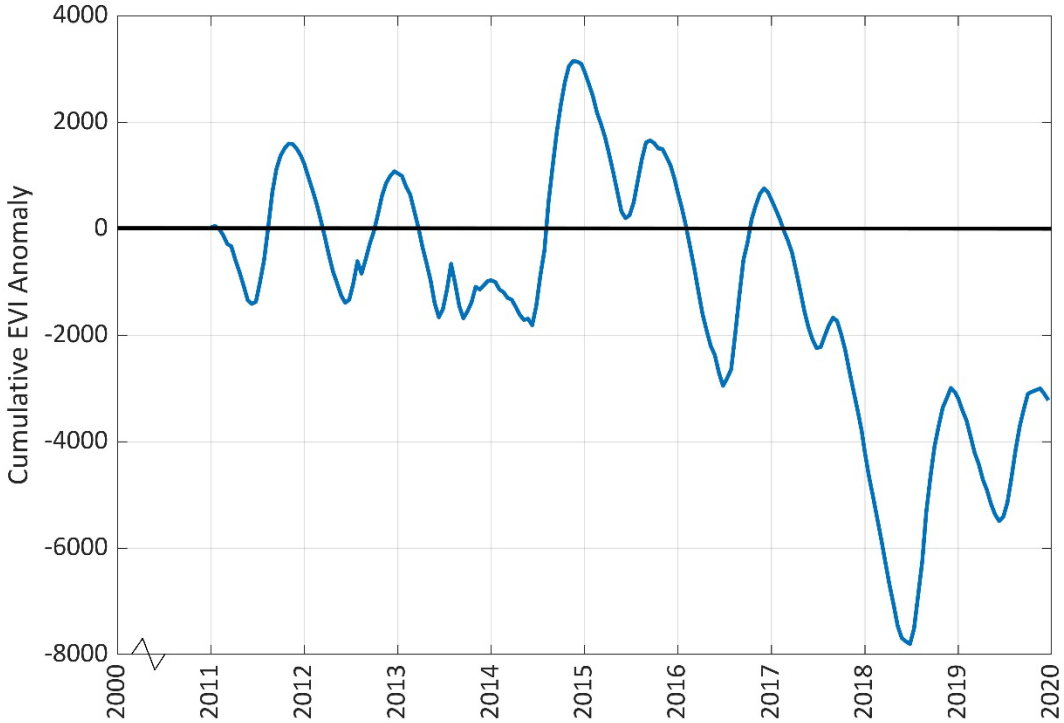


Figure III. Cumulative Enhanced vegetation index (EVI) anomaly for the Berg River Estuary from 2011 to 2020. The 2001-2011 conditions were used to create a baseline mean against which the latter years were compared. With respect to the baseline period; negative values indicate a decrease in canopy structure, while a positive number indicates an increase in canopy structure. By using 10 years as a proxy to establish the baseline EVI, we can decouple annual, intra-annual, and decadal variability in EVI and observe a minimum that occurred during the 2017-2019 Western Cape drought.

Given the expectation of a progressively greater frequency of extreme drought and reduced flooding under Scenarios F0, F1, C0 and C1, there would be increased dieback of floodplain habitat. Some plant communities that thrive at intermediate salinities could still do well under F1 and C1, when dry season minimum flows are met. Overall, these scenarios could result in a change in community composition from freshwater brackish wetlands to halophytic floodplain and saltmarsh, and from sedge pans to open saline pans. Health of the vegetation community is expected to decline even further under the future development scenario (down to 42.0%, Table X) and further still if climate change projections are factored in (down to 34.9%). Respecting the summer low flow EWRs does little to alleviate this trend (score change by no more than 1%).

Table X. Summary of projected changes in species richness, biomass and community composition of riparian vegetation under the various future flow scenarios (Hist. = Historical, P0 = Status Quo, EWR = Environmental Water Requirements , FD = Future Development, CC = Climate Change).

	Hist. 2010	PD 2020	P1	F0	F1	C0	C1
			PD+EWR	FD	FD+EWR	FD+CC	FD+CC+ Low EWR
Species richness	80.0	71.8	72.1	68.7	68.8	48.2	47.4
Abundance/ Biomass	54.0	45.7	45.2	42.0	43.0	34.9	35.0
Community composition	60.0	53.8	54.1	51.5	51.6	36.2	35.5
Health score	54.0	45.7	45.2	42.0	43.0	34.9	35.0

Invertebrates

Invertebrates that inhabit estuaries can be divided into several groups based on where they reside in the estuary. Zooplankton live mostly in the water column, benthic organisms live on and in the sediments on the bottom and sides of the estuary channel, and hyperbenthic organisms live just above the sediment surface.

While invertebrate distribution patterns are shaped by many physical characteristics, **salinity** is the most important factor. Invertebrates that are able to survive in estuaries are broadly divided into marine species, generally found in the lower reaches of the estuary; brackish water or true estuarine species found in the middle reaches of estuaries; and freshwater species which are usually restricted to the low salinity upper reaches. Estuarine invertebrate assemblages are known to shift up or down an estuary in response to shifts in environmental conditions, particularly freshwater inflow. Sediment particle size is also an important indicator of invertebrate community structure and biomass.

Under reference conditions, the system was dominated by freshwater flows and the estuary experienced lower salinity levels. As a result, biomass of the **zooplankton** and **subtidal benthos** under natural conditions was probably 30-40% lower than under present day conditions. However, **intertidal** invertebrate biomass was significantly higher under reference conditions because of the greater availability of habitat, particularly in the lower reaches of the estuary. Much of this area has been altered or lost through construction and the expansion of salt pans. The frequency of flooding and the persistent inundation of the floodplain with freshwater would have maintained relatively low salinities in the intertidal areas, with distinct invertebrate communities occupying these habitats. In addition, the persistence of reed beds in the lower estuary under reference conditions provided additional habitat for carid shrimps (DWA 2010). These reed beds and the habitat they provided have since disappeared from the lower estuary.

Sediment characteristics in the lower estuary have changed from reference conditions as a result of repositioning and permanent fixing of the estuary mouth. Under present-day mouth conditions, tidal currents are stronger, and result in courser sediments in the lower estuary. Under reference conditions, the choked mouth during summer would have resulted in finer sediments in the lower estuary. This would have led to a complete switch in species composition.

Projected changes in species richness, biomass and community composition of invertebrates under the various future flow scenarios are summarised in Table XI. Projected changes in the health status of the invertebrate community of the Berg River Estuary are similar to those for the

microalgae and macrophytes, with a drop evident between 2010 and 2020 (down from 54 to 49.1%). A further modest decline occurs under the future development scenario (F0 down to 47.1%), and a marked decline when the effects of climate change are incorporated (31.3% under C0). Respecting low flow EWRs makes a very small improvement.

Table XI. Summary of projected changes in species richness, biomass and community composition of invertebrates under the various future flow scenarios (Hist. = Historical, P0 = Status Quo, EWR = Environmental Water Requirements , FD = Future Development, CC = Climate Change).

	Hist. 2010	PD 2020	P1	F0	F1	C0	C1
			PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
Species richness	80.0	72.8	73.4	69.8	69.8	46.3	45.7
Abundance/ Biomass	54.0	49.1	49.5	47.1	47.1	31.3	30.9
Community composition	60.0	54.6	55.0	52.4	52.4	34.8	34.3
Health score	54.0	49.1	49.5	47.1	47.1	31.3	30.9

Fish

Estuaries provide a sheltered, shallow, inshore habitat for fish, and are important nursery areas for young fish. The capacity of estuaries to function as nursery areas is dependent on the condition of their habitats and their fish stocks. These, in turn, are dependent on the quantity and quality of freshwater inflows, the management of habitats, and fishing pressure.

The Berg River Estuary is one of only three permanently open systems on the west coast of South Africa. It is therefore of particular importance to fish that utilize these habitats for their lifecycles. 17 out of the 35 fish species (48%) recorded from the Berg River Estuary can be regarded as either partially or completely dependent on the estuary for their survival.

The basline monitoring study analysed all available quantitative data on the fish fauna from 1992 to 2006 (14 surveys). We compared those findings to the results of a recent (February 2020) survey that sampled the same sites on the Berg River Estuary using the same methods (Lamberth, unpublished data). The 2020 survey recorded 68 154 fish from 22 species, comparable to the diversity of catches in historical summer surveys, and close to the average catch and abundance of previous surveys. The same six species have dominated catches in all surveys, together contributing 93-99% of the catches in each year (Table XII).

The fish fauna appear to have had a limited response to changing freshwater flows. It is likely that more dramatic changes in abundance, community composition and longitudinal distribution occurred during the peak of the 2015-2018 drought, but if so, the fish community appears to have recovered by 2020, suggesting a high level of resilience. This does **not** imply that further reductions in freshwater flow could not have significant negative impacts on the estuary fish fauna. Indeed it is critical that there are sufficient freshwater flows to maintain a river-estuary interface zone required by most estuarine dependent species and that there is sufficient tidal intrusion of marine water to maintain suitable water quality.

Table XII. Total catch, diversity and percentage composition of dominant species from historical (1992-2006) and recent (February 2020) surveys of the Berg River Estuary fish fauna. Source: Clark *et*

al. **2009.**

	Range (1992-2006)	Average (1992-2006)	2020
Number of species	14 - 23	17	22
Total catch	20 402 - 243 226	62 952	68 154
Abundance (ind.m^{-2})	2 - 26	7	6.7
Composition			
Chelon richardsonii (%)	25 - 85	59	29
Gilchristella aestuaria (%)	3 - 58	22	58
Atherina breviceps (%)	0 - 25	8	4
Psammogobius knysnaensis (%)	0 - 8	3	0
Caffrogobius nudiceps (%)	0 - 6	2	6
Oreochromis mossambicus (%)	0 - 14	2	2

Based on the above, the health score of the health of the fish community was reassessed, and we arrived at a higher score for 2020 than was given in 2010 (67% vs 56% similar to natural). This is due to the reassessment of a suite of species classed as having unsustainably small populations in the previous survey. The health of the fish community is not expected to change much under future development, or when summer low flow EWRs are respected. However, fish community health is projected to decline sharply under the influence of climate change (score drops to 50%) due to declines in estuarine residents and the loss of freshwater and catadromous species with reduced freshwater inputs (Table XIII).

Table XIII. Changes in health scores for fish under the various flow scenarios (Hist. = Historical, P0 = Status Quo, EWR = Environmental Water Requirements , FD = Future Development, CC = Climate Change).

	Hist. 2010	PD 2020	P1	F0	F1	C0	C1
			PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
Species richness	56.0	66.8	66.8	66.8	66.8	50.1	50.1
Abundance/ Biomass	85.0	68.1	96.4	88.5	96.4	71.9	67.6
Community composition	86.7	88	86.3	85.8	86.3	85.5	85.5
Health score	56.0	66.8	66.8	66.8	66.8	50.1	50.1

Birds

The Berg River Estuary provides extensive and varied habitat for a wide variety of waterbirds, including a number of threatened species. The estuary is managed as an Important Bird and Biodiversity Area by BirdLife International, and is under consideration for being assigned Ramsar status as a wetland of international importance.

Increasing demands on water for domestic, industrial and agricultural use have resulted in reduced freshwater input into the lower floodplain wetlands and estuary. This has altered habitats and food resources for waterbirds. This section of the report summarises available information on the waterbird fauna of the Berg River Estuary based on a desktop review of the literature and analysis of bird counts made under the Co-ordinated Waterbird Counts (CWAC) monitoring programme, conducted since 1994.

Between 1994 and 2006 species richness remained relatively stable. However, since 2007 the average number of waterbird species recorded on the estuary has declined. There has been an exponential decline in numbers of waterbirds since 1994 (Figure IV). Bird numbers in recent counts are only 34% of the numbers recorded in the baseline count of 1994.

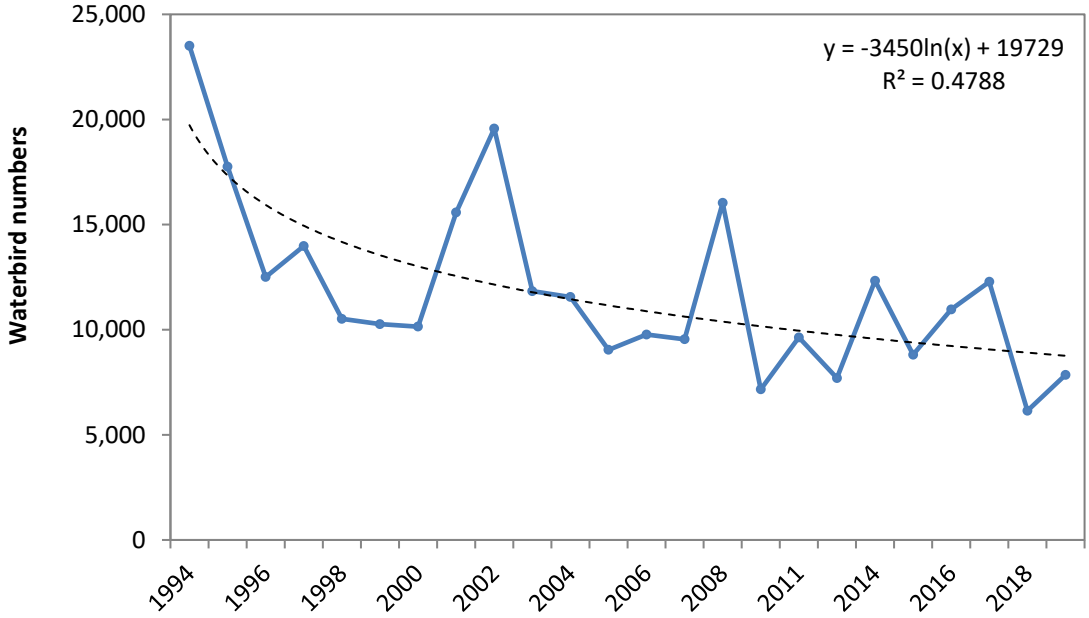


Figure IV. Changes in waterbird bird abundance at the Berg River Estuary 1994-2019 (excluding marine cormorants). Source: CWAC data.

To better understand this decline, we examined bird numbers by functional group. There has been a drastic downward trend in the abundance of Palearctic waders since 1994, especially in the last ten years. Factors outside of the Western Cape are at least partially responsible for the observed trends, and probably partly reflect global population declines (Ryan 2012). However, there has also been a concerning declining trend in the number of resident waders, with recent counts lingering at ~60% of the 1990s average. This suggests that the conditions at the Berg River Estuary are at least partially to blame for the decline in both migrant and resident waders. The most likely problems include loss or changes in feeding habitat with their associated invertebrate

fauna. Human disturbance may be another important factor, as many important feeding areas are intensively used for recreation.

Herbivorous waterfowl occur in low salinity or freshwater habitats and are associated with the presence of aquatic plants. This group of waterbirds has also shown an exponential decrease in numbers. This is expected, given their low tolerance of saline waters and their association with freshwater aquatic plants, which would have died off during the hypersaline conditions associated with the drought.

Conversely, there has been a steady increase in the numbers of piscivorous waterfowl, birds of prey, kingfisher, cormorant and darter birds on the estuary over time. Numbers of these species spiked in years with particularly low flows, suggesting that a more marine-dominated system favours these birds. Another factor is that during dry years, a number of freshwater wetlands and dams dry out, meaning the estuary could function as an important refuge for these species during excessively dry years.

The overall health of the bird community has declined markedly in the last 10 years, having dropped from 82% in 2010 to 56% today (Table XIV). This is attributed primarily to a reduction in the abundance of waders and herbivorous waterfowl due to habitat loss, human disturbance, changes in flood regimes, and reductions in global breeding populations of migratory species. Persistent low freshwater inflows and an increase in marine dominance of the system is likely to lead to a change in bird community structure. It is expected that piscivorous groups may increase in numbers as a result of more marine fish in the estuary and the use of the estuary by these species during droughts. Additional reductions in freshwater flows into the estuary may result in further decreases in the number of herbivorous and omnivorous waterfowl, or the complete loss of certain saline intolerant species. Other species likely to be affected include the rallids and herons. Lower freshwater inflows will likely see the (complete) loss of reed beds along the banks of the estuary which are favoured by these species.

Conditions are expected to improve very slightly if summer low flow RQOs are respected (P1 56.7%) but will remain largely unchanged (56.2%) under the future development scenarios if summer low flow RQOs are not respected (F0). Again, conditions are expected to improve slightly (56.6%) if the summer low flow RQOs are respected (F1). Further declines are anticipated when impacts of climate change are included (C0 49.6%), and respecting summer low flow RQOs (C1) only slightly improves the score (50.5%).

Table XIV. Summary of projected changes in species richness, biomass and community composition of birds under the various future flow scenarios (Hist. = Historical, P0 = Status Quo, EWR = Environmental Water Requirements , FD = Future Development, CC = Climate Change).

	Hist. 2010	PD 2020	P1	F0	F1	C0	C1
			PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
Species richness	82	78.9	78.9	75.7	78.9	63.3	63.3
Abundance/ Biomass	82	56.3	56.7	56.2	56.6	49.6	50.5
Community composition	84	76.5	76.6	75.9	76.4	65.6	66.1
Health score	**82**	**56.3**	**56.7**	**56.2**	**56.6**	**49.6**	**50.5**

Economic and social value

Introduction

Estuaries are rich and productive systems that produce a wide range of benefits to society. Understanding the value of conserving estuaries, or at least managing them for sustainable use, has become vital for managers and policy makers who are increasingly being faced with difficult decisions regarding water extraction and development of surrounding coastal areas. In particular, there is a need to evaluate trade offs in allocating water to the Berg River Estuary versus to other uses, such as in agriculture, industry and urban areas, and to understand how changes in flow to the estuary affect the benefits that are derived from the estuary system. These benefits are discussed below:

Subsistence Fishing

The Berg River Estuary supports a number of subsistence fishers – people who fish or collect bait personally, use low technology gear, live near to the resource and either use the catches to meet basic food requirements or sell the catches locally to gain income to allow them to meet basic food requirements. Turpie & Clark, Cape Nature Report (2007) estimated the annual catches and values for subsistence fisheries in South African estuaries using data collected as part of the Subsistence Fisheries Task Group assessment (Clark *et al.* 2002), and Hutchings *et al.* (2008). These studies suggest the value of the subsistence fishery is in the order of **R385 500 – R1.2 million** per year (in 2019 Rands), which might even have been enhanced during the drought due to more marine conditions.

Salt production

Salt production is practiced in three places on the south banks of the Berg River Estuary in areas reclaimed from former saltmarsh. The salt pans attract numerous important waterbirds, which contribute to the attractiveness of the estuary as a birding destination. Three production sites on the estuary have the capacity to produce approximately 55 000 tonnes of salt per year, although operations are currently in state of transition. The value of this production is estimated to be in the order of R8.3 million per year.

Floodplain farming

Thirteen farms along the estuary graze cattle and sheep in the floodplain. Each year the grazing capacity of the floodplain is assessed, and over the last two decades it has decreased significantly. This is due to decreases in the extent and duration of floods, which is the result of both changing climatic conditions and abstractions upstream that have reduced flow into the estuary. Over the last two decades livestock farmers along the estuary have reduced their stock numbers by 30-60%. During the recent drought the floodplain was not suitable for grazing and farmers were forced to offload about a third of their animals and bring in feed for the remainder. It was estimated that the floodplain contributes some **R11.5 million per year** to direct value added in the agricultural sector.

Tourism value

The Berg River Estuary is the dominant feature of the town of Velddrif, which is a popular West Coast tourist destination. Tourism is recognised as one of the major contributors to the regional

economy. Velddrif is a popular destination for domestic tourists to enjoy estuary-based activities, but has more recently experienced a growing regional and international tourist market attracted by cultural, heritage and nature-based experiences. The tourism industry in Velddrif/Laaiplek has increased significantly over the last few years. The estuary is considered the most important attraction for visitors, even more so than the coast.

A 2010 study estimated the tourism and recreational value of the Berg River Estuary based on a questionnaire survey. They found that the estuary contributed more than one third (35%) of local tourism value. Relaxing, walking and swimming were the most important activities carried out on the estuary (35%) followed by fishing (19%), bird watching (15%) and boating (14%). Permanent residents spent an average of 143 days visiting the estuary, while holiday homeowners and visitors visited for an average of 62 and 10 days per year, respectively. The study estimated that the total expenditure by holiday homeowners and visitors to Velddrif was R88 million per annum, with **R31 million** of this being attributable to the estuary (converted to 2019 Rands). The recent Classification and RQO study (2017) arrived at a similar estimate **R36 million per year** (in 2019 Rands) using a combination of tourism data, densities of geotagged photographs, and land cover data.

At the height of the Western Cape drought (~2017) visitor numbers to Velddrif declined, largely in response to extreme water restrictions. Given the rise in tourist numbers over the last few years, it appears that tourism has not been significantly affected by the deterioration in estuary health over this period. This could be because resort towns that developed on the back of a natural resource can eventually become part of the attraction, and so develop some resilience to deterioration of the original attraction. It is possible that people only become sensitive to estuary deterioration when the changes become significant and noticeable.

Property value

The value that residents place on access to and views of natural ecosystems is reflected, to an extent, as premiums paid in private property and real estate markets. Much of the variation in property prices is linked to distance from the estuary. Being a small town, the distance to neighbourhood amenities such as schools, shops and health care does not have a large bearing on house prices, and people choose to purchase property based on its position in relation to the estuary. The local estate agent reported a continuous stream of interest in properties situated on the estuary. Data on characteristics and value of 4218 properties from the latest Bergriver Municipal valuation roll was analysed in conjunction with locational data. This showed a significant positive correlation between house price and distance to the estuary (Figure V). Based on a statistical analysis which controlled for factors such as house size, the total property premium attributed to estuary proximity in the Velddrif area was estimated to be **R1.9 billion** out of the total property value of R2.7 billion. More than half of this is associated with the properties of Port Owen and Admiralty Island. It is likely that the property market is not particularly sensitive to changes in estuary condition, at least until the system deteriorirates noticeably, such as having lengthened periods of poor water quality, algal blooms, bad smelling water, fish kills, excessive erosion and loss of shoreline habitat.

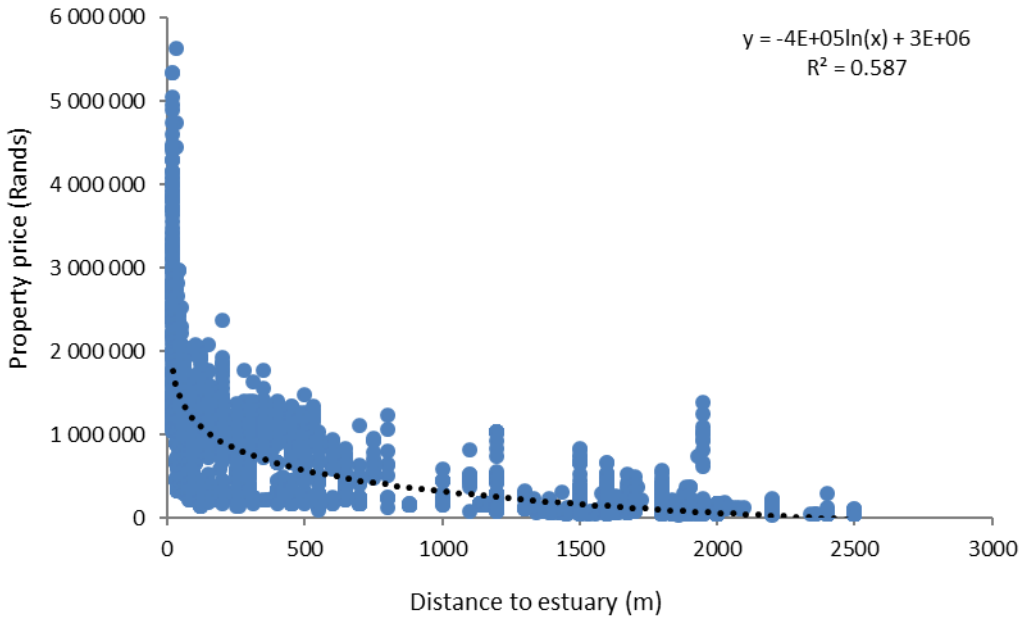


Figure V. Relationship between property price and distance to the Berg River Estuary (n=4218, 2019).

Local recreation

In addition to revenues from tourism and property values, estuaries also generate value through recreational use. Residents spend time on the estuary fishing, bird watching or swimming. They enjoy photographing or painting the various landscapes and fauna and flora of the estuary and spending time relaxing on the river banks. This type of use is an important component of the amenity value of the Berg River Estuary but is difficult to value in monetary terms. Discussions with individuals from the historically disadvantaged community revealed that access to the estuary is limited for those without boats, which has a bearing on how and when many people are able to utilise the estuary for recreation.

Nursery value

The Berg River Estuary is a crucial nursery area for recreational line fisheries, inshore commercial line and net fisheries, and inshore and estuarine subsistence fisheries along the West Coast. It contributes to the catches of about 16 marine fish species, of which harder, or southern mullet, *Chelon richardsonii*, is the most important. The small-scale commercial inshore gillnet fishery for harders in St Helena Bay is the most important fishery in the study area. Bokkoms (salted and dried harder), the most iconic food of the area, are an invaluable part of the cultural heritage in Velddrif and still make up the predominant source of protein for a majority of families. Taking into account changes in effort in the recreational and commercial line fisheries, as well as more recent data on catches in the inshore seine and gill net fishery, it was estimated that the nursery value of the Berg River Estuary is **R8.7 million**. This suggests nursery services to the value of R2 million have been lost since Lamberth & Turpie (2003) first estimated the value.

The nursery value of the estuary has been compromised by illegal gillnet fishing (Horton, NBA, 2018). The legal Berg River Estuary gillnet fishery was closed in 2001. However, monitoring has shown that illegal gillnet fishing has intensified (Horton, NBA, 2018). There has been a subsequent decline in the total reported catches by the legal commercial inshore fishery from 1500 tons of

harders in 2006 to 400-500 tons in 2018 (Horton, NBA, 2018). Thus, illegal fishing on the estuary is threatening the viability of the legal inshore fishery, with only 28 of the 84 gillnet and beach-seine right-holders in St Helena Bay remaining active in the fishery in 2019 (Horton, NBA, 2018).

Carbon sequestration and storage

Estuarine habitats are productive systems that have a high capacity to sequester carbon, and thus contribute to the regulation of both local and global climate (Barbier *et al.* 2011, Beaumont *et al.* 2014, Sidder 2018). The Berg River Estuary has 4408 ha of salt marsh and 206 ha of submerged macrophytes (Adams, NBA, 2018), which contain some 1.2 Tg C (Adams *et al.* 2019). Based on South Africa's estimated social cost of carbon of US$3.31 per ton of CO2, the share of this that will be borne by Africa, and the relative vulnerability of South Africa, it was estimated that the avoided degradation and loss of these habitats represents avoided damages of **R1.6 billion per annum** at a global scale, and South Africa's share of this would be **R12.4 million.**

Harbour facilities

The Berg River Estuary provides one of the few sheltered harbours on the West Coast. This harbour supports offshore commercial fisheries and also enhances the recreational value of the area. It has a large fish factory that processes pelagic (deep sea) fish for canning and fish meal. Without the estuary, the pelagic boats would not be able to land their fish directly at the factory and the factory would incur costs in transporting fish catches. Based on data from Hutchings *et al.* (2015) who assessed the small pelagics purse-seine fishery off South Africa, and taking into account the changes in quotas and prices, it was estimated that the Laaiplek harbour supports the wholesale production of sardine, anchovies and other industrial fish to the value of about **R107 million**, as well as being an important source of employment in the town.

Intrinsic and non-use values

Bequest and existence values, also known as non-use values, include the value of having the option to use the resources (e.g. genetic) of ecosystems in the future (option value), and the value of knowing that nature exists (existence value) and can be enjoyed by future generations (bequest value). These values can be valued using surveys to elicit peoples' willingness to pay, or based on observed donations. Although there has been an estimate of the non-use value of South African biodiversity in general (Turpie *et al.* 2003), and some attempt to estimate the existence value of estuaries (Turpie & Clark 2007), these estimates are outdated and do not fully capture the existence value of the Berg River Estuary.

The intrinsic value of biological biodiversity refers to its true, inherent and essential value – that all life forms should be accorded equal value and should be conserved because they exist and have a right to continued existence. The Berg River Estuary is one of the most valuable biodiversity assets along the South African coastline and its conservation is important to those who live near to it and visit it, as well as to those who do not, but appreciate its existence and would like to see it enjoyed by future generations. Therefore, this intangible measure is directly related to the health of the system and the need to facilitate the continued protection of all life forms in their natural habitats.

Summary of estuary values

The value of the Berg River Estuary is estimated to be in the order of **R378 million** per year (Table XV). This excludes any non-use values which have not been estimated but are expected to be substantial. The amenity or use value associated with the estuary (nature-based tourism and property values) accounts for more than half of the total value of the system. The contribution that the estuary makes to fisheries, through important nursery areas and providing a sheltered harbour, accounts for about one third of the total value.

Table XV. Summary of values associated with the Berg River Estuary in its present condition.

Benefit	Value (2019 Rands)
Subsistence fishing	1 200 000
Livestock grazing	11 500 000
Salt production	8 300 000
Tourism value	36 000 000
Property value	168 000 000
Nursery value	8 660 000
Carbon sequestration & storage	12 386 000
Harbour facilities	107 000 000
Bequest and existence value	25 000 000
Total	**378 046 000**

Potential changes in value under different flow scenarios

Changes in value under changes in estuary condition are summarised in Table XVI. Harbour facilities are unaffected by changes in flow and estuary condition and therefore the benefit remains constant across all scenarios.

The ecological condition of the estuary does not change from the status quo (P0) for Scenarios P1, F0 or F1, remaining in a D-class. Therefore, the estimated value of the benefits does not change for these scenarios. Under the climate change Scenarios C0 and C1, the overall value of the benefits considered could decline by 30% due to losses in tourism and property values, as well as losses in fishery values, carbon storage and floodplain farming. Salt production is expected to increase slightly under these two scenarios. Under scenarios P2, F2 and C2, where the estuary is returned to health category C-class, the value of the benefits could increase by more than 20%, with significant improvements in fishery values as well as small increases in tourism and property values. For comparability, these estimates are an indication of how different the values would be today, were the estuary in that condition. In future, the value of the estuary will grow, given that there will be more demand from an increased population with higher average income.

Table XVI. Estimated values of the Berg River Estuary under different scenarios. Note that the status quo is P0.

Health category	B	C	D	E
Corresponding flow scenarios	**Not feasible**	**P2, F2, C2**	**P0, P1, F0, F1**	**C0, C1**
Subsistence fishing	2	1.7	1.2	0.7
Livestock grazing	14	12.7	11.7	5.9
Salt production	8.3	8.3	8.3	10.4
Tourism and property value	293.7	280.9	204	114.1
Non-use value	50	45	25	12.5
Nursery value	14.7	12.1	8.7	4.7
Carbon sequestration & storage	14.9	13.6	12.4	6.2
Harbour facilities	107	107	107	107
Total	**504.6**	**481.3**	**378.3**	**261.5**

Synthesis and Recommendations

Understanding the value of investing in natural capital is essential to achieving policy decisions that result in maximising long-term benefits to society. Given the regional and national importance of the Berg River Estuary in terms of biodiversity and ecosystem services, there are strong calls to increase its level of protection. Understanding what values are at stake and the factors affecting those values is helpful in weighing up the costs and benefits of management decisions. This section draws together the information presented on the ecological functioning and status of the estuary, and how this will change under the different scenarios. We also introduce the option of increasing flows to the system to achieve an increase in health category. Following this, we consider the costs and benefits of supplying more water to the estuary.

Summary of the Present Ecological Status of the Berg River Estuary

The latest estuary health score produced by this study is significantly lower than that of 2010 (Table XVII). Understanding of the system hydrology and functioning has greatly improved since the 2010 assessment. The lower score for 2020 is therefore due both to a reassessment of historical hydrology as well as continued reduction of the quantity and quality of inflows. In addition, direct pressures on the estuary have increased.

MAR has declined since 2010, while irrigation demands have increased. The long-term reduction in freshwater flows was exacerbated by the unprecedented 2015-2018 drought with dramatic consequences for water quality. This included extended periods (18 months) of elevated salinity, which sometimes led to a reverse salinity profile. Nevertheless, the permanently open mouth and substantial tidal exchange ensured enough flushing to maintain water quality suitable for most marine and estuarine biota, even during the drought. Overall, The Berg River Estuary is far more marine dominated than under the Reference condition, and more so now than it was in 2010. Less salinity-tolerant biota have retreated up the system, often suffering further mortalities during the recent drought. Numbers of birds, the only biotic group that has continued to be monitored, have continued to decrease across several groups.

Table XVII. Health scores and corresponding ecological condition under historical (2010) and present day (2020) conditions.

Variable	Weight	Historical 2010	Present Day 2020
Hydrology	25	72	36.3
Hydrodynamics/mouth condition	25	90	60.8
Water quality	25	40	31.1
Physical habitat alteration	25	71	66.0
Habitat health score	50	**68**	**48.6**
Microalgae	20	75	68.3
Macrophytes	20	54	45.7
Invertebrates	20	54	49.1
Fish	20	56	66.8
Birds	20	82	56.3
Biotic health score	50	64	57.2
Estuarine Health Index Score		66.3	52.9
Ecological Reserve Category (ERC)		C	D

Since 2010, the overall health of the estuary has changed from 66.3% similarity to Reference condition to 52.9%, and has dropped a whole category – from a "C" (moderate modified) to a "D" category (largely modified)[1]. **The system is thus no longer compliant with the gazetted RQOs** which require maintaining the system in a C category. This change in health has largely been driven by reductions in physical health (down from 68% to 49%), with changes in biotic health lagging somewhat (down from 66% to 57%).

Impacts of alternative flow scenarios on estuary health

This study modelled the effects of (a) ignoring estuary water requirements, so that the quantity of flows to the estuary are effectively what is left over from water abstractions in the catchment, (b) meeting the recently-gazetted EWR requirements, and (c) allocating more water than the gazetted requirement, specifically to achieve a significant improvement in health of the estuary. These environmental flow options were considered under conditions of present-day water demand and supply infrastructure, as well as under planned future water infrastructure developments. Future scenarios were furthermore predicted with and without accounting for predicted climate changes.

Anticipated changes in health under a range of scenarios are presented in Table XVIII. Under a future development context without climate change (F0), MAR decreases from 50 to 47% of Reference (natural) flows, and the overall health of the system would be expected to decrease from 52.9 to 51.1% of Reference conditions. Respecting the low flow EWRs yields is not expected to yield much benefit in overall estuary health in future (overall health score rises to 52.6%). If expected climate change is considered, the estuarine health score drops even further (39.8% for

1 Based on the Estuary Health Index, Turpie *et al.* (2012).

both C0 and C1) and tips the estuary into an E-class. Note that the scenarios involving increasing overall water allocations to the estuary were not considered here, as their health was predetermined to be in a C-category.

Table XVIII. Health scores and corresponding ecological condition under present day conditions and for the different runoff scenarios.

Variable	Weight	Hist. 2010	P0 2020	P1	F0	F1	C0	C1
				PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
Hydrology	25	72.0	36.3	38.0	35.5	40.8	22.9	28.7
Hydrodynamics/ mouth condition	25	90.0	60.8	60.8	58.9	61.3	46.4	46.8
Water quality	25	40.2	31.1	32.5	28.7	30.8	21.5	22.1
Physical habitat alteration	25	71.0	66.0	66.0	66.0	66.0	59.0	59.0
Habitat health score	50	68.3	48.6	49.3	47.3	49.7	37.5	38.0
Microalgae	20	75.0	68.3	67.5	62.7	64.2	44.5	42.9
Macrophytes	20	54.0	45.7	45.2	42.0	43.0	34.9	35.0
Invertebrates	20	54.0	49.1	49.5	47.1	47.1	31.3	30.9
Fish	20	56.0	66.8	66.8	66.8	66.8	50.1	50.1
Birds	20	82.0	56.3	56.7	56.2	56.6	49.6	50.5
Biotic health score	50	64.2	57.2	57.1	55.0	55.5	42.1	41.6
Estuarine Health Index Score		**66.3**	**52.9**	**53.2**	**51.1**	**52.6**	**39.8**	**39.8**
Ecological Reserve Category (ERC)		**C**	**D**	**D**	**D**	**D**	**E**	**E**

Balancing Estuary EWRs with water availability in the WCWSS

Our results indicate that providing for the minimum flow requirements (i.e. 0.6 m^3/s) has very little impact on the available water yield from the system under both the current and future infrastructure scenarios. This is because the actual volume of water required is relatively small, and could relatively easily be ensured through improved operation of the system. Under the future climate change scenario, even providing the minimum flow requirements will impact on the available yield from the system, although this only results in a 2% reduction in the historical firm yield (HFY). The biggest impact on estuary health is in terms of the winter flow requirements. Under the current climate scenario, there is only a 7% reduction in the average volume relative to present day. However, under future climate change the average volume of winter water is reduced by up to 37%.

Improving estuary health and value involves securing more and/or better quality water than is reaching the estuary currently. This will come at some cost to society. If flows are to be increased to the estuary, this means that either there will be a lower firm yield to allocate to existing users, or that further investment will be needed to improve efficiency and/or maintain water supply by other means. Potential options to free up water for the estuary are summarised in Table XIX.

Table XIX. The most feasible options to free up more water for the Berg River Estuary.

Type	Options	Comment	Potential	Cost (R/m³)
Demand side	Urban and industrial demand management through pricing	Proven and feasible	50 Mm³ p.a.	0
	Agricultural demand management or curtailment	Options largely exhausted, low feasibility	Negligible potential	n/a
Supply side	Removal of invasive alien plants	Cheapest supply option	55 Mm³ p.a.	R6.76
	Conventional infrastructure and recycling	Options already planned in	Limited further potential	n/a
	Desalination	Unlimited	Balance of requirements	R8.82

On the demand side, consumption could be reduced through measures to improve efficiency, usually incentivised through increased water pricing. The recent drought experience in Cape Town has demonstrated how urban and industrial demand can be reduced with a combination of restrictions and pricing. There is probably less opportunity for demand management in agricultural areas, where efficiency gains have already been high. A more extreme option is curtailment, where the allocations to certain user groups such as farmers are reduced. This will have the impact of reducing economic outputs and employment, and as such, is likely to be politically infeasible.

On the supply side, more water could be made available through the removal of invasive alien plants (IAPs) in catchment areas, increasing conventional infrastructure such as water supply dams, and developing alternative sources such as groundwater, recycling of waste water and desalination of sea water. The conventional and recycling options have already largely been exhausted in the current plans. Thus, the main opportunities lie in clearing IAPs, recycling and desalination. A recent study has established that an investment of R372 million in IAP clearing will generate annual water gains of over 55 Mm³ a year within five years and that it is the lowest cost of the alternative supply options (Turpie *et al.* 2019). Desalination is currently the most expensive option for augmenting supply, and is considered a last resort, though the technology has become more efficient and cost-effective in recent years.

Costs and benefits of securing environmental flows

In general, supplying a higher proportion of the Reference level flows to the estuary, in line with seasonal requirements, results in a healthier and more valuable estuary. We calculated a marginal value in the order of R673 per ML at present. The marginal value of water inputs to the estuary will decrease with increasing inputs, however. Based on the above options, we provide a range of estimates of the costs of meeting water supply (Table XX). Depending on how many interventions are used to tackle the problem, the cost of supplying the additional water needs to the estuary range from about R4.70 to R8.80 per m³.

Finally, the estimated changes in value of the estuary are compared with the estimated additional costs of water supply in Table XX. From this it is evident that, while increasing flows to improve the condition of the estuary to a C-category will have a measurable impact on yields, the value gains could outweigh the costs, especially if efforts are made to restore catchment areas and curb urban demands through more appropriate pricing strategies.

Table XX. Summary of the scenario water requirements, health, and value, compared with the associated water opportunity costs. The latter are based on three alternative policy scenarios: L includes demand management, IAP clearing and desalination, M includes the last two and H is based on desalination alone.

Scenario	MAR (Mm3/a)	Health category	Firm Yield (Mm3)	Difference in yield from planned (Mm^3)	Asset value (Rm)	Difference in asset value (Rm)	Net benefit of supplying full C-class EWRs (Range, in R m NPV)
Reference	912	A	-		-		
P0	459	D	508		5 376		
P1: PD + EWR	**469**	**D**	**508**		**5 376**		
P2: PD + C class	593	C	374	-134	6 839	1464	283 to 837
F0: Future, no EWR	433	D	528		6 386		
F1: Future + EWR	**438**	**D**	**528**		**6 386**		
F2: Future + C class	593	C	322	-206	8 125	1739	-76 to 478
C0: Future + CC	303	E	480		4 414		
C1: Future + CC + EWR	**312**	**E**	**470**		**4 414**		
C2: Future + CC + C class	593	C	205	-265	8 125	3710	1370 to 1925

Water trade-offs can be eased by improving water quality

It is very important to note that this study has only considered changes in freshwater inflows to the estuary. However, the quantity of inflows required to achieve a certain class of health can be lowered if measures are taken to restore the quality of inflowing water. If water quality were improved, then then the percentage of MAR required to reach a C-class would decrease.

Conclusions and recommendations

The conclusions that can be drawn from the overall results are:

- **The Berg River Estuary is a highly valuable asset**. Not only is the estuary of key biodiversity importance, it also produces goods and services worth at least R370 million per annum. The asset value of the Berg River Estuary is currently just over R5 billion.

- **As at 2020, the estuary's health is in a "largely modified" (D) category**. This is due to (a) a reassessment of the scores with better information, and (b) an actual decrease in health due to continued decreases in flows and water quality and other factors.

- **The gazetted environmental water requirements (EWRs) are not being met at present**. While winter flows largely meet the requirements, summer flows fall considerably short. In addition, water quality falls well short of requirements.

- **Meeting the EWR requirements as gazetted will not be sufficient to achieve a "moderately modified" (C) category, even under historical ("present-day") climate conditions**. The estuary will require more and/or better quality water than currently gazetted in order to

be restored to the desired C-category of health that befits it's biodiversity conservation and socio-economic importance.

- **Without increasing the EWR allocation, climate change will reduce health to a "severely modified" (E) category**. This could lead to the estuary losing about 30% of its value. What is required is a higher EWR allocation.

- **A "moderately modified" (C) category estuary will require more water than previously estimated.** Based on this study, it is estimated that 65% of MAR would be needed to achieve a C category. Restoring water quality could lower the flow quantity requirements by as much as 30%, but the relative costs of these options have not been explored here.

- **Securing more water to increase estuary health can be justified**. Supplying enough water to the estuary to restore it to a C category would currently cost an estimated R0.6-R1.2 billion in water demand and supply measures, but this would increase the ecosystem asset value by an estimated R1.5 billion. Under climate change, system yields would be reduced by 56%, and their replacement costs would amount to between R1.7 and R2.3 billion. This is less than the estimated difference in value of the estuary of around R3.7 billion. Given that the full value of the estuary goes beyond economic outputs and can never be fully quantified, these results suggest that as much as possible should be done to free up water for the estuary to allow it to recover to and remain in a C-category, as befitting its biodiversity importance and highly-regarded sense of place.

The following recommendations are made:

- Establish a rated section at the head of the estuary for **monitoring freshwater inflow** so that the actual amount of flow reaching the estuary can be recorded; given the low flows in summer, a weir is not advised as this will impact negatively on the system;

- **Update the gazetted flow requirements** to restore estuary from "largely modified" D to a "moderately modified" C-category by (a) increasing MAR to at least 55% of natural + (b) halving anthropogenic nutrient inputs from the catchment;

- Reinforce and support measures to **reduce anthropogenic nutrient inputs** from the catchment and estuary shores, through a range of measures including riparian buffers, control of agricultural return flows, higher treatment standards and polishing of treated wastewater;

- Implement measures to **increase water flows from the catchment** and thereby minimise the cost of water supply to the estuary, including clearing of invasive alien plants, better compliance monitoring of water users;

- Reinforce and **support implementation of the estuary management plan**, including:

    - conservation planning to avoid excessive river bank development and protect sensitive areas;
    - securing formal protection;
    - visible patrols and better enforcement of estuary activities, particularly fishing; and
    - raising public awareness of the value of conserving the system.

1 INTRODUCTION

1.1 Background and context

The Berg River Estuary is one of the largest estuaries in South Africa, covering an area of 61 km^2, and encompassing about 60% of the estuarine habitat on the West Coast. The estuary supports large areas of salt marsh, and high numbers and diversity of water birds and fish, making it one of the most important estuaries in the country in terms of its biodiversity (Turpie *et al.* 2002, Turpie & Clark 2007). It is also an extremely important nursery area for angling species (elf, white steenbras, leervis) and other commercially important fish species (harders) and provides a range of other ecosystem goods and services that support the local economy.

The delivery of ecosystem goods and services from the Berg River Estuary is contingent on two broad sets of policy decisions: (1) adequate quantities and quality of freshwater inflows in relation to seasonal requirements, and (2) management of activities in and around the estuary itself. The freshwater flow requirements are gazetted as Resource Quality Objectives following the Classification of the system. In 2008, an RDM study put the estuary in a C-class of health (on a scale of A (best) to F (worst). The Classification study of 2018 determined that the estuary should be retained in a C-class, but flow requirements were only gazetted for the dry season. The *in situ* management of the system is guided by the Berg River Estuary Management Plan (EMP), updated in 2019, but this has not yet been fully implemented.

There are concerns that the Berg River Estuary still faces significant threats to its biodiversity, sense of place and value. Critically, the gazetted freshwater requirements were set on the basis of historical climate conditions and may not achieve the intended C-class under climate change. Furthermore, it is based on an outdated body of work, with little research having been carried out over the last 15 years, which includes the most intense drought ever recorded.

Much of the work that has been done in the past occurred during the period 1993-2008, in response to the planned construction of the Berg River Dam. These studies included:

- an environmental impact assessment for the dam in the early 1990s;
- the Berg River Baseline Monitoring Study undertaken by Anchor for the Department of Water and Sanitation (DWS) in 2002-2005;
- the RDM study for the Berg River system also undertaken by Anchor for DWS in 2009/10 to determine the health of the estuary and to make recommendations on environmental flow requirements; and
- the Berg River Estuary Management Plan prepared by Anchor for the C.A.P.E. Estuaries Programme in 2008/09.

More recently, the DWS commissioned the "*Determination of Water Resources Classes and Resource Quality Objectives in the Berg Catchment*" completed by Aurecon and Anchor in 2017/18. This drew heavily on the above studies. In addition, some relevant studies have been carried out on flows, water quality and alien clearing (e.g. Cullis *et al.* 2019, Turpie *et al.* 2019, data collected by DEA&DP & DWS). The City of Cape Town has also developed an updated water strategy (City of Cape Town, 2019). Also, the EMP was updated by the Western Cape Government in 2019.

The Resource Quality Objectives (RQOs) for the estuary were finalised in 2019. The gazetted environmental water requirements (EWR) stipulate a minimum flow of 0.6 m^3/s of flow entering the estuary to prevent the formation of a hyper-saline wedge developing and persisting in the estuary; a certain proportion of the natural monthly flow requirements during the winter months; and flood requirements that are necessary for habitat maintenance and for periodic scouring of the estuary and flooding of riparian vegetation. As well as stipulating the flow requirements, the RQOs outline a range of standards for the biota of the system (Appendix 1).

In spite of the considerable amount of work done on the Berg River and estuary systems, gaps in knowledge remain on the hydrology, water availability and functioning of the system, and there has been relatively little in the way of monitoring and analysis of changes to the estuary since 2005. In the interim, the system has also undergone a major drought of unprecedented intensity which has impacted on water availability in the system.

This study was commissioned by Department of Environmental Affairs and Development Planning (DEA&DP) in order to update understanding of the estuary system and the implications of its protection for biodiversity and as a socio-economic asset. The project is aligned to the Western Cape Estuary Management Programme which has prioritised the development and implementation of the Berg River Estuary Management Plan in partnership with Cape Nature and the Bergrivier Local Municipality.

1.2 Aim of the study

The aim of the study was to provide an updated understanding of the ecological functioning, intrinsic, cultural and socio-economic value of the Berg River Estuary and the potential costs of maintaining or enhancing these benefits through protection of habitat and environmental flows, taking the socio-economic and climatic context of the region into account.

1.3 Structure of the report

The rest of the report is structured as follows:

Chapter 2 (STUDY APPROACH) outlines the overall approach and describes the scenarios evaluated in the study and their rationale, as well as outlining the frameworks for assessing the health and value of the estuary.

Chapter 3 (HYDROLOGY) provides a description of the catchment and its water supply system, and the hydrology of the estuary. It compares present-day hydrology with the reference (natural) hydrology and provides metrics of the health of the system hydrology and system yield under the alternative scenarios.

Chapter 4 (HYDRODYNAMICS & SEDIMENTS) provides an overview of interrelationships between floods, floodplain, tidal forces and sediment and how these have been affected by changing hydrology and other anthropogenic interventions. It compares the present-day patterns with the reference (natural) state and provides metrics of the relative health of the system hydrodynamics and physical habitats under the alternative scenarios

Chapter 5 (WATER QUALITY) provides a description of the salinity, nutrient and dissolved oxygen concentration and levels of pathogens in the estuary and their relationship to flows, based on earlier studies and analysis of data collected from 2013-2019. Water quality is scored in terms of

similarity to natural conditions under present day activities as well as under the alternative scenarios.

Chapter 6 (BIODIVERSITY) provides descriptions of the vegetation, phytoplankton, invertebrate, fish and bird communities and their sensitivities to freshwater flows and habitat, with emphasis on the fish and birds, and estimates the implications of scenarios for biodiversity, highlighting links to human activities.

Chapter 7 (SOCIO-ECONOMICS) describes the uses and values of the estuary, in terms of key ecosystem services and activities.

Chapter 8 (SYNTHESIS) brings together the above elements, updates the health assessment of the estuary and examines the potential trade-offs between environmental flows to the estuary and water supply to other users.

2 STUDY APPROACH

2.1 Updated baseline description

The project commenced with a compilation of available data, information and literature on the Berg River Estuary and related areas, processes and sectors. The catchment hydrology and functioning of the estuary system were described, with a focus on extending current understanding to include the recent drought. To this end, estuary data collected since 2005 (water quality and birds) were analysed along with an extended rainfall and hydrology time series.

For the historical time series analysis, the observed flows at Misverstand Dam were considered at both a monthly and daily value and routed to the estuary taking into account estimated irrigation demands based on an assessment of changes in irrigated areas and crop-water requirements.

In addition to the development of an updated historical daily and monthly flow time series, analysis of individual flood events was also done using real time data from Misverstand Dam. This was done to determine if there had been any observed changes in the magnitude and frequency of flood events that could have contributed to a change in the ecological health of the estuary.

Increased bank erosion has been noted as a concern in the Berg River Estuary. As part of this study a provisional assessment of identified areas of concern was undertaken and recommendations given in terms of the likely causes and recommendations for possible remedial action including the potential for both hard and soft engineering solutions including some already being trialled.

The updated description of the ecosystem was undertaken sequentially, so that once each section was completed the cumulative findings fed into the next sections. Thus the hydrology analysis was completed, followed by hydrodynamics and sediments, water quality and then biodiversity. These descriptions were used to estimate the present ecological status of the estuary. In addition, the study included comment on the need for and costs of bank stabilisation.

Understanding of the socio-economic value of the estuary was also updated, based on available information and discussions with stakeholders. The economic and water management context were also described in order to understand the economic factors when valuing the estuary.

2.2 Scenario analysis

In order to arrive at effective and efficient policy decisions, It is important for policy makers to be able to understand the factors that define the status quo, as well as the benefits and costs associated with alternative options (Loch *et al.* 2014). Ideally, this involves a thorough knowledge of the ecological, economic and social systems, ecosystem services, the demand for public goods and the likely costs and benefits. In reality, this information is often incomplete or accompanied by high levels of uncertainty. One of the best ways to inform policy makers in such a situation is through scenario analysis (Russel-Smith 2015). This has been the approach

generally used in the setting of environmental flows in South Africa and elsewhere (King & Brown 2010).

Following the description of the historical and present situation, the study then examined options for the future through a scenario analysis. This aimed to articulate the trade-offs between allocating available water and/or augmenting the system to maintain or restore the estuary health, versus for use in urban, industrial, or agricultural use, taking the reality of climate change into account.

A set of scenarios was defined in conjunction with the client and stakeholders (Table 2.1). These effectively compare the options of (a) ignoring estuary water requirements, so that the quantity of flows to the estuary are effectively what is left over from water abstractions in the catchment, (b) meeting the recently-gazetted EWR requirements for freshwater inflows, or (c) allocating more water than the gazetted requirement in order to achieve a significant improvement in health of the estuary. These hydrological flow scenarios are considered under both present-day and future infrastructure and demand. The latter included the raising of Voelvei Dam and increased abstraction from the Berg River as part of the Voelvlei Augmentation Scheme (VAS) as well as the complete re-use of treated effluent from the wastewater treatment works (WWTWs) currently discharging into the Berg River. The effects of future infrastructure were analysed for a future with minimal climate change and with the expected climate change for 2040-60, so that the relative effects of infrastructure and climate change could be discerned.

Anticipating that the combination of climate change and growing water demands would make meeting EWR requirements difficult, **the future scenario C1 is a shared-losses scenario**, in which only the dry season EWRs are met. Note that in scenarios P1, F1 and C1, the models only need to be set up to meet dry season EWRs, but the wet season EWRs were nevertheless met in P1 and F1. This is explained more in Chapter 3.

Table 2.1. Hydrological scenarios assessed in the study. P0 is the assumed status quo. The naming of the scenarios is linked to development and climate context: present context (P), future hypothetical context without climate change (F) and future predicted context with climate change (C); and to the EWRs: ignoring EWRs (0), meeting gazetted EWRs (1); or meeting flow requirements for a C-class (2; this flow is higher than gazetted).

		No EWRs.	Gazetted low flow EWRs (0.6 m³/sec)	Meeting requirements* for a C-class
Present-day WCWSS Infrastructure (PDI)	Present-day	**P0**	**P1**	**P2**
Future Infrastructure (FI) as planned	Without climate change	**F0**	**F1**	**F2**
Future Infrastructure (FI) as planned	With climate change	**C0**	**C1**	**C2**

* As updated in this study following reassessment of hydrology.

The hydrological scenarios were modelled using the latest configuration of the Water Resource Yield Model (WRYM) for the Western Cape Water Supply System (WCWSS) using stochastic analysis of the 89 years historical time series (1928 to 2016) to determine the impact on the historical firm yield (HFY) as well as a simulated time series of monthly flows downstream of Misverstand Dam which were then routed to the estuary. The future climate change scenario considered and altered time series of monthly runoff generated based on the latest climate

change scenarios for the Western Cape. Long-term monthly inflow sequences for the estuary were also generated for each scenario. These were then used to estimate the corresponding changes in hydrodynamics, water quality, biodiversity, and to derive estuary health scores and values for each scenario.

Given the constraints on the WRYM it was only possible to impose the minimum flow requirements for the estuary in terms of meeting the EWRs. This is because the EWRs refer only to the average flow requirements and therefore do not represent a fixed annual demand on the system. It was also noted that the proposed future infrastructure developments were also very unlikely to impact on in particular the high flows as these would usually occur when the system was at full capacity and all the dams were spilling and also because the proposed VAS only includes the construction of an additional diversion weir on the Berg River. Any shortfalls in meeting the average winter flow EWRs where however assessed and where appropriate consideration was given in terms of the likely impact of providing additional winter water to meet any shortfalls. This was particularly relevant for the future climate change scenarios where the average resultant winter flows were found to be significantly reduced and well below the requirements for the EWR. The likely impact of this on water availability and the associated costs with providing additional augmentation into the system where discussed in terms of the analysis of results and in the final estuary valuation.

2.3 Framework for assessing estuary health

The present status and the alternative scenarios were scored in terms of the Estuary Health Index (EHI; Turpie *et al.* 2012). This involves scoring of each of the abiotic and biotic components, to produce an overall measure of the health of the estuary as a percentage resemblance to the reference or natural condition. It is therefore a tool for determining the Present Ecological Status (PES) of an estuary. The components studied are as follows:

- Abiotic (or driving components):
  - Physical dynamics (measured in terms of seasonal river inflow patterns, floods, mouth dynamics, water level variations, water movement patterns, changes in sediments and deposition and erosion areas)
  - Water quality (measured in terms of system variables, nutrients and toxic substances); microbiological contaminants - linked to human health - are excluded as it does not pertain to the ecological component;
- Biotic (response) components:
  - Estuarine flora (microalgae and macrophytes)
  - Estuarine fauna (invertebrates, fish and birds)

The EHI is calculated as the average of the Abiotic and the Biotic score; the Abiotic score is the average of the hydrology (HL), hydrodynamics (HD), physical habitat (PH) and water quality (WQ) scores, and the biotic score is the average of the microalgae (MI), macrophyte (MA), invertebrate (I), fish (F) and bird (B) scores (Figure 2.1):

$$EHI = \frac{\left(\frac{HL, HD, PH, WQ}{4} + \frac{MI + MA + I + F + B}{5}\right)}{2}$$

All of these components are scored as percentage resemblance to the reference or natural condition, following the specific guidelines. The biotic components consider average species richness, community composition and overall abundance.

Determining the EHI score therefore involves (a) estimating what the estuary was like in its natural condition (the Reference condition) in terms of physical and biological characteristics and processes, (b) scoring the present condition of each component relative to this estimated Reference as a score out of 100, and (c) aggregating the overall score and converting the score to its Present Ecological Status category using a simple scale of A to F (Table 2.2). Scoring was done using the same models used for the previous RDM study on the estuary, with some minor modifications to models based on updated understanding.

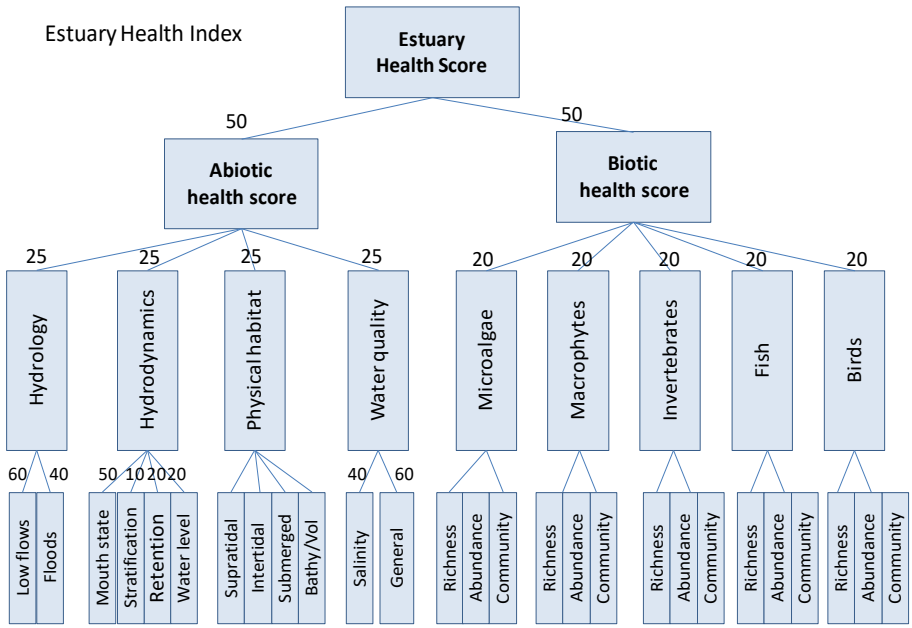


Figure 2.1. Contribution of abiotic and biotic parameters to the Estuary Health Index. Numbers indicate weighting.

Table 2.2. Estuary health rated on a scale from A-F.

Category	Health score	Description
A	91 – 100	Unmodified, natural.
B	76 – 90	Largely natural with few modifications. A small change in natural habitats and biota may have taken place but the ecosystem functions and processes are essentially unchanged.
C	61 – 75	Moderately modified. A loss and change of natural habitat and biota have occurred but the basic ecosystem functions and processes are still predominantly unchanged.
D	41 – 60	Largely modified. A large loss of natural habitat, biota and basic ecosystem functions and processes have occurred.
E	21 – 40	Seriously modified. The loss of natural habitat, biota and basic ecosystem functions and processes are extensive.
F	0 – 20	Critically/Extremely modified. Modifications have reached a critical level and the system has been modified completely with an almost complete loss of natural habitat and biota. In the worst instances the basic ecosystem functions and processes have been destroyed and the changes are irreversible.

2.4 Ecosystem services valuation framework

Until recently, ecosystems and the benefits that they provide were treated as "free" goods, not recognised for their true value. When goods do not have a value attached to them, then damage costs go unrecognised and generally decisions surrounding their use or consumption tend to favour damaging activities leading to environmental degradation and loss of biodiversity.

The understanding of ecosystem services and benefits and their valuation has advanced considerably over the past decades. In general, managers and policy makers need to understand how changes in ecosystem extent and condition affect economic outputs and human welfare. This requires understanding more about the components of biodiversity and the underlying links between ecosystem structure and function and the supply of ecosystem services. Biodiversity can be described in terms of structure and organisation, and it is this structure and organisation that determines the functioning, resilience and productivity of ecosystems (Figure 2.2). The delivery of ecosystem services and the benefits they provide depends on the condition of the ecosystem.

Each of these elements contributes to the supply of ecosystem services that contribute to human wellbeing. The productivity of a system gives rise to provisioning services and the production of wild biomass, such as fish. The functioning and resilience of a system provides regulating services such as carbon sequestration. Attributes relating to the structure and composition of a system, such as beauty, rarity and diversity, give rise to cultural services (Figure 2.2). Cultural services include the less tangible values such as spiritual, educational, cultural and recreational value which are associated with sense of place.

Since the Costanza *et al.* (1997) popularised the concept of ecosystem services, they have been classified in a number of ways. The most commonly-used classifications are those of the Millennium Ecosystem Assessment (2003) and the Common International Classification of Ecosystem Services (CICES, Haines-Young & Potschin 2013).

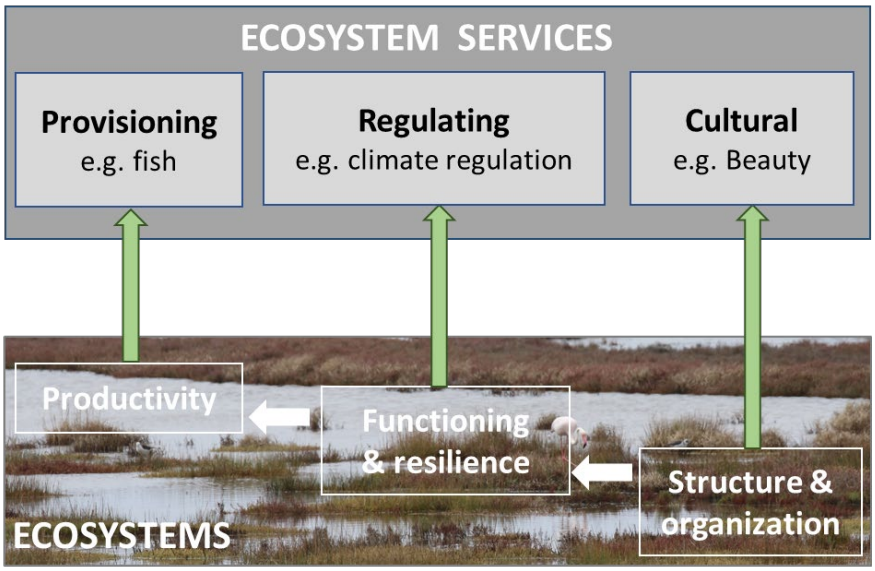


Figure 2.2. The link between ecosystems and the services they provide (adapted from Turpie *et al.* 2001).

The values associated with ecosystem services can be framed in the Total Economic Value typology, which was developed before the ecosystem services concept. This framework identifies that nature can produce direct and indirect use value, option value and non-use value. Direct use value is linked to provisioning and some cultural values such as recreation and tourism. Indirect use value is linked to regulating services, and is indirect because these services provide inputs into benefits derived beyond the ecosystem in question. In this context, non-use values would be linked to the feeling of satisfaction from knowledge of the continued existence of elements of biodiversity, without necessarily visiting the estuary. This value is closely linked to the concept of intrinsic value, which is the value of biodiversity in itself.

It is also important to realise that ecosystem values are generated through the combined use of natural and man-made and other capital. For example, if one further developed the tourism infrastructure of a location or invested in more marketing, the tourism value would be likely to increase. The valuation of ecosystem services attempts to determine how changes in natural capital affect the values derived, holding other inputs constant.

There are different ways to evaluate how changes in the environment affect human welfare. Economic analysis, such as cost-benefit analysis, considers how choices affect people using welfare measures of value. The welfare measurement of value is the sum of producer surplus (net earnings to producers) and consumer surplus (the benefits that people derive from consumption, over and above what they had to pay for it). Many policy makers are more familiar with the indicators of economic production from the national accounts, such as gross value added (GVA) and gross domestic product (GDP). These effectively estimate the total value of goods and services produced, and by extension, the total income generated to the various actors in the economy. In natural capital accounting, a developing field which is attempting to track the contribution of ecosystems to production, values of ecosystems and their services are presented in terms of "exchange value" which is compatible with national accounts (UN 2017). This does not include consumer surplus and is not a measure of welfare. However, a marginal change in GDP provides a reasonable indication of change in welfare.

The following estuary benefits are assessed in this study:

- Contribution to livelihoods through provision of living resources that are harvested for nutrition, energy and raw material purposes
- Contribution to marine fishery values through provision of nursery areas for the maintenance and productivity of marine fish populations
- Contribution to economic production in the town of Velddrif through provision of a sheltered location for harbour development
- Contribution to the amelioration of climate change damages through sequestration of carbon from the atmosphere
- Contribution to recreation, tourism and property values as a result of ecosystem attributes that lend aesthetic beauty and are attractive for recreational activities
- Contribution to sense of wellbeing through the knowledge of their contribution to the continued existence of nature and biodiversity

We also describe the production in the former saltmarsh areas that have been converted to salt pans. While the saltpans have replaced estuary habitat, they are intricately linked to the estuary. They depend on saline water pumped directly from the estuary, and they also provide an extension of the habitat supporting bird populations that use the estuary and surrounds.

2.5 Assessing estuary health – water supply trade-offs

This study considers a range of scenarios in which the quantities of river inflows is varied with different impacts on estuary health. In general, estuary health is correlated with the degree to which the quantity and quality of freshwater inflows reaching the estuary resemble the natural or Reference condition. Thus, there is an inherent trade-off between water available for use in economic sectors utilising water from the catchment, and the health and value of the estuary.

If one were to be planning for water allocation in an unstressed catchment area in which supply exceeds demand, the problem would best be analysed on the basis of the marginal value of water in different economic activities and the marginal value of the water reaching the estuary. Marginal values in all cases decrease as water is added to the activity or ecosystem. In other words, the initial inputs of water lead to a greater change in output or value than later additions of the same quantity of water. The most efficient allocation of water for the system as a whole is that where the marginal value of water is equal in all activities. This solution can be determined using optimization models, provided one can estimate the production functions for all the activities. Because water tends to be poorly priced, water allocation decisions are often made without considering the true opportunity costs to different sectors, and especially to the environment (Pulido-Velazquez *et al.* 2013). In an unstressed catchment, one can then determine which sectors (including the environment) should receive more water in order to maximise the benefits to society as a whole.

In the case of a stressed catchment, in which the demands for water exceed the supply, one could determine how water should have been allocated optimally among the competing sectors, but this can create a policy dilemma if the optimal solution requires that some sectors need to give up water to others. In South Africa, water licences are issued under strong competition and once issued, are seldom revoked. To reduce the water allocation to a sector would mean reducing water allocations to productive activities with direct and indirect consequences for employment in those sectors. While water may be transferred into more productive sectors, the labour in those sectors may not transfer as easily. Furthermore, water rights in South Africa are not tradeable, which not only precludes the economy from finding its own efficient equilibrium but precludes the environmental sector from purchasing rights from other sectors. Thus, in this study, we take the approach of considering the costs of supplying more water to the estuary in terms of the costs of achieving greater sectoral efficiency through demand management, and the costs of augmenting water supply. These measures are more feasible than water reallocation in a stressed catchment, developing country situation such as is the case for the Berg River. The analysis, including cost assumptions, are explained in more detail in the synthesis chapter.

3 HYDROLOGY

3.1 Introduction

The hydrological analysis for this study was conducted at two different temporal scales, namely daily and monthly. The daily resolution is required to assess the role of floods in the estuary dynamics while the monthly resolution is required for examining the longer-term water balance of the estuary under both current-day and future scenarios of changes in the Western Cape Water Supply System (WCWSS) regarding water use, bulk infrastructure and climate change.

Unfortunately, there is no reliable stream gauge measuring flow reaching the head of the estuary. The closest gauge is located at Misverstand Dam which is approximately 60 km upstream of the estuary. Hence it was necessary to estimate the flow to the estuary. An estimate of the historical flood volumes reaching the estuary was derived from analysis of the daily flow record at Misverstand Dam as the lowest available stream gauge on the Berg River. The observed flow at Misverstand Dam was also used to estimate the historical time series of likely flows actually reaching the estuary taking into account irrigation demands, river losses and incremental run-off downstream of Misverstand Dam. The irrigation demands were estimated based on the current DWS allocations and also using estimates of the irrigation requirements based on observed changes in irrigated areas and based on individual crop types and estimated crop water requirements.

The natural, historical and present-day hydrology of the estuary was reviewed, and a summarised account is presented of mean annual runoff (MAR), mean monthly flows, seasonality, water allocations in and abstractions from the WCWSS, impacts of agricultural dams, commercial forestry and invasive alien plants. These data are derived from the latest configuration of the Water Resources Yield Model (WRYM) of the WCWSS as used by the DWS and City of Cape Town.

A historical monthly estuary inflow time series was constructed for the period June 1974 to May 2019 using the observed flows at Misverstand Dam which represent an initial estimate of the actual observed monthly and low-flow history of the estuary. To this end, the observed monthly flows, extracted from the record for DWS gauging station closest to the estuary, G1H031 (Misverstand Dam), were adjusted by using modelled monthly historical downstream irrigation water abstractions, net evaporation and bed losses. This time series (estimated historical flows 1974-2019) was used in the evaluation of changes in the observed estuary health over a similar period in order to determine the potential impact of reduced flows to the estuary and identify critical thresholds.

In order to investigate the potential impacts of changes in flow to the estuary, a set of scenarios were constructed. These scenarios were analysed using the latest configuration of the Water Resources Yield Model (WRYM) for the Western Cape Water Supply System (WCWSS). The outputs from this analysis included a time series of natural, current and possible future flows downstream of Misverstand Dam and reaching the estuary that could then be used to determine the trade-off between water availability for the estuary and the impacts on the overall system yield. The simulated time series was based on observed precipitation data from 1928 to 2016 (i.e. 89 years).

3.2 Overview of the Berg River System

3.2.1 The Berg River System

The Berg River is situated in the Western Cape and its catchment lies between latitude 23° 45' and 33° 50' south and longitude 18° 15' and 18° 55' east. The river has its headwaters in the Jonkershoek and Franschhoek mountains and flows in a north-westerly direction through Paarl and Wellington to Misverstand Dam, continuing north through Porterville and Moorreesburg eventually discharging into the sea at Laaiplek on the West coast. The major tributaries are the Franschhoek, Wemmers, Krom, Kompagnies, Klein Berg, Vier-en-Twintig Rivieren, Matjies, Platkloof, Boesmans and Sout Rivers.

The Berg River is about 160 km long from the headwaters to the sea. The lower reaches of the river are extremely flat resulting in sea water intrusion nearly 70 km from the river mouth under high tide conditions (Bath 1989). Bath (1989) stated that the Berg River is a geologically old river system. This was based on the rapid fall in profile from the headwaters which then flattens out in the Paarl area, the degree of meandering of the main river channel, the existence of multiple channels separated by low lying islands in the lower reaches, and the great width of the river valley.

The basin of the Berg River is bounded on the eastern side by a range of mountains (Reduced Level 1500 m) including the Franschhoek, Wemmershoek, Limietberg mountains, Witzenberg and Groot Winterhoek mountains. On the western side, north of Paarl and Wellington, the basin flattens out to a hilly plain. Downstream of Paarl and Wellington, sandstone formations give way to shales, thereafter tributaries on the eastern bank of the Berg River drain areas with Table Mountain Sandstone, while the western bank drains areas with the saline Malmesbury Shale as the dominant geological formation (DWAF, 1993b).

Water use in the Berg River is primarily for irrigation of vineyards and orchards. There are also demands for afforestation in the higher-lying areas as well as riparian and upland alien vegetation infestations. There are some municipal abstractions to Paarl and Wellington, Tulbagh and Saron. There are a number of large dams and many small farm dams in the catchment as well as major water transfers from the Riviersonderend and from the Upper Breede River catchments. A significant portion of water from the Berg River is, however, supplied to the City of Cape Town as part of the allocation from the Western Cape Water Supply System (WCWSS).

Major dams located in the catchment are the recently constructed Berg River Dam (2007), Wemmershoek Dam, Voëlvlei Dam and Misverstand Dam and these are managed as part of the WCWWS which supplies water to the City of Cape Town as well as a number of other smaller municipalities and to agriculture. Water is also supplied to major industrial developments such as the Saldanha Industrial Development Zone (IDZ) via the West Coast Municipality which receives water from the WCWSS via a diversion and treatment plant at Misverstand Dam. The allocation to agriculture is largely managed through the Upper and Lower Berg River Irrigation Boards (IBs) or Water User Associations.

Environmental Water Requirements (EWRs) are included in the allocations from the WCWSS and in particular, the Berg River Dam has been specially designed to allow for environmental flow releases. This includes both a large capacity outlet in the wall of the dam as well as a multi-level tower to match water quality to water volume/level. Special operating rules have also been developed to try and best mimic natural events for EWR releases.

3.2.1.1 The Western Cape Water Supply System

The WCWSS comprises six large dams and a number of transfer schemes (Figure 3.1): the Upper and Lower Steenbras and Wemmershoek Dams owned by the City of Cape Town; the Voëlvlei and Theewaterskloof dams owned by Department of Water & Sanitation (DWS); the Berg River Dam and Supplement Scheme that is owned by the Trans Caledon Tunnel Authority (TCTA) and operated by DWS. In addition, there are a number of smaller dams and weirs including the Kogelberg and Rockview Dams that serve Eskom's Palmiet Pumped Storage Scheme and the water transfer scheme, Kleinplaas Dam in the Jonkershoek River on the route of the Riviersonderend-Berg River Tunnel System and Misverstand Dam on the Berg River. Details of the main dams, including their contributions to the system yield, are given in Table 3.1.

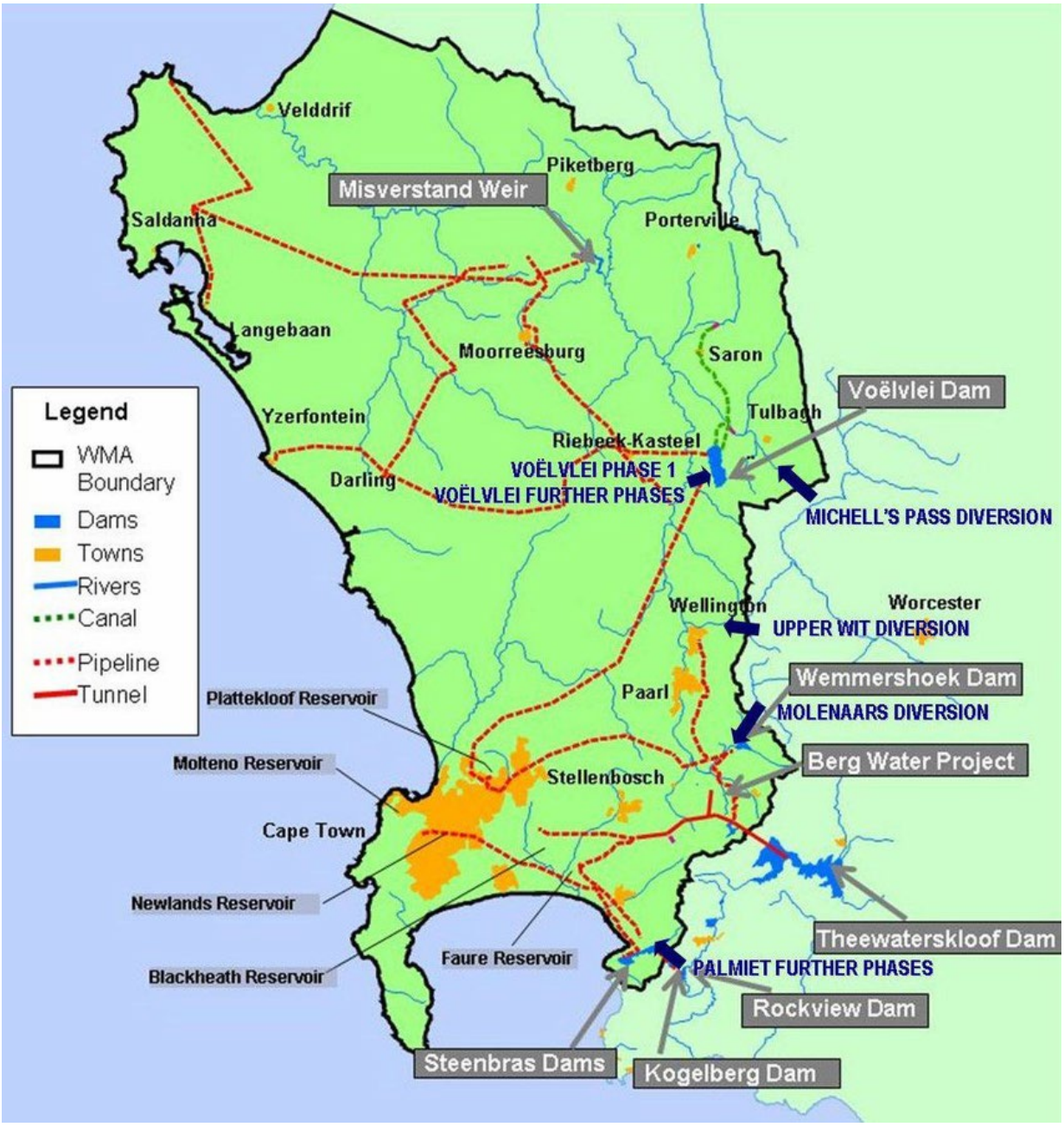


Figure 3.1. Bulk water infrastructure of the WCWSS (from DWS, 2014).

Table 3.1. Major dams of the WCWSS.

Main dam	Full Supply Capacity (Mm^3)	Incremental 1:50 Year Yield (Mm^3/a)	Owner	User
Kogelberg-Rockview	17	23	DWS	City of Cape Town; Eskom
Upper Steenbras	32	40	City of Cape Town	City of Cape Town
Lower Steenbras	34			
Wemmershoek	59	54	City of Cape Town	City of Cape Town; Drakenstein
Voëlvlei	172	105	DWS	City of Cape Town; West Coast; Irrigators
Theewaterskloof (includes Banhoek & Wolwekloof)	480	219	DWS	City of Cape Town; Stellenbosch; Overberg; Irrigators
Berg River Dam and Supplement Scheme	127	80	TCTA	City of Cape Town; Others

3.2.2 Lower Berg sub-systems of the WCWSS

The Lower Berg River sub-system of the WCWSS is made up of the Voëlvlei Government Water Scheme (GWS) and Misverstand Dam. The Voëlvlei GWS essentially comprises Voëlvlei Dam and canal diversions from the Klein Berg, Twenty-Four and Leeu Rivers, which convey water into Voëlvlei Dam. The off-channel Voëlvlei Dam has a full supply capacity of 172 Mm^3. It provides water to the City of Cape Town and the West Coast District Municipality (WCDM) which distributes water to local authorities and other consumers in the area from Malmesbury to St Helena Bay. During winter, the weirs on the Twenty-Four Rivers and on the Leeu River divert up to 34 m^3/s into Voëlvlei Dam. Similarly, a weir on the Klein Berg River diverts up to 20 m^3/s of water into Voëlvlei Dam. Both diversions are via canals. The Twenty-Four Rivers canal is also used for supplying irrigators along the canal during summer.

Water is released from Voëlvlei Dam into a canal which discharges into the Berg River downstream of Sonqwasdrift to supply irrigators during the summer months and also to supply the WCDM Withoogte Water Treatment Works, which abstracts water at Misverstand Dam.

Misverstand Dam has a capacity of 7 Mm^3, although currently this storage capacity cannot be fully utilised due to possible cavitation of the intake pipeline to the Withoogte pump station. A proposal has been made to DWS to fully utilise the potential storage capacity in Misverstand Dam by changing the outlet structure. This was initially intended to provide additional storage capacity to increase the yield to West Coast, but an alternative option could be to use this to better manage downstream flows to the estuary. The main purpose of Misverstand Dam is to divert water to the WCDM pump station which delivers water to the 72 Mℓ/day Withoogte Water Treatment Works and thence to the Vredenburg/Saldanha area. The dam also provides limited regulation of the summer releases from Voëlvlei Dam which are re-released at Misverstand Dam to downstream irrigators.

Water is conveyed from the City of Cape Town high-lift pump station at the Voëlvlei Water Treatment Works, which has a capacity of 273 Mℓ/d, to the Platteklooof Dam on the outskirts of

Cape Town. Water is also released from Voëlvlei Dam to the 30 Mℓ/day Swartland Water Treatment Works of the WCDM. From there the water is distributed to various towns.

3.2.3 Hydrology of the WCWSS

The hydrology of the WCWSS, including the Berg River Estuary, is available for the period 1928 to 2016 as a monthly time series of simulated natural inflows. This has been used in the WCWSS modelling for the annual operating analysis (AOA) for the 2017/2018 and 2018/2019 operating years by the Department of Water and Sanitation (DWS). The Annual Operating Analysis (AOA) is commissioned by the DWS on an annual basis and is a process whereby the management and operation of the WCWSS is reviewed for the coming year based on the total storage in the system at the end of the winter season and the anticipated water requirements of the water users to determine whether water restrictions will be needed or not. This analysis takes place every year in November. The hydrology adopted in the legacy WCWSS modelling since 2005 spanned a 77-year (hydrological) record period, i.e. 1928/1929 to 2004/2005. However, given the unusual rainfall-runoff patterns experienced since 2014, the DWS determined a need to extend the hydrology to the 2016/2017 hydrological year, i.e. 89-year record period, and then incorporate that hydrology into the modelling of the system (WCWSS Annual Operating Analysis (AOA) 2017/2018, DWS 2017). As such, the rainfall records for the WCWSS were extended to September 2017 and the Pitman rainfall-runoff model was used to generate stream flows for the extended rainfall record using the calibrated Pitman parameters from the Berg River Water Availability Assessment Study (WAAS).

Previous analyses of the impact of environmental water requirements for the Berg River Estuary, undertaken as part of the DWS RQOs study were completed prior to this updating of the hydrology and so did not include the recent drought. It is important to note that the hydrology for the WCWSS has simply been extended by including additional precipitation to include the recent drought periods and the models have not been re-calibrated. A full recalibration of the hydrology of the system is required, particularly if it is suspected that there have been fundamental changes in the rainfall runoff responses possible due to climate and land use change that are not addressed in the currently simulated demands.

3.2.4 Water allocation and use in the Lower Berg

Aspects of the catchment context are described by Stuckenberg (2012), including maps from 1986/1987, 1999/2000 and 2007. Nyemba (2013) described spatial and temporal changes in the riparian zone of the Berg River in the vicinity of Hermon (mid-river close to Voëlvlei) from 1955 to 2012. The WAAS (DWAF, 2008) looked at land use and water requirements for the purposes of calibrating the catchment hydrology. The WCWSS Feasibility Studies (DWA 2012) looked at the irrigation allocations down the Berg River including the lower Berg.

Urban water requirements and irrigation allocations are updated annually as necessary for the Annual Operating Analysis for the WCWSS and these are available in the latest configuration of the model, however, these tend to all be lumped together at Misverstand Dam and considered in the total system allocation rather than making a detailed study of the specific irrigation demands downstream of Misverstand Dam and from the contributing catchments.

The current allocations for urban, industrial and agricultural use in the lower Berg Catchment are given in Table 3.2 which was provided by the Western Cape branch of the Department of Water

and Sanitation (DWS). These are separated into summer and winter allocations with additional comments from DWS (Mr John Roberts, personal communication, 22 January 2020).

Table 3.2. Summary of Current Water Use Allocations for the Lower Berg Catchment as provided by DWS (January 2020). Overall use is capped at 18.1 Mm³/a for Lower Berg Irrigation. Upstream and downstream are relative to Misverstand Dam.

Agricultural, Domestic & Industrial	Summer (Nov-Apr) Allocation/ Water Use Licence (Mm³/a)	Winter (May-Oct) Allocation/ Water Use Licence (Mm³/a)	Summer Percentage of Sector Allocation	Comments
Lower Berg Irrigation Board (LBIB) (Summer @3000m³/ha)	**10.97**		**43% of full summer allocation (3000/7000)**	**@ 3000 m³/ha from Voëlvlei Dam**
LBIB upstream	*6.94*			
LBIB downstream	*4.03*			
LBIB (Summer @4000m³/ha)	**14.63**		**57% of full summer allocation (4000/7000)**	**@ 4000 m³/ha not from VVD – from run of river**
LBIB upstream	*9.25*			
LBIB downstream	*5.37*			
LBIB (Winter)		**16.04**		**@7000 m³/ha from run-of-river**
LB IB upstream		*8.44*		*1205.9 ha @ 7000 m³/ha (roughly)*
LB IB downstream		*7.60*		*1086.1 @ 7000 m³/ha (roughly)*
Sub-total agricultural allocation upstream	16.19	8.44		
Sub-total agricultural allocation downstream	9.40	7.60		
Other licences	3.02		60%	
Withoogte WTP (Saldanha, Swart & Berg)	23.44		60% of allocation	Voëlvlei-Berg River & Misverstand Dam
Piketberg	0.70		60% of allocation	Voëlvlei-Berg River & Misverstand Dam
PPC (De Hoek)	0.84		60% of allocation	Voëlvlei-Berg River & Misverstand Dam
Other Industry (Voëlvlei - Berg River)	0.55		60% of allocation	Voëlvlei-Berg River & Misverstand Dam
BBBEE Licences downstream	9.00		90% of allocation	Supply only 1 from TWK Dam & 9 from Voëlvlei Dam
TOTAL allocations upstream	**16.19**	**8.44**		
TOTAL allocations downstream	**46.95**	**7.60**		
Note on periods for allocation. Summer could be considered as year-round (hence farmers may abstract their full allocation when the river flow increases in early winter (May or Aug/Sept). Winter only applicable in set period				

DWS has undertaken a Verification and Validation (V&V) study for the Berg River to determine existing lawful use (ELU) and current allocations. These data, however, have not been released and as a result cannot be used in the analysis of estimated flows reaching the estuary.

It was also noted that subsequently many of the irrigators in the Berg Catchment are now being monitored by the relevant irrigation board, but as yet these data have not been made available for analysis in order to determine the actual use necessary to effectively manage the system. It is also recommended that a temporary or permanent stream gauge be installed in the lower Berg.

3.2.5 Assurance of Supply

Water availability is highly variable from year to year in a system. This is particularly true in South Africa and the Berg River. The provision of storage capacity does provide some ability to manage the variability in supply to meet the demands of different users, however as part of the sustainable management of the system agreements are reached in terms of different levels of assurance of supply that are also linked to the price that users pay for water. Typically the price of water increases with a higher level of assurance of supply. The current operations of the WCWWS are based on the principal of assigned assurance of supply (AoS):

$$AoS = \gamma_1 \, \delta_1 + \cdots + \gamma_n \, \delta_n$$

where: AoS = Assurance of Supply, γ = Percentage of allocation assured at a specific recurrence interval and δ = recurrence interval expressed as a percentage.

Table 3.3 shows the current assumptions around AoS for the different users in the 2019/20 WCWSS AOA. The AoS used in the 2019/20 WCWSS AOA for the urban sector was marginally higher than the 'Water Pricing Strategy at 97.6%. The agricultural sector, however, got a noticeably higher AoS at 95.9% (4.9% higher AoS). Figure 3.2 graphically shows how these values translate operationally. At a 1 in 10 year recurrence interval, there is a higher yield but also higher risk of failure whereas at a 1 in 200 year recurrence interval, there is a lower yield but at a lower risk of failure. To mitigate the impact of risk of failure on the supply allocations, a higher percentage of the urban allocation is assured at a higher recurence interval (50%) compared to 20% of the irrigation allocation.

Table 3.3. Current Assurance of Supply used in the WCWSS AOA (Source: DWS)

Category	Percentage of allocation assured at a specific recurrence interval				Assurance of Supply
	1:200 year (99.5%)	1:100 year (99%)	1:20 year (95%)	1:10 year (90%)	
Urban – City of Cape Town and other	50	20	20	10	97.6%
Irrigation (Low assurance, i.e. full allocation taken up)	20	30	25	25	95.9%

The AoS for different users needs to be taken into account when considering water availability for different users during a drought and also the potential tradeoff when considering how the impacts of providing additonal water to the estuary (if required) could impact on the water provided to different users particuarly in the situation when the available water is reduced by climate change.

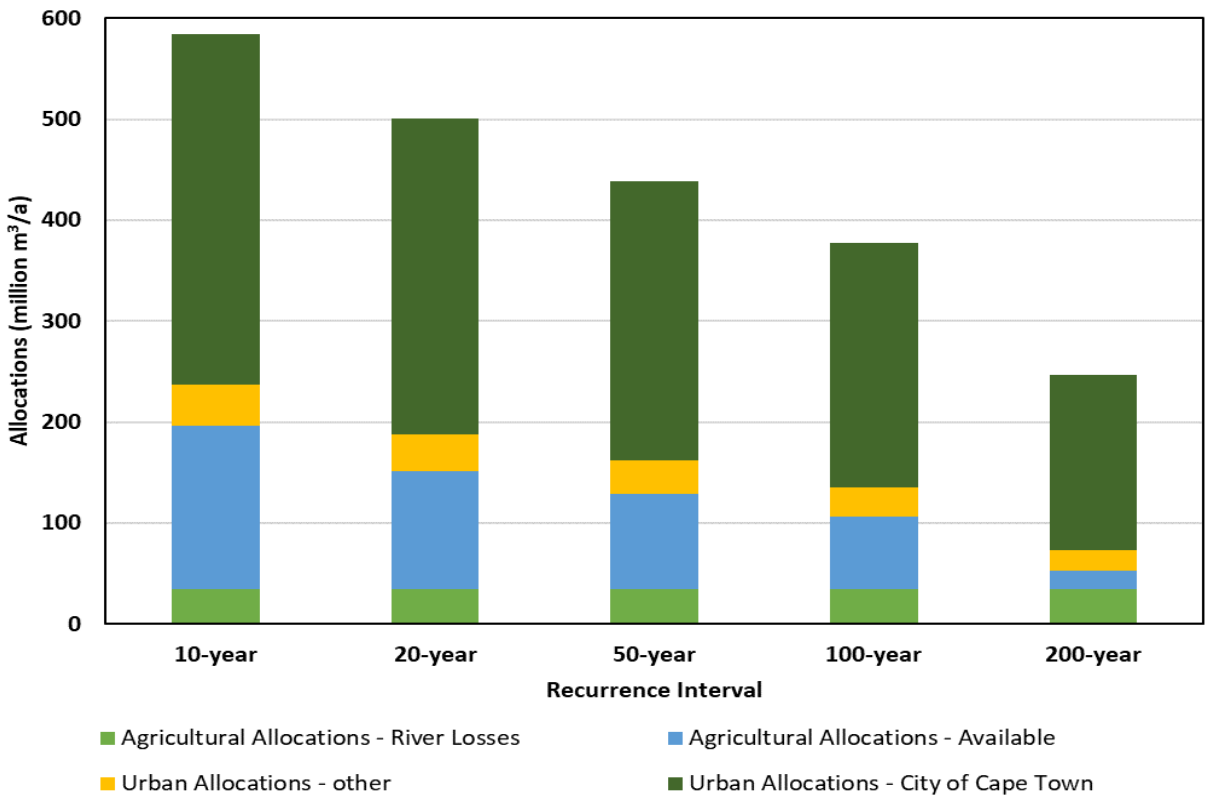


Figure 3.2. WCWSS Allocations at different recurrence intervals for different users – as per Annual Operating Analysis (AOA) WCWSS

3.2.6 The Impacts of Invasive Alien Plants (IAPs)

At a national scale invasive alien plants (IAPs) are thought to contribute to a reduction in mean annual runoff (MAR) of 1 444 Mm^3/a or 2.88% of the MAR for the current level of invasion (Le Maitre *et al.* 2016). This is estimated to be around 55 Mm^3/a for the WCWSS (TNC, 2018). In terms of the potential impact on water supply, it is critical to determine not only the impact on MAR, but on the yield of the system, which is defined as the ability of a system to provide a certain level of supply at a given level of assurance. At a national level, the impact of current estimated IAPs on the available yield from our surface water resources (i.e. dams, rivers, lakes and wetlands), at a 1:50-year level of assurance of supply, has been estimated to be around 4% of currently registered total water use for current levels of invasion, but this has been projected to increase to as much as 16% of registered water use in the future, if IAP spread is not reversed (Cullis *et al.*, 2007).

In the Berg Catchment it was estimated that the impact of IAPs on the water supply system (at a 1 in 50 year level of assurance) is currently around 19 Mm^3/a or 2.6% of the registered water use but that this could increase to as much as 66 Mm^3/a or 9.2% if not addressed (Cullis *et al.*, 2007).

The Consolidated Drought Support Project for the City of Cape Town (Aurecon 2019) undertook a validation of the extended hydrology which included consideration of the impact of riparian vegetation, streamflow reduction activities i.e. alien invasive plants (AIPs) and afforestation, and evaporation (Figure 3.3). This study showed that IAPs currently have an impact of around 27 Mm^3/a on the total system yield and that this could increase to as much as 95 Mm^3/a or 17% of

total system yield by 2045. This is consistent with previous studies that have indicated the likely impact of IAPs on water availability across South Africa (Cullis, 2007; Le Maitre 2016; TNC, 2018).

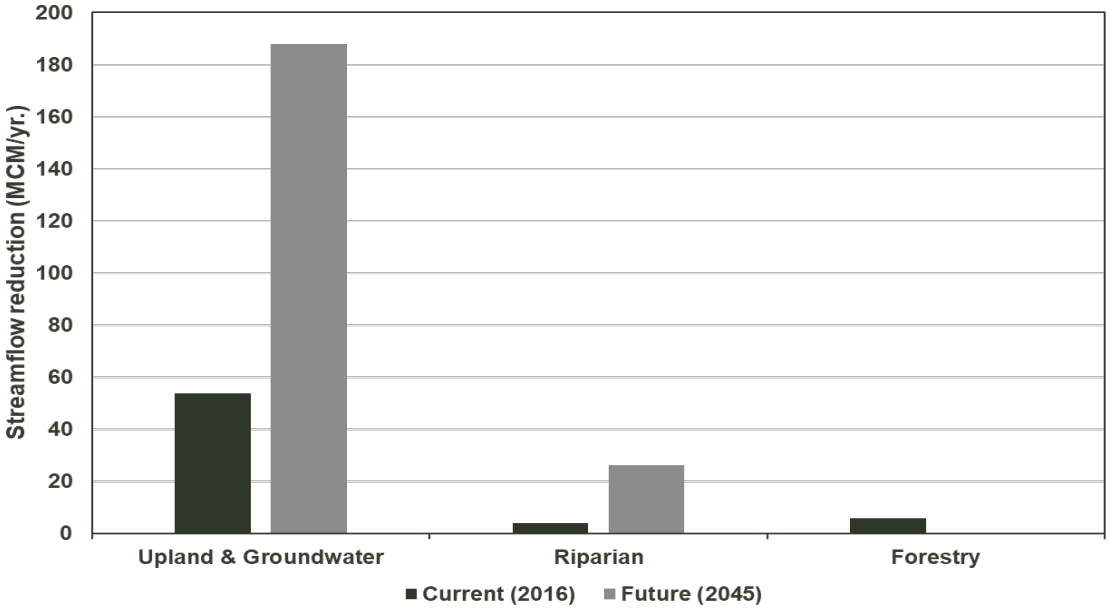


Figure 3.3. Current and future impacts of IAPs and Forestry on streamflow of the WCWSS (Source: Aurecon 2018)

3.2.7 Climate change impacts

The Western Cape is considered to be one of the most vulnerable regions of the country in terms of the potential for reduced rainfall and associated runoff as a result of climate change. When examined through a hybrid frequency distribution (HFD) plot of likely climate change impacts across all the primary catchment areas of South Africa (Figure 3.4), it is clear that the Berg and Breede Catchments (G1 and H6, respectively) are the only primary catchments where all scenarios considered under this analysis show a reduction in mean annual runoff by 2050 (after Cullis *et al.* 2015). This is consistent with the latest analysis by the CSIR (Engelbrecht *et al.*, 2019).

An analysis of the impacts of climate change on the available yield from the WCWSS suggested that climate change could reduce the MAR by around 15% and historical firm yield by around 60 Mm^3 or 8% by 2050 (DWS 2019). This was based on the analysis of the 10th percentile climate change risk scenario (dry) from the HFD study and including the future augmentation options for the WCWSS (Cullis *et al.* 2015). Currently the City of Cape Town is analysing additional climate change scenarios and the impact that these will have on the yield from the WCWWS.

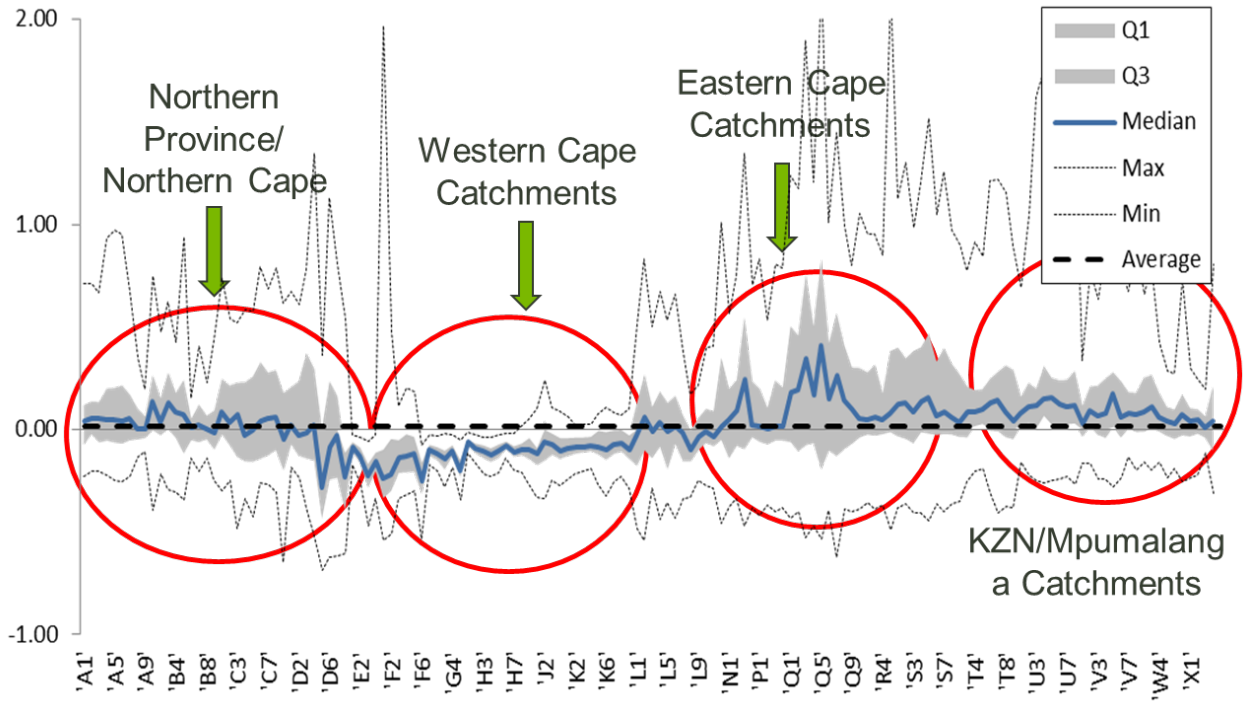


Figure 3.4. Range of potential climate change impacts on Mean Annual Runoff (MAR) for Secondary Catchments across South Africa for the Unconstrained Emissions Scenario (UCE) by 2050 (Cullis *et al.* 2015).

3.2.8 Berg River Estuary environmental water requirements (EWRs)

A summary of the recommended Resource Quality Objective (RQOs) for the Berg River Estuary (DWS, 2019) are given in Appendix 1. These include the requirement for a minimum flow of 0.6 m^3/s entering the estuary which is required to prevent the formation of a hyper-saline wedge developing and persisting in the estuary, and a certain proportion of the natural monthly flow requirements during the winter months, consisting primarily of high flow and flood requirements that are necessary for habitat maintenance, and for periodic scouring of the estuary and flooding of riparian vegetation.

Currently the EWRs are not being applied and it is estimated that these water requirements are not reaching the estuary. It has been proposed that these EWRs come into force with the development of future augmentation options including the planned Voëlvlei Augmentation Scheme (VAS) (John Roberts, pers comm). A challenge with implementing the EWRs is that there are no stream gauges downstream of Misverstand Dam with which to measure compliance with the EWRs and up until recently very few of the users were being metered.

According to Bertrand Van Zyl of DWS (pers. comm) a streamflow gauging site has been identified and a preliminary design done but unfortunately the founding conditions are challenging and it would be too expensive to construct. It is however recommended that alternative options for improved stream gauging downstream of Misverstand Dam be explored as part of the BRIP. As an alternative, it is recommended that DWS and/or the Western Cape Government implement a simple stream gauging section at the identified site (or other potential sites) using a rated section and that this be monitored on a regular basis as part of the Berg River Improvement Plan (BRIP).

3.3 Simulated observed flows reaching the estuary

There is no stream gauge located at the head of the estuary, with the closest available stream gauge (G1H031) being located downstream of the Misverstand Dam, which is located some 100 km upstream of the head of the estuary. The head of the estuary itself is also located some 70 km from the mouth. Hence there is no historical record of observed streamflow entering the Estuary.

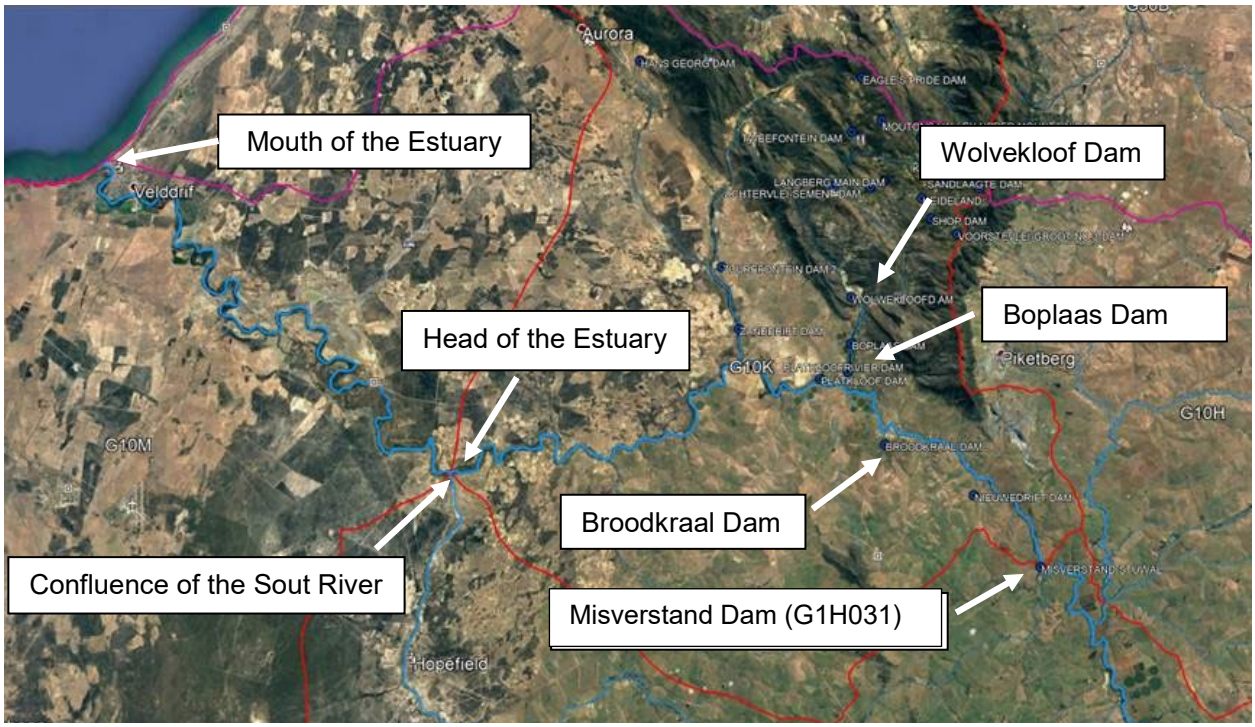


Figure 3.5. The location of Misverstand Dam and the head of the estuary.

Analysis of observed flow at Misverstand Dam (G1H031) provides some indication of the general trends in streamflow reaching the estuary, but in order to have a better understanding of the likely volumes of water actually reaching the estuary, particularly during the summer months, it is necessary to estimate the irrigation demands downstream of Misverstand Dam as well as the potential contributions from the incremental catchments (G10K and G10L), and the impact of any natural river losses including potential losses due to the impact of invasive alien plants (IAPs).

There are also a number of farm dams located in the catchment below Misverstand Dam (G10K) including Broodkraal (9.1 Mm^3), Platkloof Dam (1.2 Mm^3) and Nieuwedrift Dam along the Berg River with a combined total volume of 10.6 Mm^3, and 22 farm dams on the tributaries including Platkloofrivier (3 Mm^3) and Mountain Dam (1.95 Mm^3), and other smaller dams with a combined total volume of 9.4 Mm^3. The location of some of these dams is shown on Figure 3.5. These dams capture runoff from the incremental catchment and are also used to abstract and store water from the Berg River, including some of the releases from Misverstand Dam.

In this section we present:

- A summary of the observed flows at Misverstand Dam (G1H031)
- An assessment of the irrigation demands downstream of Misverstand Dam
- The resulting simulated Natural and Present Day flows reaching the estuary.

3.3.1 Summary of observed flows at Misverstand Dam

The flow gauge just downstream of Misverstand Dam (DWS gauge code G1H031) has a daily record from 1 June 1974 to 28 May 2019. The daily flow at the flow gauge for the observed period is shown by the orange line in Figure 3.6, while the average daily winter flows are presented on Figure 3.7. Together these data highlight the extreme nature of the drought in 2017, where the average daily flow rate for the winter months in 2017 was only 1.2 m^3/s compared to the daily average winter flow rate of 43.8 m^3/s for the full observed period (1974 to 2018) as shown by the orange line on Figure 3.7. A box and whisker plot of the mean monthly flows observed at G1H031 are shown in Figure 3.8. This emphasises how variable the flows are in the Berg River, particularly in the winter months. A comparison of the simulated natural (from the WRYM) and observed MAR at Misverstand is shown in Figure 3.9. These reveal a gradual downward trend in both the observed MAR and in the simulated natural flows, which suggests that this may be due to increasing upstream demands as well possible effects of climate change.

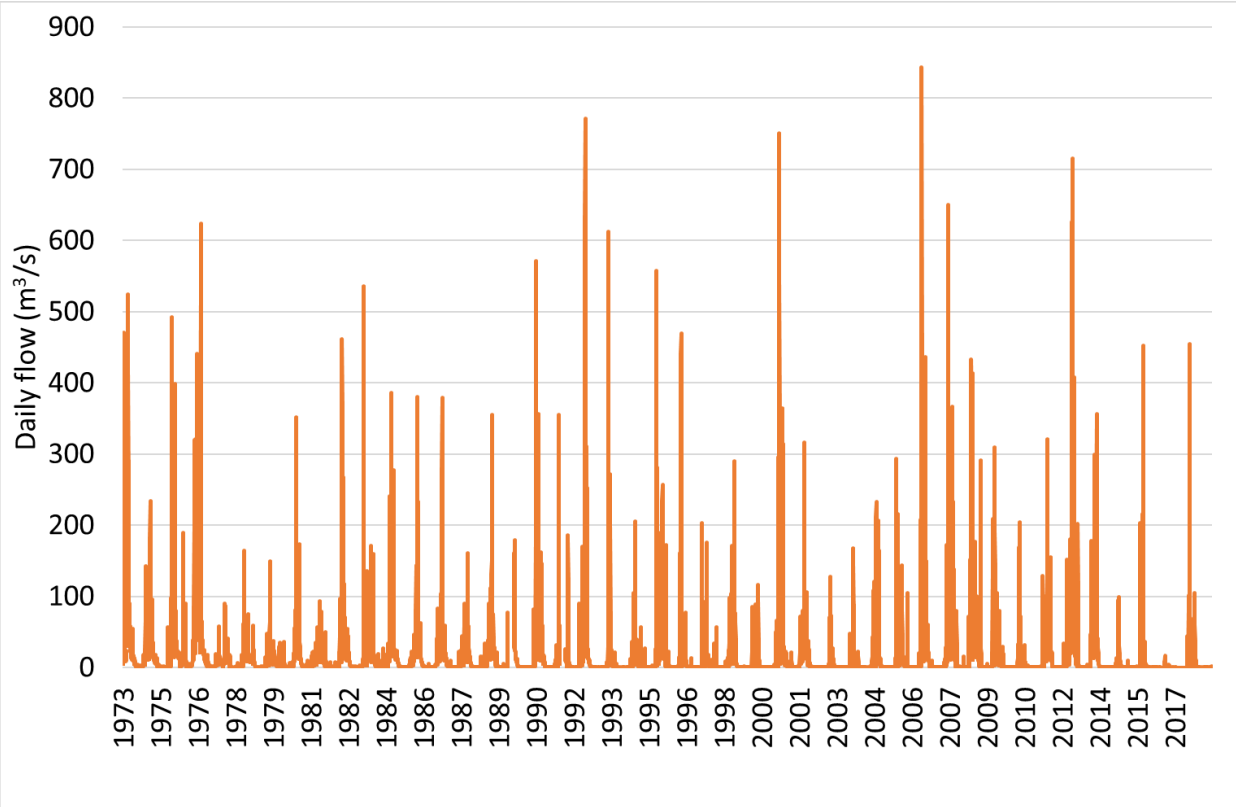


Figure 3.6. Observed daily flows at G1H031 downstream of Misverstand Dam.

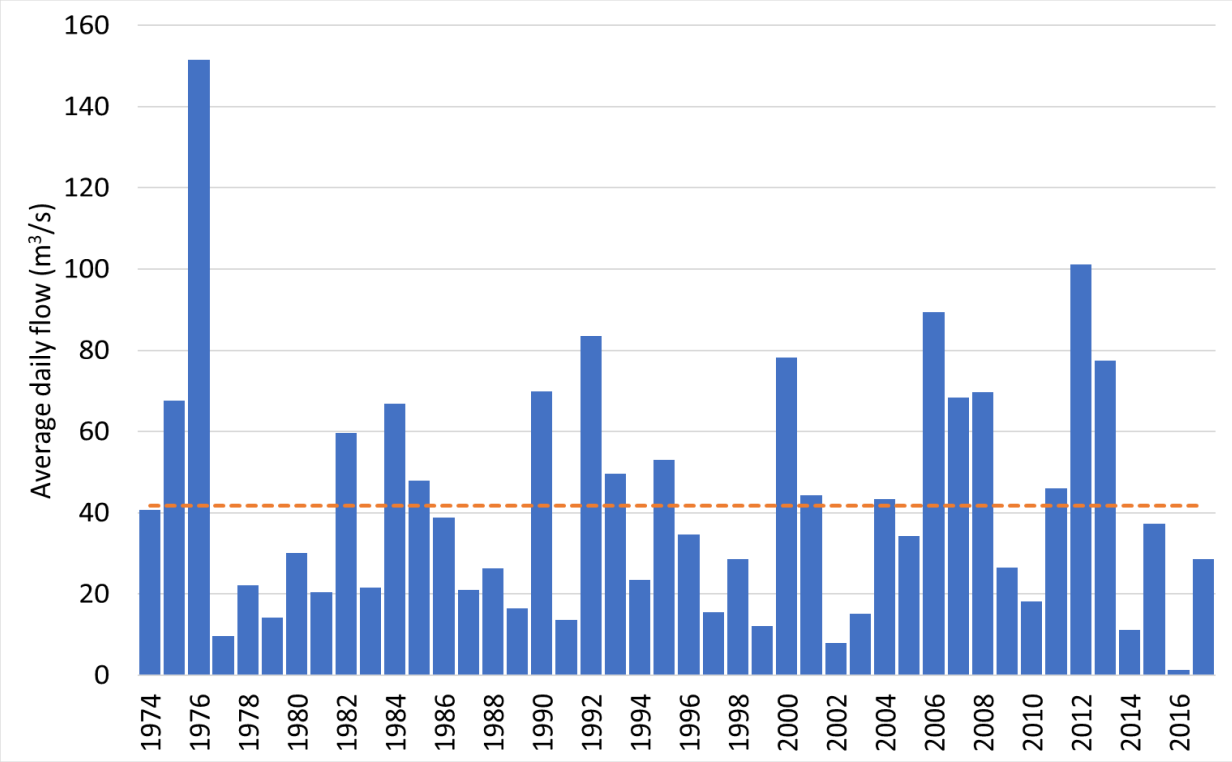


Figure 3.7. **Average observed daily winter flows (June, July, August) at G1H031 downstream of Misverstand Dam (long term average daily winter flows shown by orange line).**

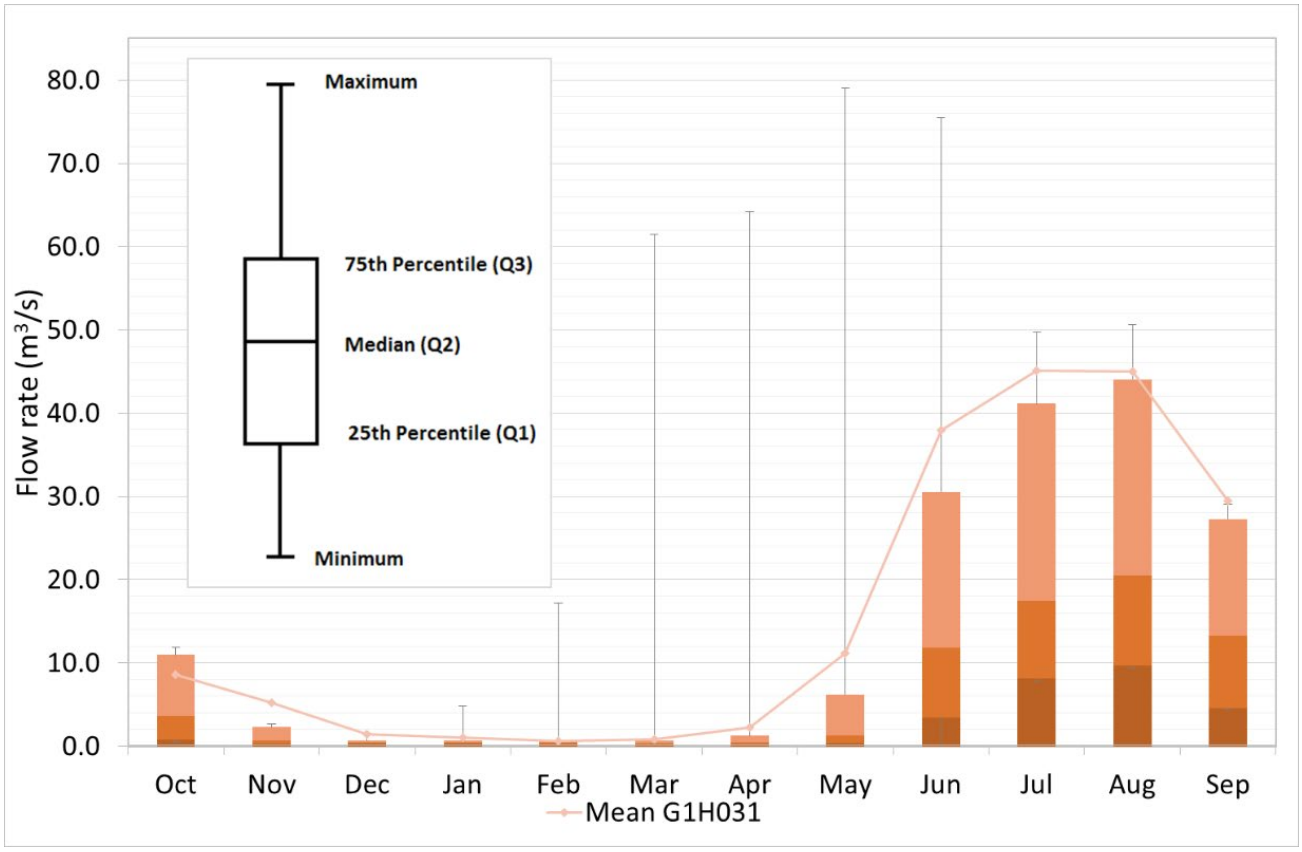


Figure 3.8. **Box & whisker plot showing range of observed mean monthly flows at G1H031 downstream of Misverstand Dam (1974-2018).**

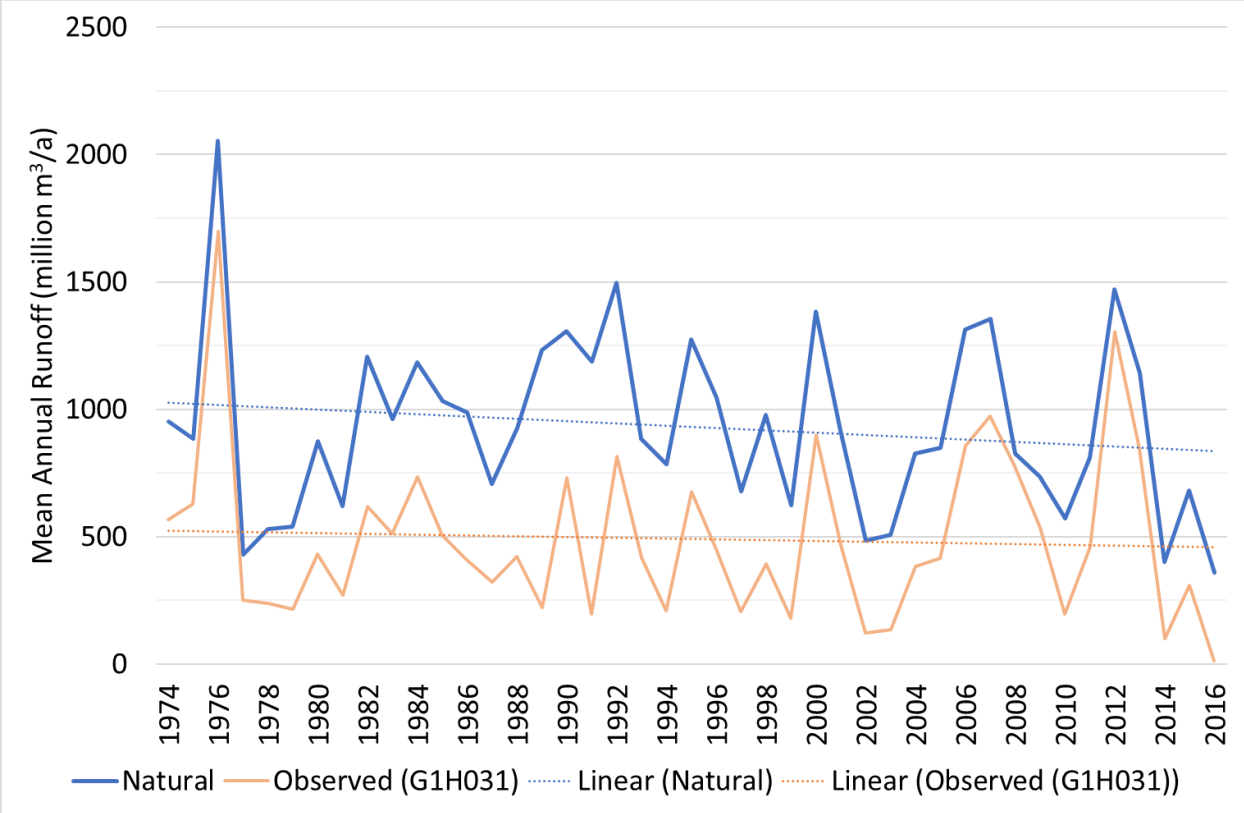


Figure 3.9. Simulated Natural compared with Observed mean annual runoff (MAR) at G1H031 downstream of Misverstand Dam.

3.3.2 Assessment of irrigation demands downstream of Misverstand Dam

A number of sources of information were used to estimate the irrigation demands downstream of Misverstand Dam that could then be used to derive an estimate of the flow actually reaching the estuary. It is important to note that water use downstream of Misverstand Dam is largely controlled by releases made from Voëlvlei Dam which are made according to current allocations from DWS.

The irrigation water requirements downstream of Misverstand Dam were estimated based on the areas of different crop types from the Crop Census and by determining the net water requirement based on crop factors and the calculation of the crop water requirements not met through rainfall. This estimate was compared to the allocations downstream of Misverstand Dam from DWS.

The estimated crop areas downstream of Misverstand Dam (i.e. in catchment G10K) were obtained from the 2013 Western Cape Crop Census (WCDOA 2013) and the 2017/18 Western Cape Crop Census (WCDOA 2018). The total irrigated area consisted of a range of different crop types and was estimated to be 3612 ha in 2013 and 4891 ha in 2018. In the 2013 dataset there are 33 different crop types identified and 50 different crops in the 2018 dataset. Crop types making up 90% of the estimated irrigated area for the 2013 and 2018 crop census are listed in Table 3.4. The remaining 10% irrigated area is comprised of fields less than 100 ha. The 2017/18 census makes a distinction between summer and winter irrigated fields which indicates a relatively small area of irrigation in the summer months, however the coverages were combined

as advised by the DOA and overlapping areas were removed. The extent of the 2018 crop census irrigated areas is shown in Figure 3.10.

Table 3.4. Estimated Crop Areas below Misverstand Dam, i.e. G10K from 2013 and 2018 crop census (WC DOA)

Crop type	Total irrigated crop area from 2013 crop census (ha)	Total irrigated crop area from 2018 crop census (ha)	% Change from 2013 to 2018
Wheat	462	1322	186%
Table grapes	993	831	-16%
Potatoes	396	428	8%
Apple	288	352	22%
Citrus	218	328	51%
Wine grapes	238	269	13%
Onions	44	209	379%
Pear	143	175	23%
Lucerne	203	168	-17%
Small grain grazing	102	162	59%
Planted pastures	47	109	134%
Various mixed crop types <100ha	470	537	14%
TOTAL	3612	4891	36%

Crop factors for each crop type were obtained from the 1990 Surface Water Resources of South Africa Study (WR90, Midgely *et al.* 1994) and then multiplied by the mean monthly evaporation for quaternary catchment G10K to determine the gross crop water requirement. Rainfall efficiency and irrigation efficiency factors were applied to obtain a net water requirement. The total mean monthly crop water requirements for the crop census 2013 areas below Misverstand Dam are shown in Figure 3.11. The total gross requirement is 21.3 Mm^3/a, and the net demand is 18 Mm^3/a (taking into account effective rainfall and irrigation efficiency). Similarly, the total mean monthly crop water requirements for the crop census 2018 areas below Misverstand Dam are shown in Figure 3.12. The total gross requirement is 26.3 Mm^3/a, and the net demand is 21.6 Mm^3/a (taking into account effective rainfall and irrigation efficiency).

The current DWS allocations for the Lower Berg Irrigation Board (LBIB) were provided to the study team by Mr John Roberts at the DWS and were used to update the model configuration. "According to the Raw Water Supply Agreement (DWS & City of Cape Town, 2003) the original allocation for irrigation water use to the Lower Berg IB was 18.1 Mm^3/a. However, an additional 10.4 Mm^3/a has subsequently been approved by the DWS bringing their total allocation to 28.50 Mm^3/a" (City of Cape Town 2019).

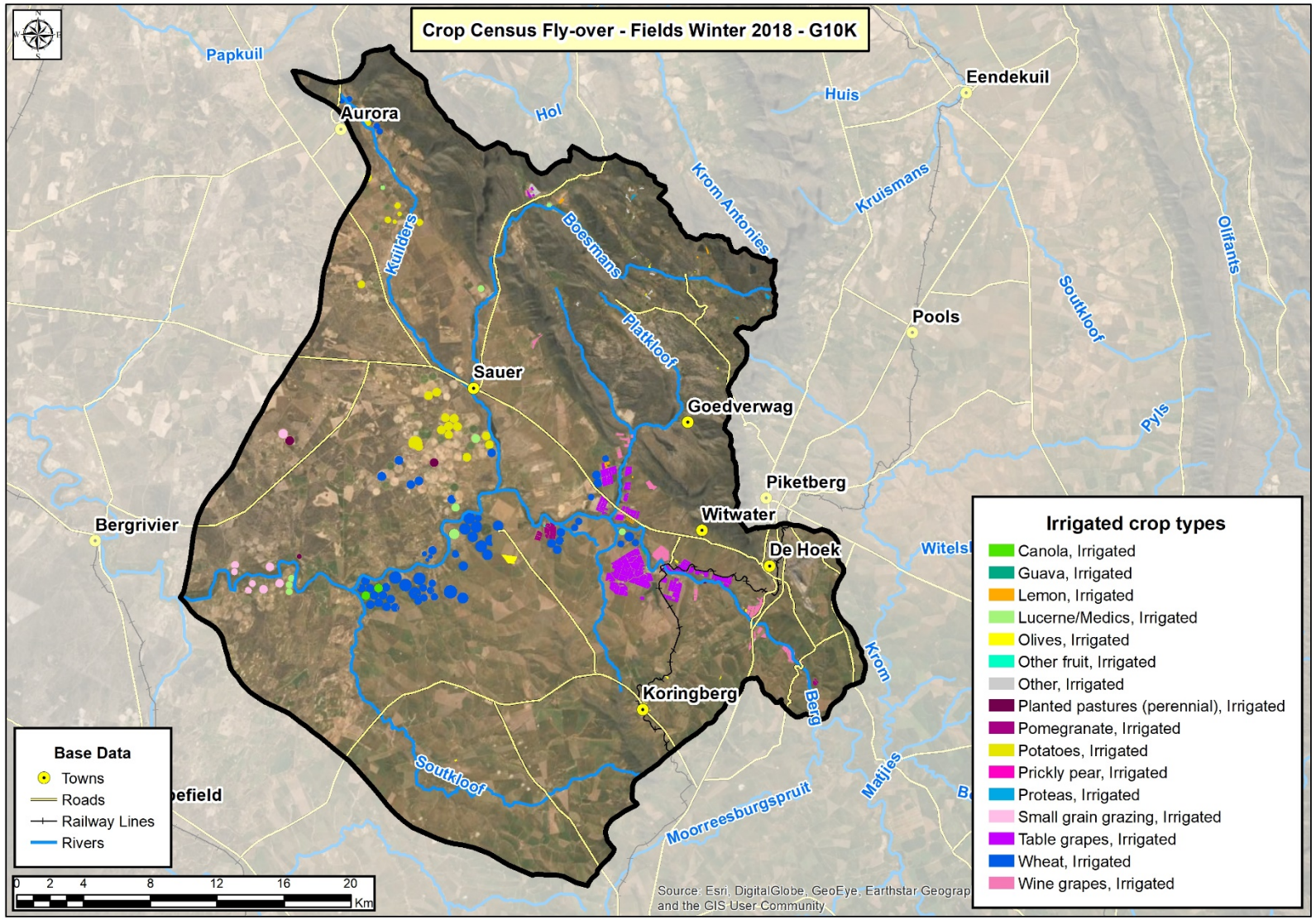


Figure 3.10. Extent of irrigated crops from crop census 2018 in G10K between Misverstand Dam and top of the estuary.

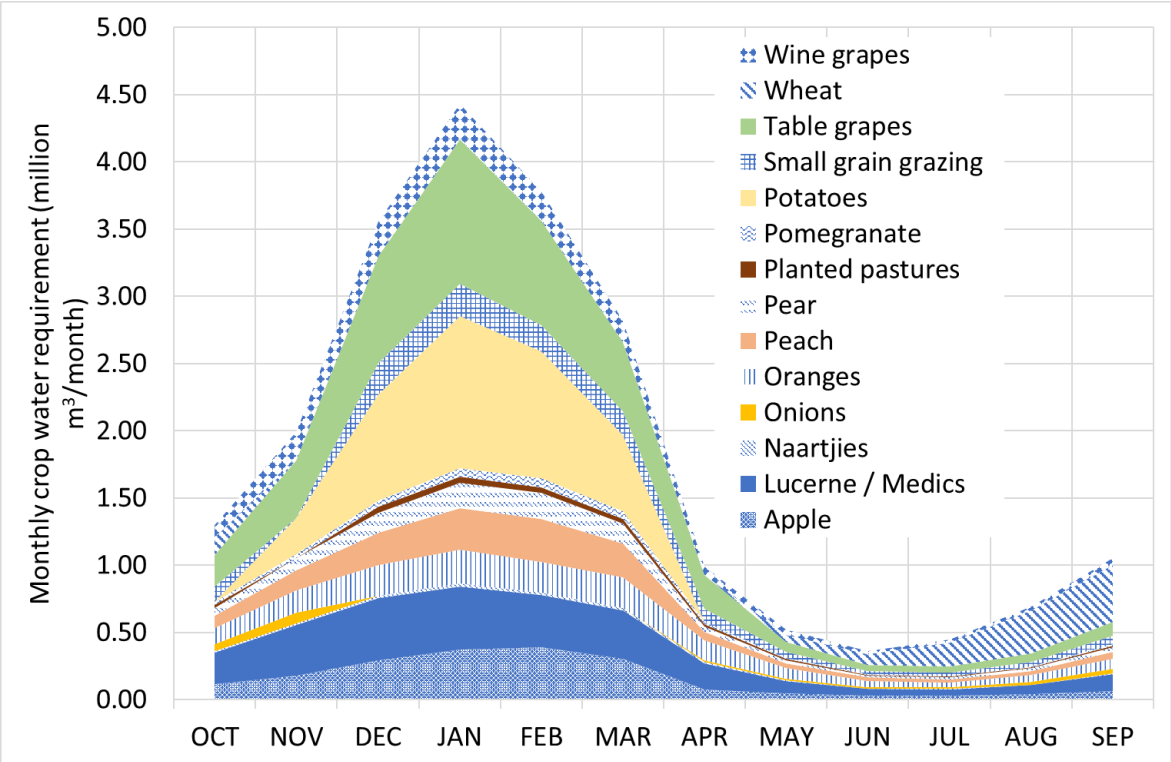


Figure 3.11. Mean monthly crop water requirements (million m³/month) for crop census 2013 areas in catchment downstream of Misverstand Dam (G10K).

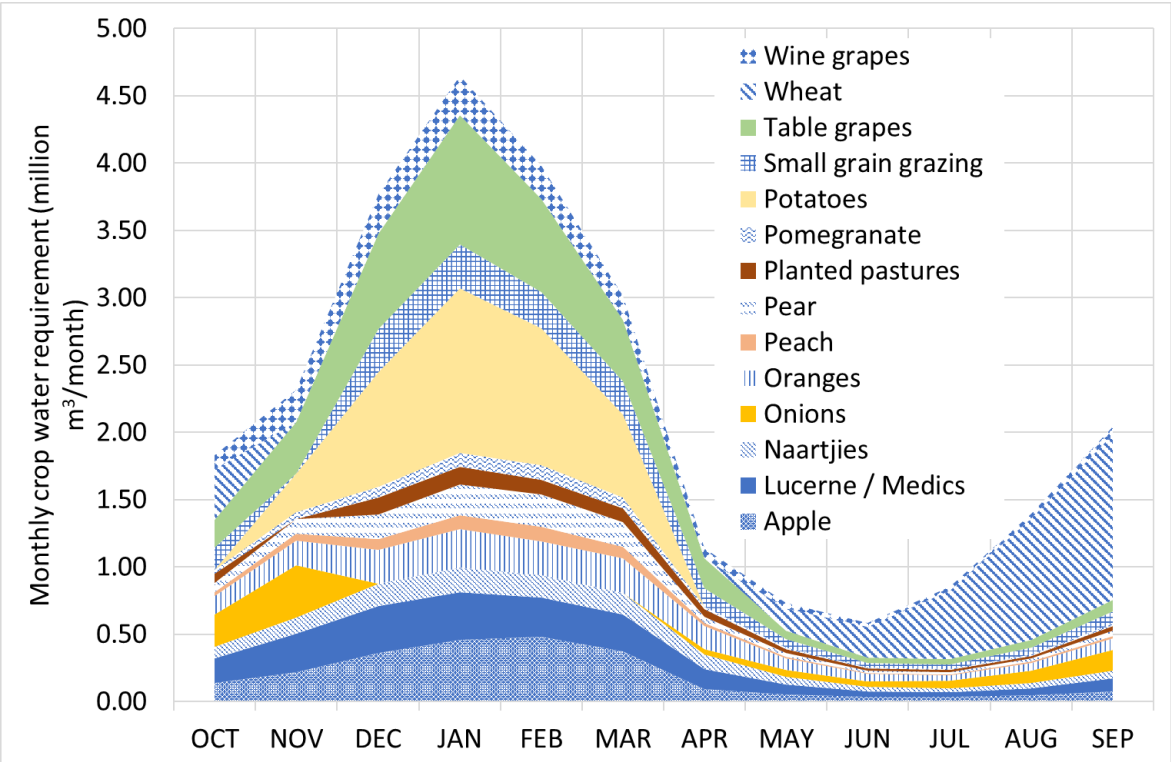


Figure 3.12. Mean monthly crop water requirements (m³/month) for crop census 2018 areas in catchment downstream of Misverstand Dam (G10K).

The full allocation for the LBIB is 7000 m^3/ha, which is split between summer and winter. For the summer allocation, there is a further split where 3000 m^3/ha is provided from storage (i.e. from Voëlvlei and Misverstand dams) and the remaining 4000 m^3/ha is run-of-river. For the winter portion, the total allocation comes from run-of-river. The 18 Mm^3/a allocation in the Water Resources Yield Model is based on the 3000 m^3/ha for the Lower Berg Irrigation Board (LBIB) users upstream **and** downstream of Misverstand Dam. The total allocation for the LBIB users *downstream* of Misverstand Dam is 9.4 Mm^3/a in the summer months and 7.6 Mm^3/a in the winter months. The information provided by DWS is summarised in Table 3.5.

Table 3.5. Comparison of Irrigated areas and estimated demands from different sources

Information Source	Area (ha)	Area (km^2)	Volume (Mm^3/a)	Comment / Notes
DWS allocations (2002/2003) - downstream Misverstand Dam	2429	24.3	17.0	Based on 7000m^3/ha/a allocation. 4.03 Mm^3 summer portion from Voëlvlei (1343.1 ha @ 3000 m^3/ha) 5.37 Mm^3 summer portion from run-of river (1343.1 ha @ 4000 m^3/ha) **(total summer portion = 9.40 Mm^3)** 7.60 Mm^3 winter portion (1086.1 ha @ 7000 m^3/ha)
DOA crop census 2013 (G10K, downstream of Misverstand Dam only)	3612	36.1	18.0	Estimated irrigation demands using WR90 crop factors **16 Mm^3 in the summer** 2 Mm^3 in the winter
DOA crop census 2018 (G10K)	4891	48.9	21.6	Estimated irrigation demands using WR90 crop factors
NLC, 2000	6123	61.2	42.9	High level estimate of demand based on 7000 m^3/ha/a
DEA, 2014	7139	71.4	50.0	High level estimate of demand based on 7000 m^3/ha/a
DEA, 2018	7607	76.1	53.2	High level estimate of demand based on 7000 m^3/ha/a
Berg WAAS				no assessment d/s Misverstand Dam
Western Cape Feasibility study				no assessment d/s Misverstand Dam - Used allocation of 18.1 Mm^3
WR90	1710	17.1		Irrigation area in quaternary G10K
WR90	220	2.2	2.3	From WR90 Appendix 5.2.1 (quota 12 m^3/ha from Misverstand Dam)
WR2005 total	4740	47.4		Source NLC database (CSIR, 2000)
- permanent	630	6.3		from dams
- temporary	1440	14.4		Run-of-river
- irrigation	2670	26.7		Run-of-river
WR2012		47.4		Same as for WR2005

A comparison of the estimated irrigation water requirements downstream of Misverstand Dam based on the 2013 and 2018 crop census data with the current allocations from DWS is presented in Table 3.5. In addition to these sources, the irrigated areas extracted from three national landcover databases are also presented for reference and were used to determine whether the area of irrigation in the Lower Berg Catchment has changed significantly over the past 20 years. These are the National Land Cover (NLC 2000) database (CSIR, 2000), the Department of Environmental Affairs Land Cover 2014 and 2018 respectively. The land cover databases do not provide a detailed breakdown of crops but they do distinguish between cultivated areas of permanent irrigation, temporary irrigation and dryland. Map coverages of these databases are included in Appendix 2.

The land cover databases indicate that there has been an increase in irrigated areas of approximately 24% from 2000 to 2018. However, when the areas of irrigation from the crop census are compared to the corresponding land cover database time slice i.e. Crop census 2013 vs. DEA 2014 and Crop census 2018 vs. DEA 2018, the crop census areas are 51% and 71% of the land cover database areas, respectively. Therefore, one could infer that the irrigated crop areas in 2000 might be approximately 50% of the NLC land cover areas which would be 30.6 km^2 and the change in crop areas from 2000 to 2018 could have increased by as much as 76%.

The comparisons shown in Table 3.5 suggest a much greater irrigation demand in summer than is currently allowed for under the existing DWS allocations i.e. 16 Mm^3/a in the summer months compared to 9 Mm^3, although the irrigation demand is calculated based on the total irrigation area in the quaternary catchment G10K as opposed to the DWS allocation which is only for the portion allocated to the Lower Berg Irrigation Board downstream of Misverstand Dam which is a smaller area. Nevertheless, this would suggest that if not suitably regulated, the downstream users could be looking to abstract more water than they are allocated. It is, however, important to note that not all crops are irrigated every year and also that the allocations are subject to restrictions which are imposed on all users of the WCWSS by DWS.

The actual evapotranspiration estimates for the irrigated fields downstream of Misverstand Dam from the analysis of satellite information through the FruitLook system was obtained for the catchments downstream of Misverstand Dam. The FruitLook service is funded by the Western Cape Department of Agriculture and is executed by eLEAF from the Netherlands in cooperation with its South African partner Blue North. The FruitLook service package provides crop information on a weekly basis disseminated through the data portal www.FruitLook.co.za. It provides weekly updates on 9 parameters for an understanding of the spatial and temporal changes of water, nitrogen and biomass status for particular crops in an area. This is achieved through the processing of Earth Observation data (optical, infrared and thermal infrared sensors) and use in advanced models. The models are calibrated and validated against in-situ soil moisture and energy balance measurements. The parameters that are of interest to this study are: Biomass production, Cumulative Biomass Production, Evaporation deficit, Actual evapotranspiration and Biomass water use efficiency. The data is available at a 20x20 m resolution; therefore at the quaternary scale the data is quite detailed.

It was not possible to extract the crop data from the website portal because this has to be done on a field by field basis for a particular crop. A request was made to eLEAF who extracted the available data for quaternary catchments G10K, G10L and G10M. We received weekly estimates of actual evapotranspiration in a raster database from August 2017 to January 2020. However, this was not linked to a crop type in any way and so it was then necessary to intersect

the 2018 crop census data with the evapotranspiration estimates to determine the actual crop water use for the given period. This can then only be used as a verification of the crop water estimates we have derived in the analysis.

While it would be possible to use this information to better inform the estimate of irrigation demands both current and future, this would require a significant amount of time and effort to process. It is, however, something that should be considered for future improvements to the Western Cape Water Supply System (WCWSS) model.

3.3.3 Historical irrigation demands

The historical irrigation demands downstream of Misverstand Dam in quaternary catchment G10K were determined using the areas identified by the databases described in the previous section. For the period before 2000, the estimated irrigation areas from the Water Resources 1990 study (WRC, 1990) were used. A summary of the irrigated areas from 1920 to 2018 are shown in Table 3.6. A linear growth was applied to obtain the irrigation areas for each year in the time series.

Table 3.6. Historical irrigation areas downstream of Misverstand Dam

Year	Irrigation area (km^2)	Source
1920	6.32	Water Resources 1990 (WRC)
1942	9.15	Water Resources 1990 (WRC)
1970	15.25	Water Resources 1990 (WRC)
1986	17.25	Water Resources 1990 (WRC)
1989	17.25	Water Resources 1990 (WRC)
2000	30.61	National Land Cover CSIR 2000 (adjusted)
2013	36.12	Crop Census 2013, WCDOA
2018	48.91	Crop Census 2018, WCDOA

The crop water requirements for the irrigated areas was determined using the crop factors from the 1990 Surface Water Resources of South Africa Study (WR90). The distribution of crops was assumed to be the same as for the Crop Census 2018. Data tables from this analysis are included in Appendix 3.

The historical water requirements shown in Figure 3.13 below indicate a steady increase in irrigation demand from 1920 to about 1989 when there was a sharper increase in irrigation areas from the period 1990 to 2000, and then another very sharp increase after 2012.

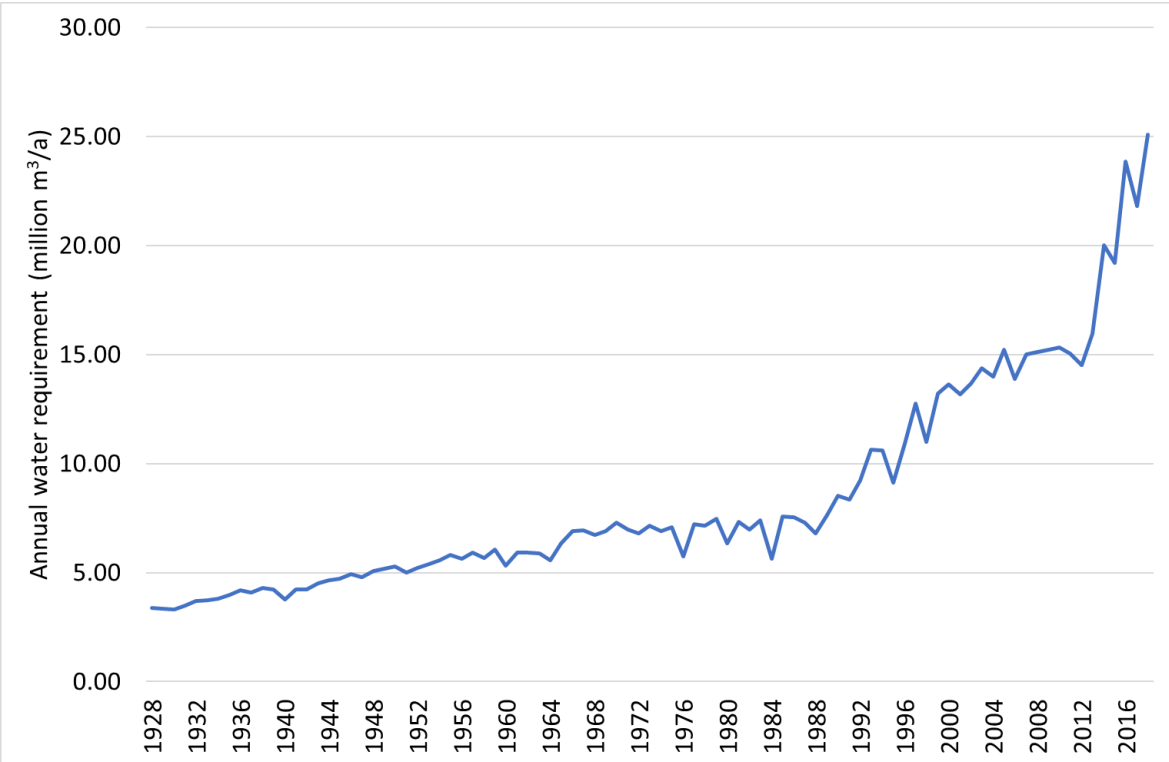


Figure 3.13. Estimated annual historical irrigation demand downstream of Misverstand Dam

3.3.4 Estimated historic inflows to the estuary

As mentioned above, in order to estimate the historic inflows to the estuary it was necessary to start with the observed flow record at Misverstand Dam and then to add any incremental inflows from the downstream catchments and take out the estimated irrigation demands as well as river losses. This was done on a monthly basis according to the following basic equation:

Inflow to Estuary = Observed Flows + Incremental Inflows – Irrigation Demands – River Losses

For the historical inflow estimate, this was done using the historical irrigation requirements determined in Section 3.3.3 and described further below. It was also done for the two scenarios of irrigation demands described above, i.e. firstly based on the estimated crop water requirements from the 2018 Crop Census and secondly according to the allocations, however both of these methods assume a constant irrigation area over the full historical time period. These results are included in Appendix 4.

The mean monthly historical inflows to the estuary are shown in Figure 3.14 and Figure 3.15. The historical inflows to the estuary compare well with flows recorded at G1H031 but there are some small differences in the low flows and high flows. The mean summer low flows to the estuary drop below 0.6 m^3/s in January, February and March. The flow duration curve in Figure 3.16 shows how often this occurs over the period June 1974 to May 2019.

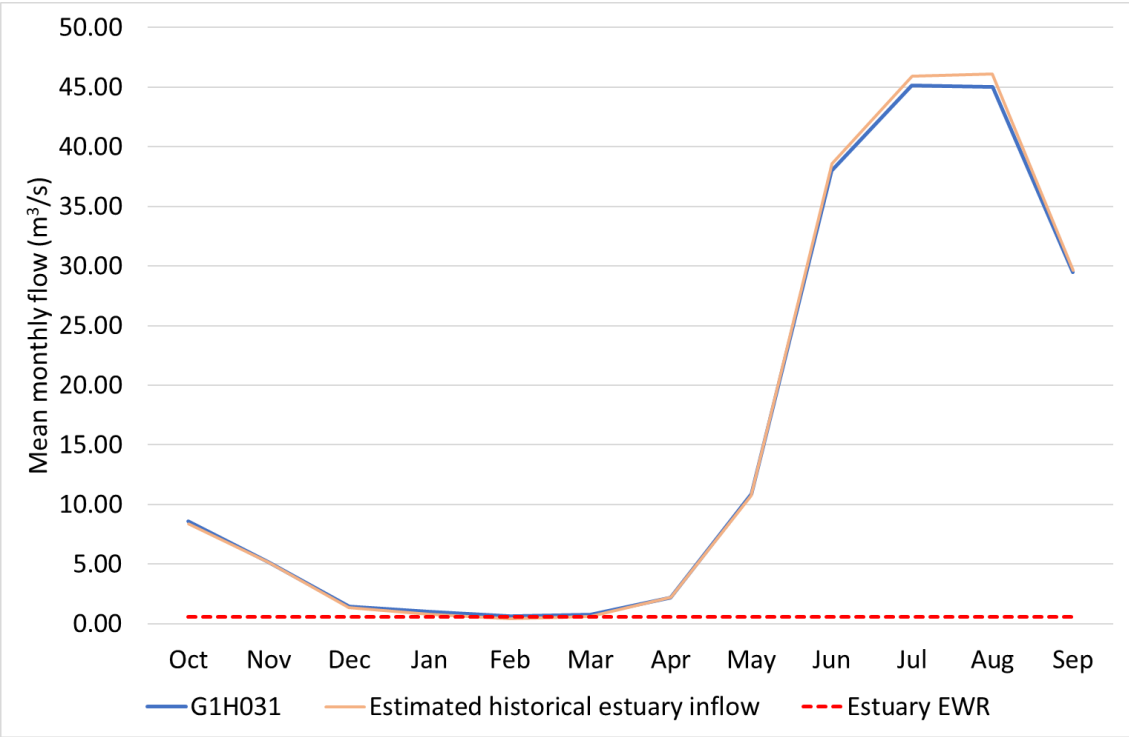


Figure 3.14. Mean monthly flows at G1H031 downstream of Misverstand Dam and historical inflows to the estuary (1974-2019).

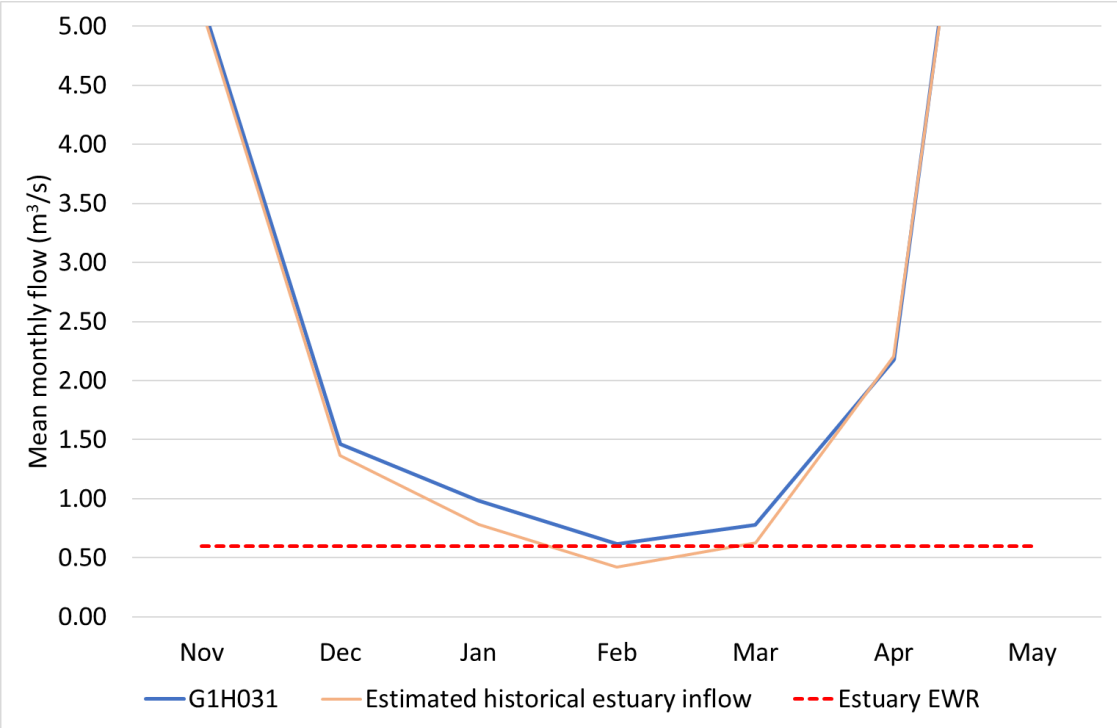


Figure 3.15. Mean monthly flows zoomed to summer months with <5m³/s (Misverstand Dam gauge G1H031 and historical estuary inflow).

The flow duration curve for the estimated monthly flows reaching the estuary are shown in Figure 3.16. Historical summer flows only meet the low flow EWRs less than 15% of the time from 1974 to

2019. Historical annual inflows to the Berg River Estuary together with the historical irrigation demand and the annual summer flows (Jan, Feb, Mar) for the period 1974-2019 are shown on Figure 3.17. Most notable is the decrease in estuary summer flows (green bars) associated with the increased irrigation demand (blue line) from the early 1990s to the present.

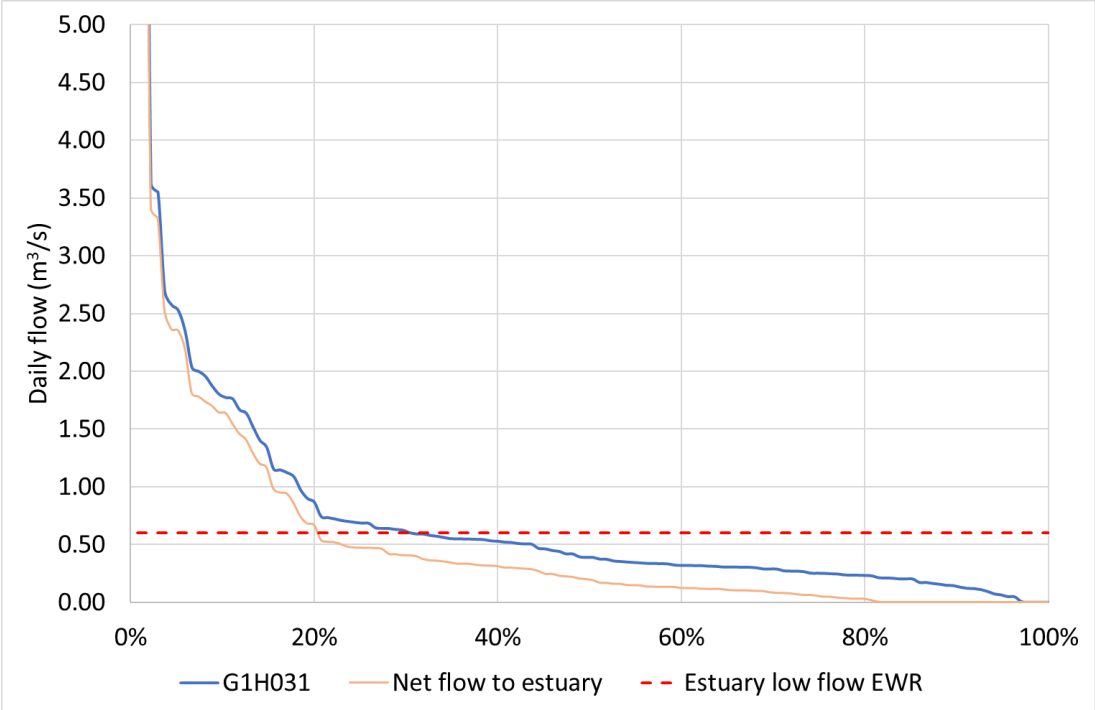


Figure 3.16. Flow Duration Curve for Summer Flows (Jan, Feb, Mar) at G1H031 downstream of Misverstand Dam and reaching the Estuary.

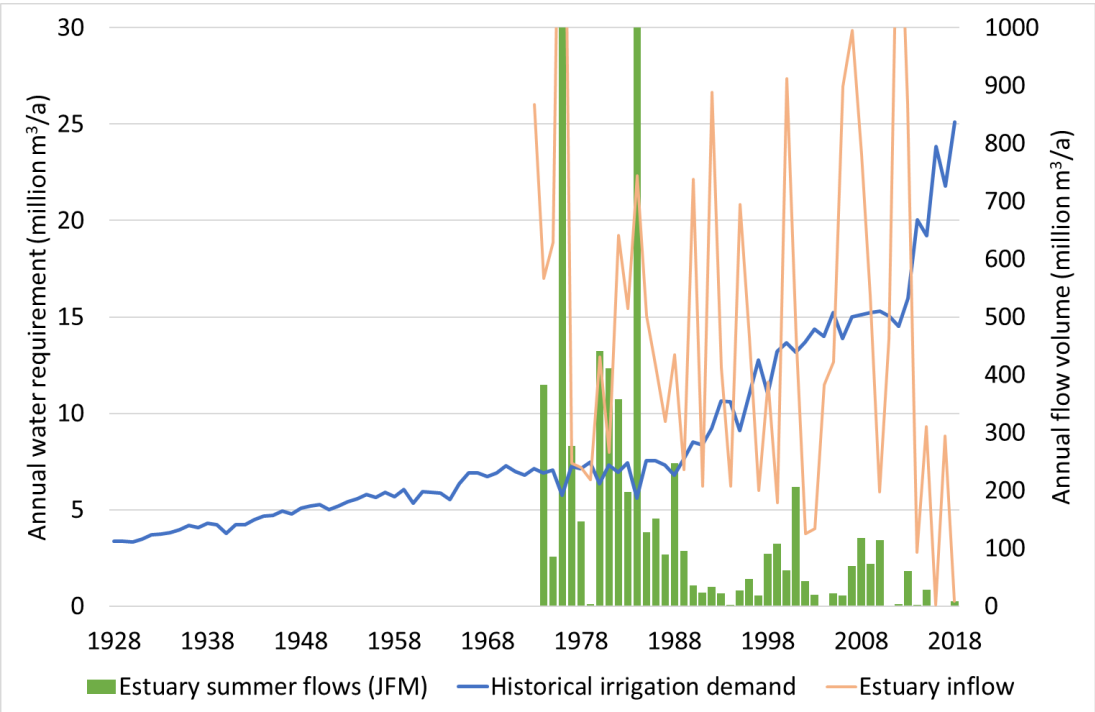


Figure 3.17. Annual historical inflows to the Berg River Estuary.

The resulting estimated monthly flows reaching the estuary for the period 2000 to 2017 are shown in Figure 3.18 and Figure 3.19. These show that during winter there is often more flow reaching the estuary than at Misverstand Dam due to the contributions from the downstream catchments, while during summer there is typically no flow reaching the estuary as it is all being taken up to meet irrigation demands downstream of Misverstand Dam. This appears to confirm the observation that almost no water reaches the estuary in summer.

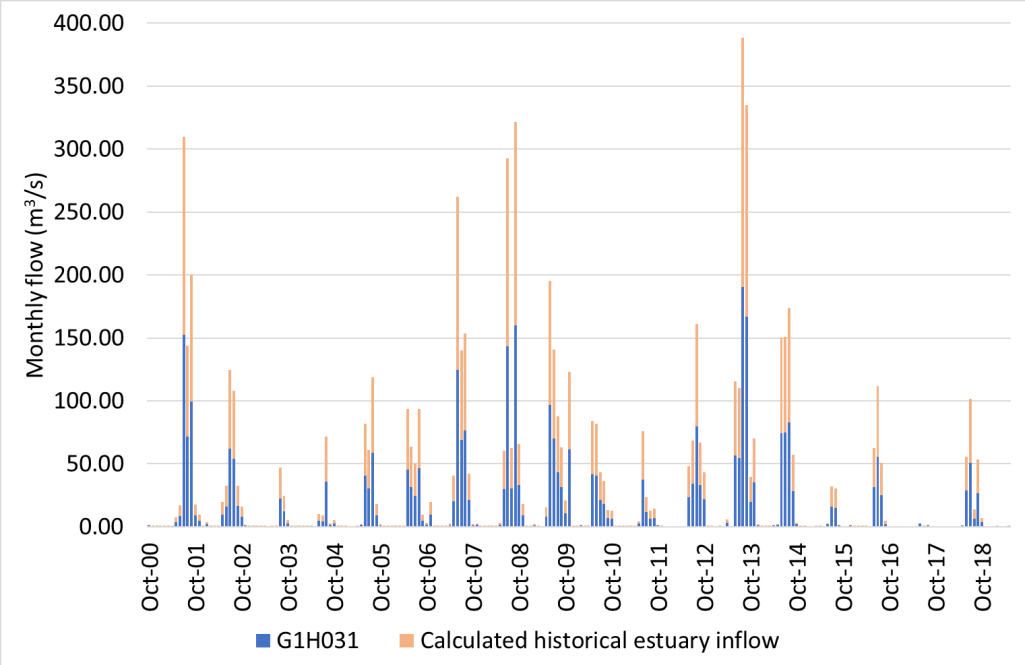


Figure 3.18. Observed monthly flows at Misverstand Dam gauge G1H031 and estimated monthly flows reaching the estuary for the period October 2000 to May 2019.

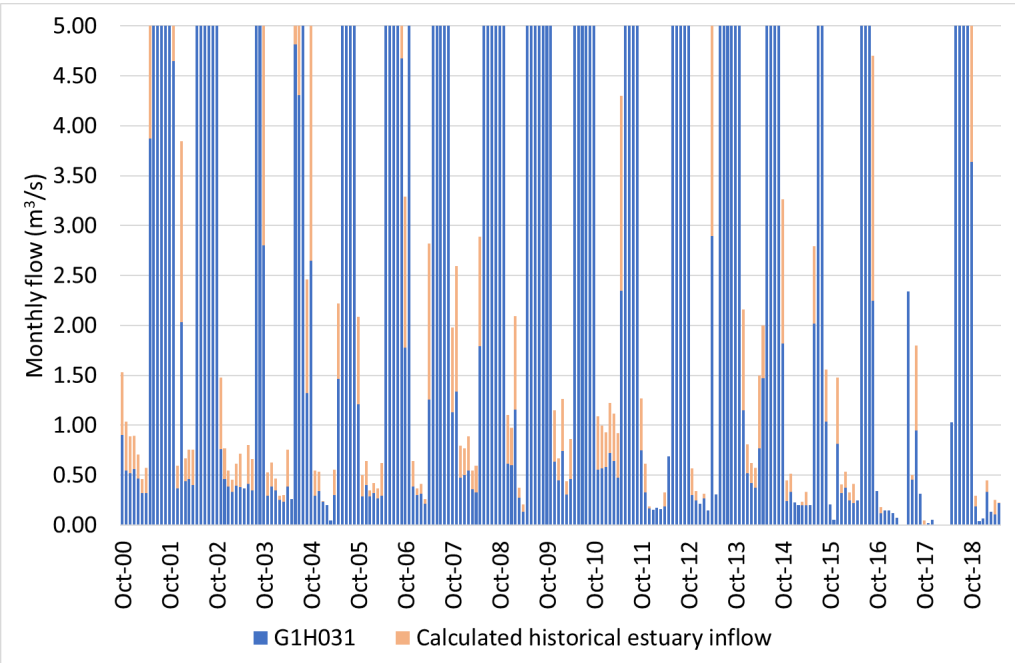


Figure 3.19. Observed monthly flows <5 m³/s at G1H031 downstream of Misverstand Damand estimated monthly flows reaching the estuary from October 2000 to May 2019.

3.4 Analysis of flood flows reaching the estuary

3.4.1 Approach

Our assessment of the characteristics of flood flows upstream of the Berg River Estuary is based on the DWS record of hourly flood discharge rates measured at DWS gauging station G1H031, immediately downstream of Misverstand Dam. This record covers the period 1975 to 2018. We recognise that the tributaries downstream of G1H031 contribute to Estuary inflows, but we deem their contributions to maximum flood discharge rates at the Estuary likely to be minimal, because such tributary flood peaks would arrive at the Estuary well in advance of their related main-stem flood peaks.

3.4.2 Seasonality and variability

Given the wet-winter/dry-summer climate of the Berg River Basin, it follows that the flood regime of the Lower Berg River is highly seasonal with maximum discharges occurring mostly during the months of June, July and August, and much less frequent high flows during May and September. The flood regime is further characterised by high inter-seasonal as well as notable intra-seasonal variabilities of maximum discharge rates, as illustrated by Figure 3.20 and Figure 3.21, respectively. Furthermore, the dotted line in Figure 3.20 indicates the presence of multi-year fluctuations in annual maximum flood peaks, seemingly with a quasi-decadal character.

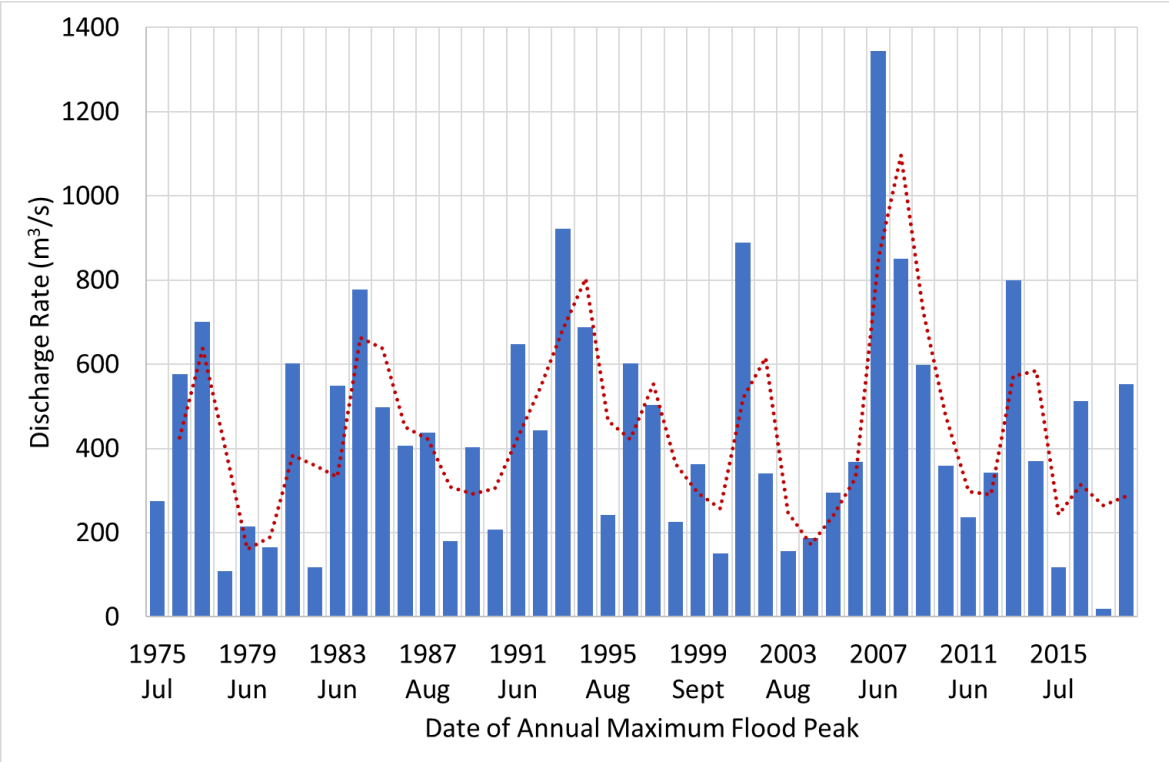


Figure 3.20. Inter-seasonal and multi-year variability of annual maximum flood peaks at G1H031 ddownstream of Misverstand Dam.

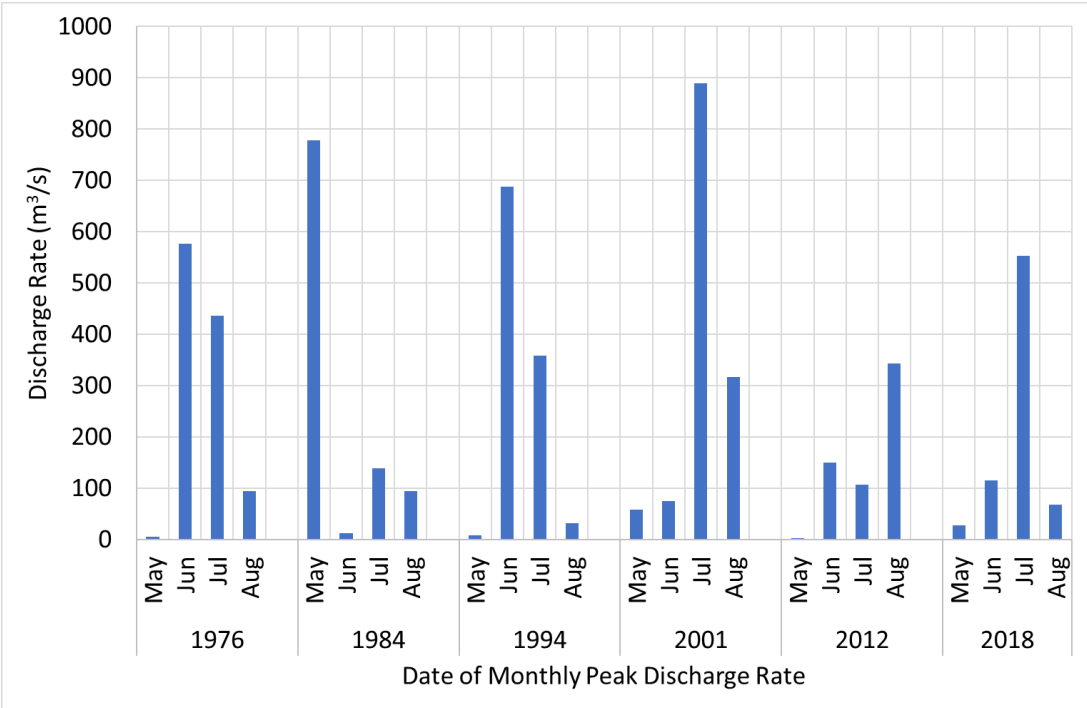


Figure 3.21. Examples of intra-seasonal variability of monthly maximum discharge rates at G1H031 downstream of Misverstand Dam.

3.4.3 Flood discharge exceedance characteristics

The inter-seasonal and intra-seasonal variabilities of maximum discharge rates are also reflected in the exceedance percentage values presented in Table 3.7 for both annual maxima and monthly maxima during the months of May to August for the period 1975 to 2018.

Table 3.7. Exceedance characteristics of annual and monthly maximum discharge rates for the months of May to August for the period 1975 to 2018.

Exceedance Percentage	Annual Maximum Discharge Rate (m^3/s)	Monthly Maximum Discharge Rate during May to August (m^3/s)
Largest Measured	1344	1344
10%	825	549
25%	601	281
50%	386	119
75%	210	58
90%	117	14
Smallest Measured	18	0

The variability of monthly maxima during the May-August winter season is particularly strongly illustrated by the respective 10% and 90% exceedance discharge rates: during the 44 years of streamflow gauging there were 18 individual winter months with maximum discharge rates of 549 m^3/s or more, as well as 18 individual winter months with maximum discharge rates of 14 m^3/s or less.

3.4.4 Probabilistic analysis of flood peak recurrence intervals

Results of a probabilistic analysis of the G1H031 annual maxima depicted in Figure 3.20 are shown on Figure 3.22 . For the purpose of this analysis, two different exceedance probability distributions were fitted to the observed values (green circles), which are plotted in Figure 3.22 in log-probability space by means of the Cunnane Plotting Position. We regard the goodness-of-fit of the General Extreme Value (GEVpwm) distribution as somewhat more representative than that of the Log-Pearson III (LP3) distribution. The resulting flood peaks for a range of recurrence intervals (RIs) are presented in Table 3.8.

It is of interest to note that the 44-year period of record includes a 1:200-year RI flood peak (7 June 2007). A further point of interest is that the 1:2-year RI flood peak is quite substantial at 407 m3/s.

NB: It should be noted that, for this exercise, we considered the extremely low annual maximum of 18 m^3/s on 25 June 2017 to be an anomalous low outlier and, consequently, omitted that value from the probabilistic analysis.

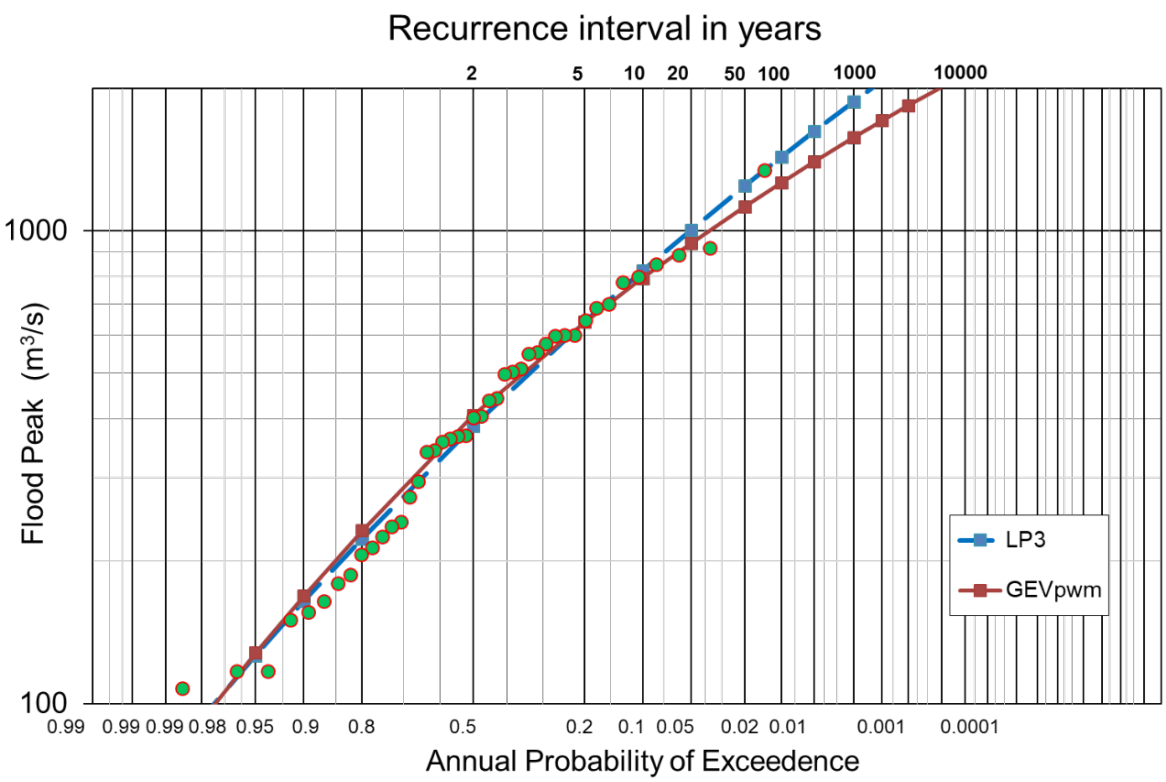


Figure 3.22. Probabilistic analysis of G1H031 annual maximum flood peaks downstream of Misverstand Dam. GEVpwm = General Extreme Value distribution and LP3 = Log-Pearson III distribution.

Table 3.8. Recurrence interval results of the probabilistic analysis

Recurrence Interval (y)	Flood Peak (m^3/s)
2	407
5	641
10	794
20	939
50	1125
100	1263
200	1399
500	1577
1000	1710

3.4.5 Long-term trends/frequency changes in observed flood peaks

We surmised that, if time-based changes might be found to be present in the statistical characteristics of G1H031 observed maximum annual and/or maximum monthly winter discharge rates, then their causes might be the gradual onset of Climate Change impacts, and/or the capturing of Upper Berg River stream flows by Berg River Dam after June 2007.

We examined the 1975-2018 time-series of annual maximum flood peaks at G1H031, depicted in Figure 3.20, for long-term increasing or decreasing linear trends and found the time-series to be effectively stationary, i.e. no statistically significant evidence of Climate Change impacts on annual maximum flood peaks in the Lower Berg River.

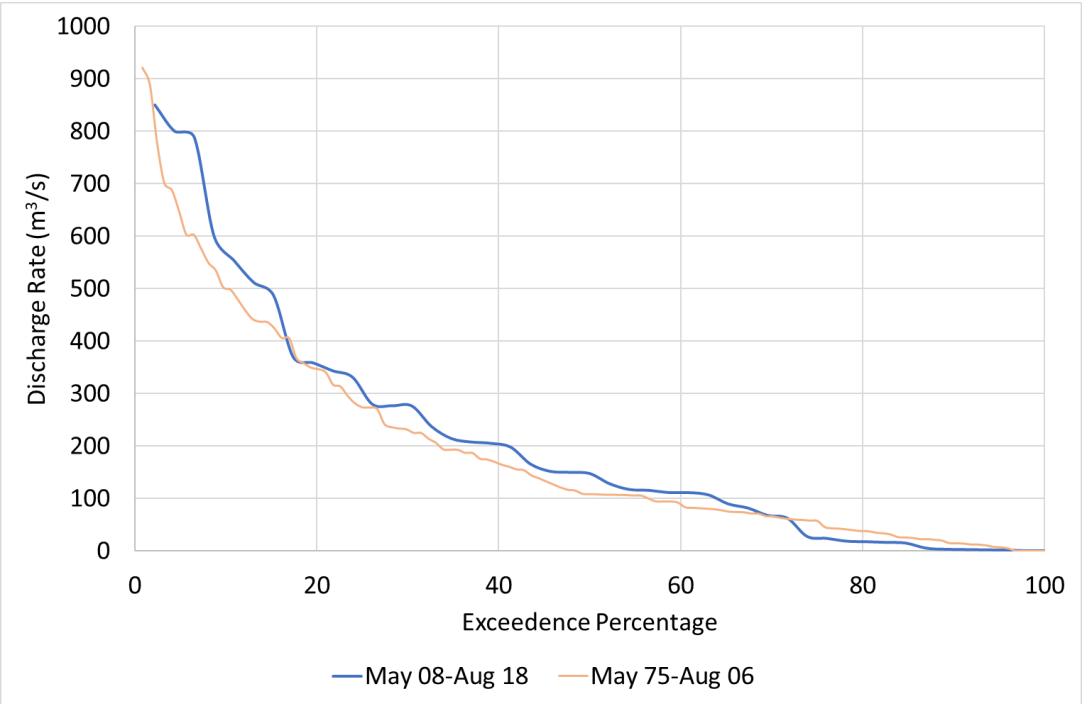


Figure 3.23. Misverstand Dam – pre and post Berg River Dam monthly winter maximum discharge rate exceedance frequencies.

Given the above finding, we then switched the focus of our analysis to possible changes in the frequency of maximum discharge rates at G1H031 during the winter months of May to August after start of storage of inflows into Berg River Dam in July 2007. Exceedance frequencies for winter month discharge maxima at G1H031 before the presence of Berg River Dam, as opposed to the winter month discharge maxima post-Berg River Dam presence, are shown on Figure 3.23.

This analysis suggests that the exceedance frequencies of G1H031 observed monthly winter maximum discharge rates after the building of the Berg River Dam were generally greater than the frequencies of equivalent pre-Berg River Dam maximum discharge rates, except during the recent drought years. This outcome is actually unexpected, but we surmise that it might be related to the quasi-decadal fluctuations evident on Figure 3.20.

3.4.6 Flood Characteristics During Drought Years

Observed maximum annual flood hydrographs at hourly resolution during the four most extreme drought years in the G1H031 record, namely during August 2003, August 2004, July 2015 and June 2017 are shown on Figure 3.24. Notable is that the G1H031 observed winter maximum monthly discharge rates during the four most extreme drought years of 2003/2005 and 2015/2017, never exceeded the 1:2-year recurrence interval of 407 m^3/s presented in Table 3.8.

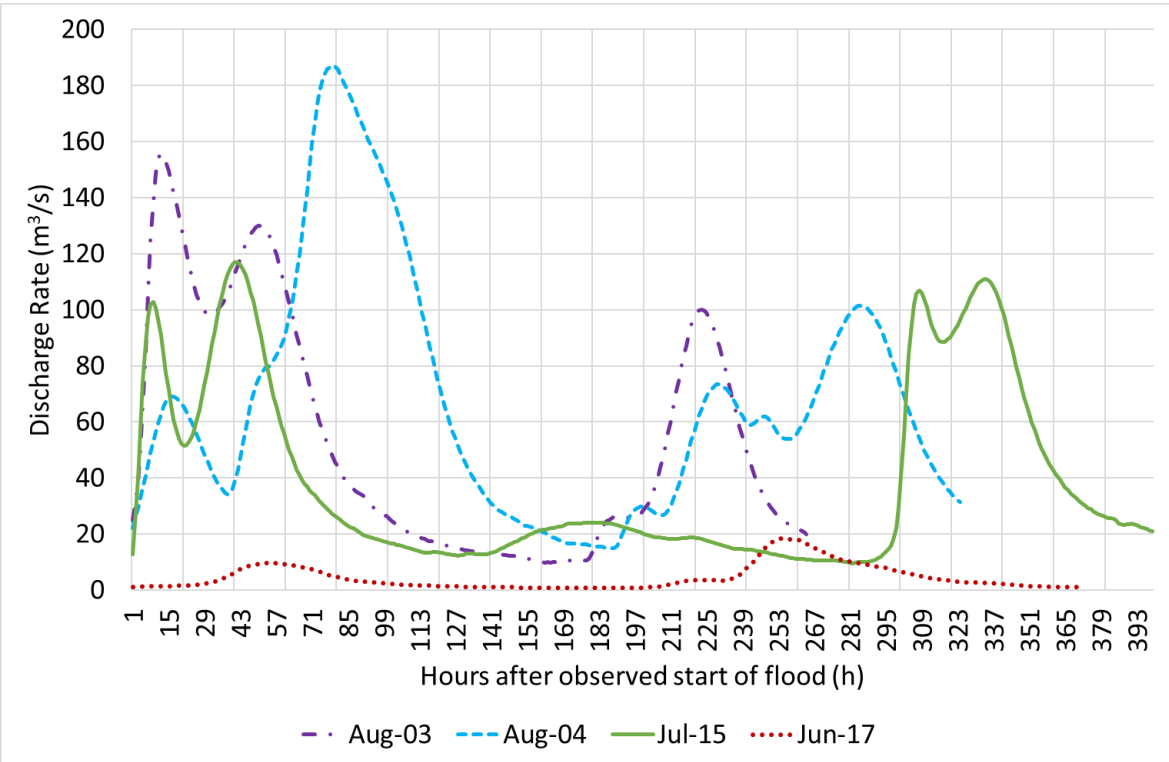


Figure 3.24. Observed maximum annual flood hydrographs during four most extreme drought years in the record at G1H031 downstream of Misverstand Dam.

3.5 Hydrology scenarios and their impacts on yield

3.5.1 Scenarios considered

The flow scenarios examined how the inflows to the estuary and how system yield (water available for sectoral use) would be affected by different EWR allocations and by climate change, taking into account planned future development to meet water demands. Scenarios involving increased allocations to the estuary to meet a higher ecological category were not modelled here, since (a) the health is predetermined and (b) changes to firm yield could be extracted from the classification study. These scenarios are therefore only revisited in the final analyses.

Hydrological flow scenarios were generated using the latest configuration of the WCWSS model for the period 1928/29 to 2016/17 (i.e. 89 years)[2]. The outputs from the hydrological analysis include an estimate of the natural MAR reaching the estuary, the impact of present day and future infrastructure on the historical firm yield (HFY) and the volume of water reaching the estuary with and without the application of the minimum EWR flow requirement. The HFY is the volume of water that can be provided on an annual basis without the system failing, i.e. the dams running empty, based on the historical observed inflows and the current average monthly demand distribution on the system. These results were required to determine the likely ecological condition and value of the estuary taking into account additional benefits such as improved water quality in both the river and the estuary. These scenarios are linked to changes in water abstractions – including infrastructure development scenarios such as the Berg River-Voëlvlei Augmentation Scheme (BRVAS) and increased re-use of treated effluent – as well as projected climate change impact scenarios.

The climate change scenarios were obtained from the climate change analysis undertaken for the City of Cape Town (Aurecon 2019). These were then used to produce potential future representations of the climate related inputs to determining the available yield from the Western Cape Water Supply System (WCWSS). Two sets of future climate scenarios were considered – sourced from the Climate Systems Analysis Group (CSAG) at the University of Cape Town and the Council for Scientific and Industrial Research (CSIR). After preliminary analysis, it was decided that the CSAG model needed further investigation, and that the study would be based on the CSIR model (Engelbrecht *et al.*, 2019). The CSIR scenario is based on their 10th percentile climate change scenario (also on the dry side), but then derived simply by adjusting precipitation (and evaporation) for each individual month in the original rainfall time series files.

[2] Note that a hydrological year runs from October to September.

3.5.2 Summary of Scenario Results

Results of the hydrological scenario analysis in terms of the impact relative to present day on the historical firm yield (HFY) and the MAR of the inflow to estuary are presented in Table 3.9.

Table 3.9. Results of hydrological scenario analysis.

Scenario	Scenario	HFY (Mm^3/a)	Percentage change from present day HFY (%)	Average Annual inflow to Estuary (Mm^3/a)	Percentage change from present day MAR (%)
Natural	Reference	n/a	n/a	912.4	98%
Present day No Estuary EWR	P0	507.8	0%	459.2	0%
Present day Min flow Estuary EWR	P1	507.8	0%	468.6	2%
Future Infrastructure No Estuary EWR	F0	528.1	4%	432.8	-6%
Future Infrastructure Min flow Estuary EWR	F1	528.1	4%	438.4	-5%
Future Infrastructure Climate Change (CSIR) No Estuary EWR	C0	479.9	-5%	303.2	-34%
Future Infrastructure Climate Change (CSIR) Min flow Estuary EWR	C1	470.2	-7%	312.3	-32%

These results suggest that while the historical firm yield does not seem to be affected by providing the Estuary low flow EWRs of 0.6 m^3/s in both the present day and the future infrastructure scenarios (P0, P1, F0, F1), the mean annual runoff to the estuary is affected, by a small percentage. This is likely due to the fact that the minimum low flow requirement of 0.6 m^3/s in the summer months does not represent a significantly large portion of the MAR and that the additional impact of imposing this in the few months when it is required can generally be compensated for across the whole system, if it is managed correctly as an integrated system.

It is also important to note that this is based on allocations made from the system and it is likely that even with the current allocations the available water that is meant to be reaching the estuary is not actually getting there. **In future it will be necessary to ensure compliance with these allocations in order to meet the minimum estuary flows**. The impact of climate change on the historical firm yield and volume reaching the estuary is much more substantial, however, particularly for the flow volume reaching the estuary. The difference in average summer (November to April) and average winter (May to October) flows compared to the present day is shown in Table 3.10.

Table 3.10. Results of hydrological scenario analysis – change in summer and winter flows.

Scenario	Scenario	Average summer flow (Nov-Apr) Mm^3/a)	Percentage change from present day summer (%)	Average winter flow (May – Oct) (Mm^3/a)	Percentage change from present day winter (%)
Natural	Reference	100.0	156%	812.4	92%
Present day No Estuary EWR	P0	36.0	0%	423.2	0%
Present day Min flow Estuary EWR	P1	45.4	26%	423.2	0%
Future Infrastructure No Estuary EWR	F0	36.0	0%	396.7	-6%
Future Infrastructure Min flow Estuary EWR	F1	45.4	26%	393.0	-7%
Future Infrastructure Climate Change (CSIR) No Estuary EWR	C0	24.5	-32%	278.7	-34%
Future Infrastructure Climate Change (CSIR) Min flow Estuary EWR	C1	33.9	-6%	278.4	-34%

The mean monthly flows to the estuary are shown in Figure 3.25 and Figure 3.26. When the EWRs are applied in Scenarios P1 and Scenario F1, the average minimum flows in summer meet the requirement of 0.6 m^3/s, however, there are still some months when this may not be the case.

The flow duration curve showing summer flows in Figure 3.27 indicates how often the minimum flow is met for the different scenarios. When no provision is made for the minimum flow, the estuary flows drop below 0.6 m^3/s more than 50% of the time.

The winter flow duration curve is shown in Figure 3.28 and shows clearly how climate change reduces the frequency of high winter flows which are essential to the functioning of the estuary.

The mean, minimum and maximum monthly flows from the 89-year simulation period (1928 to 2016) for the natural, Present Day scenario with no minimum low flow EWR (i.e. P0), future infrastructure scenario with the 0.6 m^3/s minimum EWR (i.e. F1) and climate change scenario are shown in Figure 3.29. These are shown relative to the average monthly EWRs from the RQOs. These results show how even under the natural condition there is a very high degree of variability in the flow reaching the estuary during the winter months, but that on average, both the current and future scenarios plot close to the monthly requirements from the RQOs.

During the summer months it is clear that under the present day scenario, the minimum low flow EWRs are not being met and that under the future scenario EWR releases will need to be made during the summer months to maintain ecosystem function, and effort will need to be invested to ensure that these actually reach the estuary. This is expected to have a limited impact on the total system yield, and it could be addressed through the management of the system.

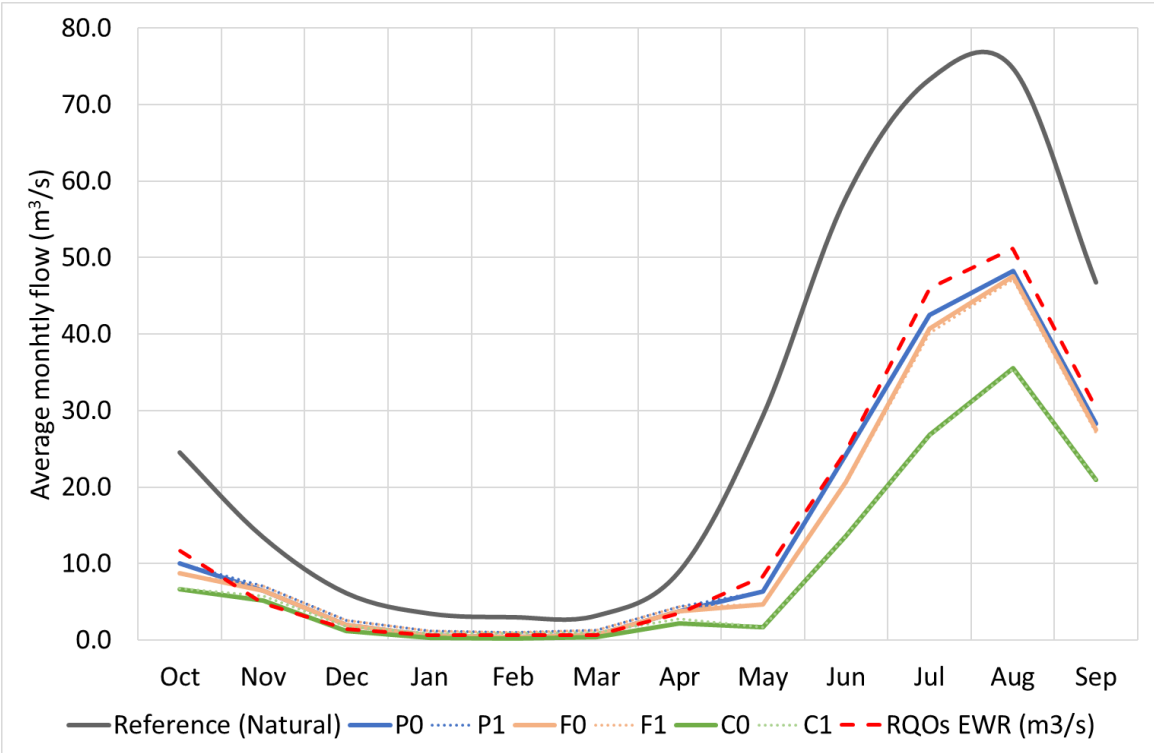


Figure 3.25. Simulated Mean monthly flows to the Berg River Estuary for all months based on 89 years of simulation (1928 to 2016) for the different scenarios considered.

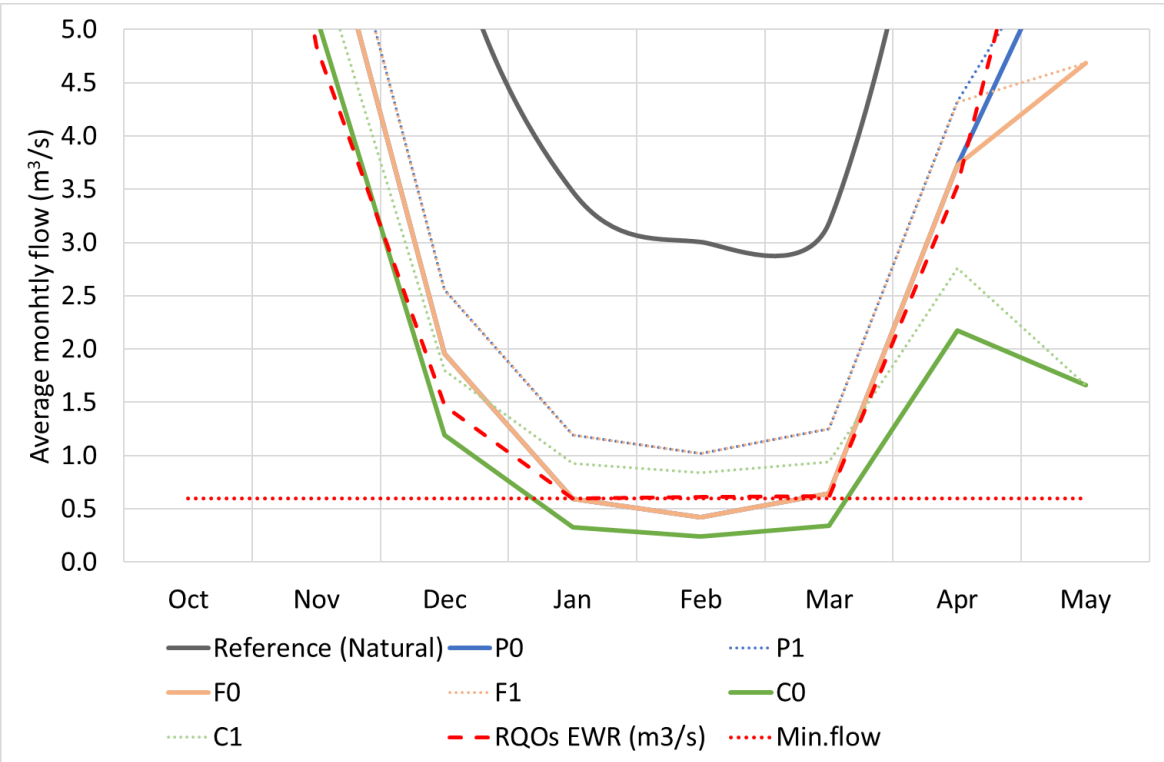


Figure 3.26. Simulated mean monthly flows to the Berg River Estuary for summer months only based on 89 years of simulation (1928 to 2016) for the different scenarios considered.

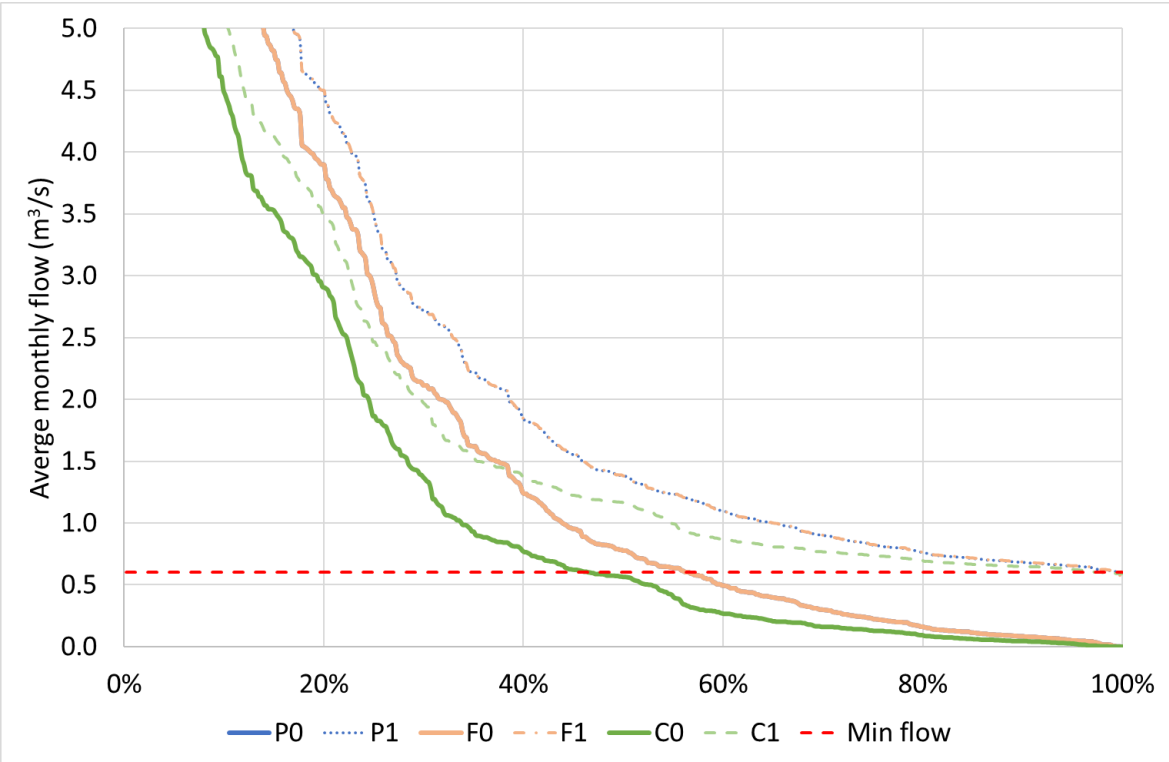


Figure 3.27. Flow duration curve for summer months (November to April) (Note that F0 plots on top of present day (P0) flows)

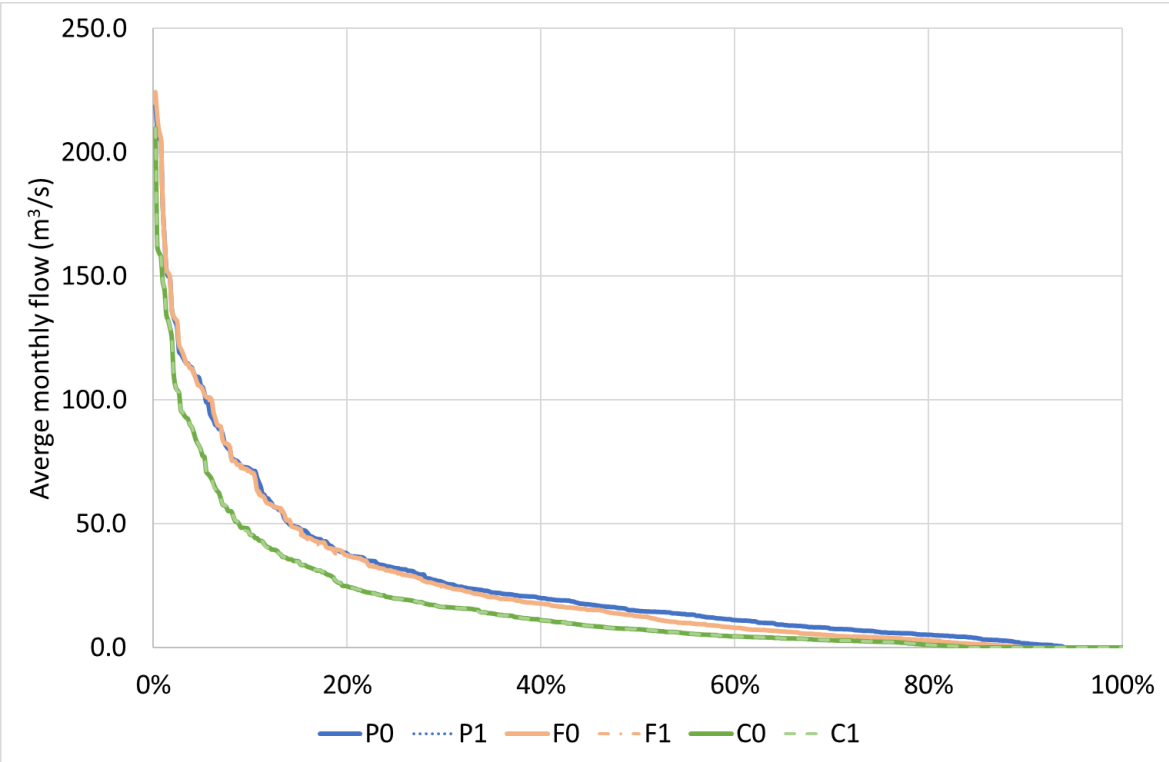
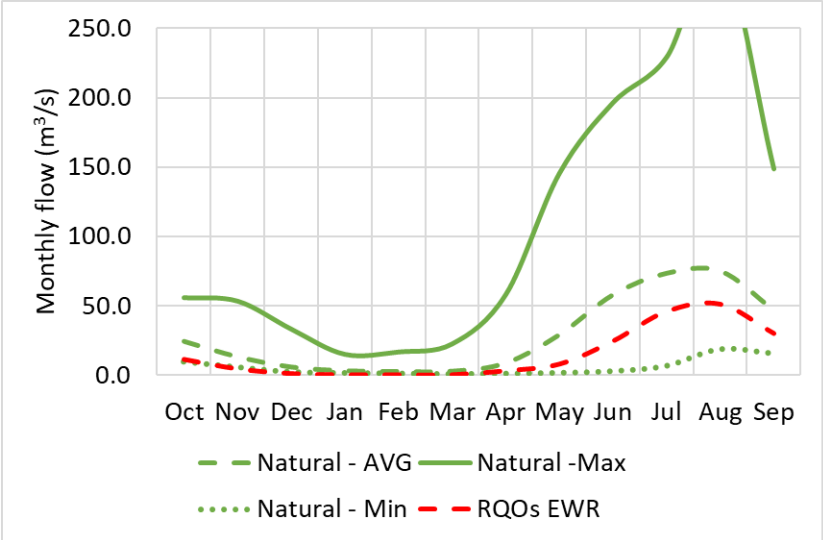
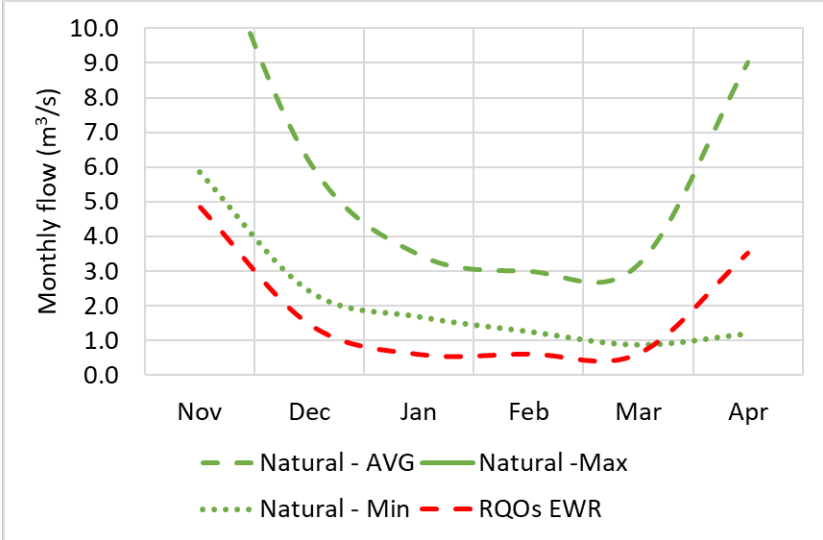
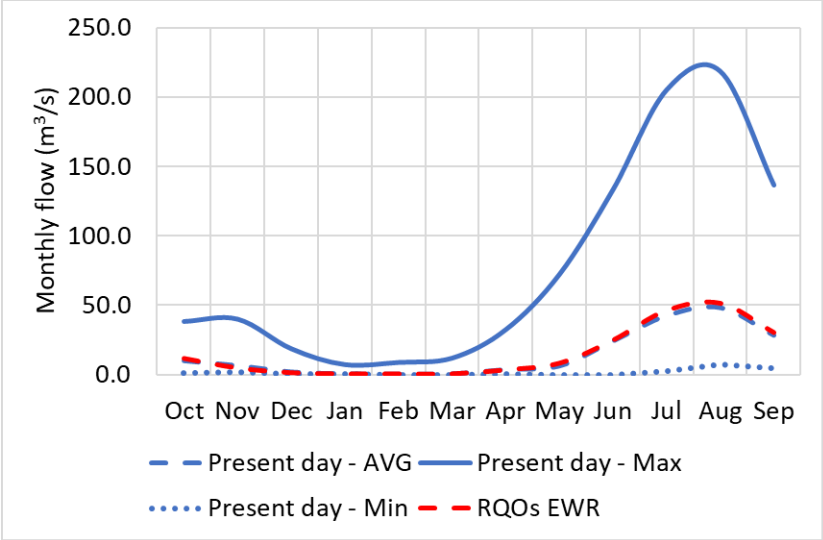


Figure 3.28. Flow duration curve for winter months (May – October) for hydrological scenarios.


250.0
200.0
150.0
100.0
50.0
0.0
Monthly flow (m³/s)
Oct Nov Dec Jan Feb Mar Apr May Jun Jul Aug Sep
Natural - AVG
Natural -Max
Natural - Min
RQOs EWR


10.0
9.0
8.0
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
Monthly flow (m³/s)
Nov Dec Jan Feb Mar Apr
Natural - AVG
Natural -Max
Natural - Min
RQOs EWR


250.0
200.0
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100.0
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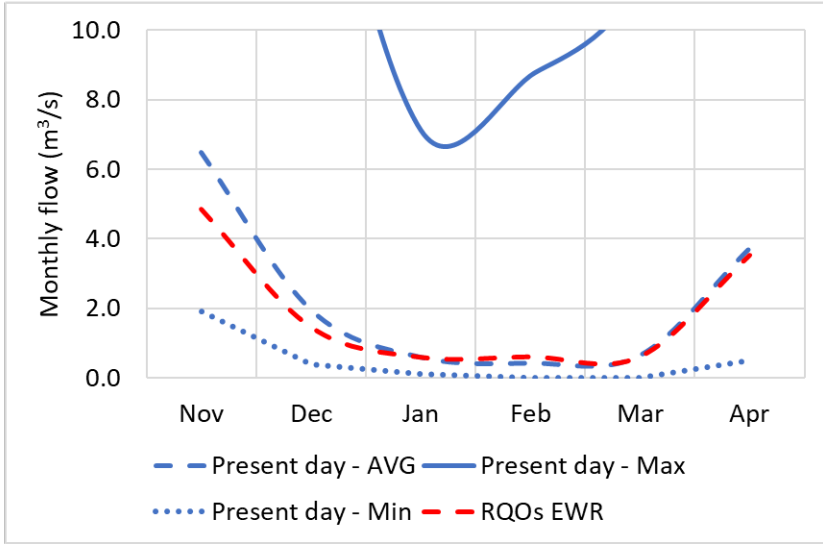
Monthly flow (m³/s)
Oct Nov Dec Jan Feb Mar Apr May Jun Jul Aug Sep
Present day - AVG
Present day - Max
Present day - Min
RQOs EWR


10.0
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Monthly flow (m³/s)
Nov Dec Jan Feb Mar Apr
Present day - AVG
Present day - Max
Present day - Min
RQOs EWR

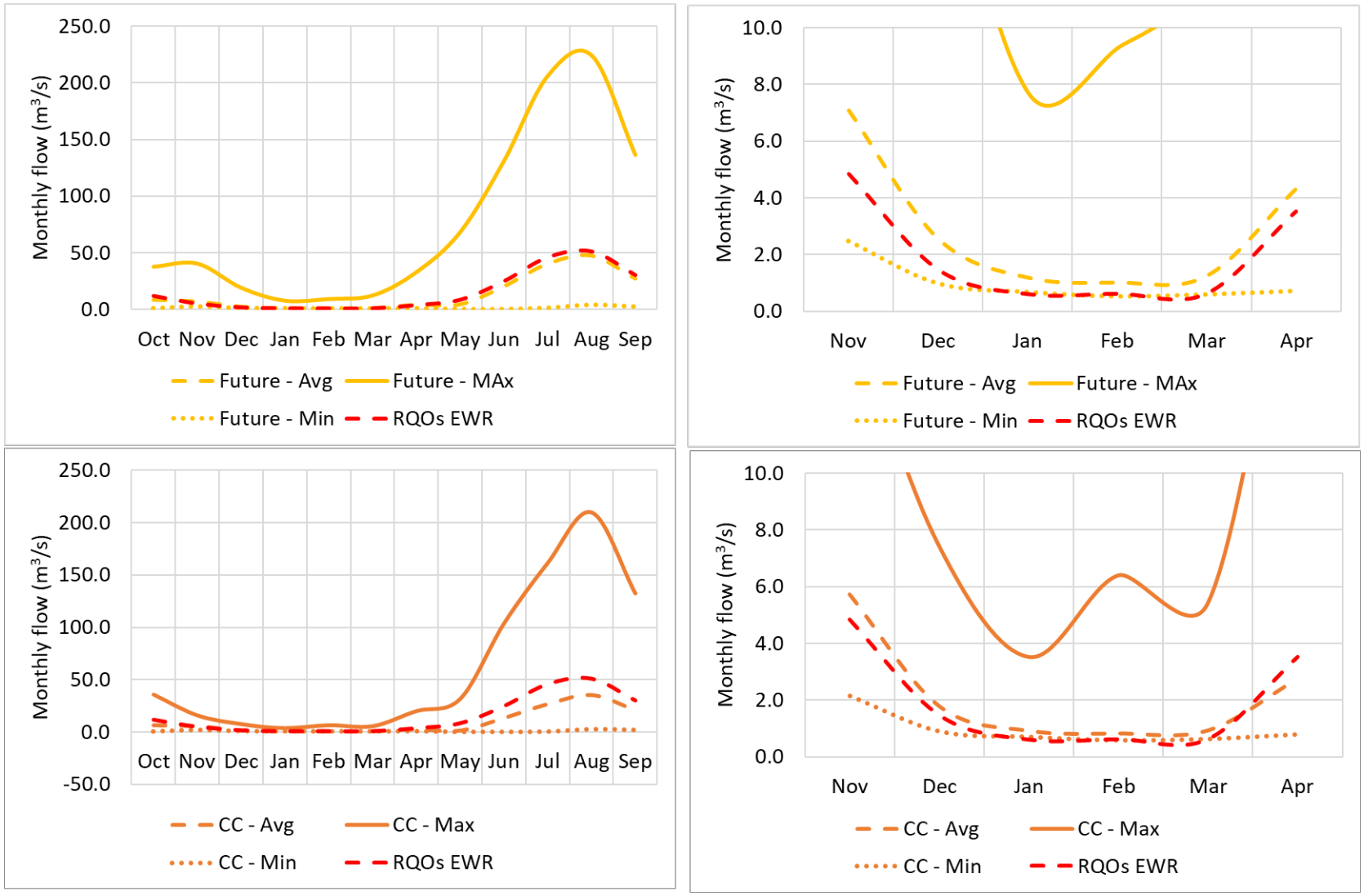

Figure 3.29. (Two pages) Monthly EWRs from the RQOs as compared to the Mean, Max and Min simulated monthly flows reaching the estuary under the natural (top), present day (second row), future (third row) and future with climate change (bottom) scenarios with the minimum 0.6 m³/s EWR applied for all months (Left) and zoomed in to show the summer months only.

3.5.3 Estuary health under different hydrological scenarios

Impacts of changing flow on estuary health[3] under Present Day condition and under the future scenarios was evaluated using the Estuary Health Index (Turpie *et al.* 2012) and the methodology for the determination of the ecological reserve for estuaries (DWA 2012). Hydrology is considered to be one of the most important components of the assessment, as it helps to establish the extent to which modification in river inflow (in terms of water volume, water quality and sediment supply) is responsible for the deviation of health from Reference Condition.

According to the methods of Turpie *et al.* (2012) the hydrology health is calculated on the basis of the extent to which current inflow patterns resemble those of the Reference state, estimated on the basis of two parameters (a) general inflow patterns, highlighting the changes in low flows, and (b) the frequency and magnitude of flood events; the relative weighting of these two parameters (60:40) being set according to their assumed importance as drivers of the estuarine system. For this study we have elected to use both "% similarity in low flows" and "present MAR as a % of MAR in the reference state" under component "a" since efforts to meet established environmental flow requirements under the various scenarios assessed in this study are focussed on meeting summer low flow requirements only, and impacts of reduced base flows experienced during the rest of the year (which are highly significant for this system) would otherwise be missed. These two components were weighed equally (30% each) as indicated in Table 3.11.

Results of this assessment suggests that the **hydrological health of the Berg River Estuary is in a very poor state** (36.3% - i.e. seriously modified). This is significantly lower than the previous assessment in 2010 (DWA 2010, health score: 72%). While there have been some real changes in the hydrological health of the system since 2010, particularly reductions in summer low flows linked to increased use of water by agriculture, we believe that this is mostly linked to the fact that an effort was made in this study to calculate the actual volumes of water reaching the estuary rather than assuming that flow released from Misverstand Dam is all reaching the estuary. Respecting the minimum summer low flow requirements that have been specified for the system will, in all likelihood, only result in a very slight improvement in hydrological health of the system (up to 38.0%). The reason for this is that while summer low flows are very important for the system winter baseflows and floods are almost equally as important and these remain more or less "as is" under the scenarios that simply meet the gazetted EWRs. (Note that further scenarios in which a larger component of MAR is allocated to achieve a higher level of estuary health are considered in the synthesis chapter. In these cases, the health is pre-set and the additional water requirement would roughly equate to a further loss in system yield).

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3 Health in this context refers to the extent an estuary's state, characteristics and functioning differ from its pristine (= "reference") condition.

Under the future development scenarios F0 and F1 hydrological health declines slightly further (F0 – 35.5%), but is clearly better if low flow EWRs are respected (F1 – 40.8%). Incorporating the effects of climate change is expected to have a severe impact on the hydrological health of the Berg River Estuary, with scores dropping to 22.9% without any protection of the low flow EWRs and less (28.7%) if low flow EWRs are respected.

Table 3.11. Present MAR as % of Ref. MAR and Present Low flows as % of Ref. Low flows.

	Ref	PD (2020)	P1	F0	F1	C0	C1
			PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
MAR (Mm^3/a)	912.4	459.2	468.6	432.8	438.4	303.2	312.3
Present MAR as % of Ref. MAR	100.0%	50.3%	51.4%	47.4%	48.0%	33.2%	34.2%
Low flows (Mm^3/m): Jan-Mar	8.4	1.5	1.9	1.5	3.0	0.8	2.4
Present Low flows as % of Ref. Low flows	100.0%	17.3%	22.1%	17.3%	35.9%	9.5%	28.0%

Table 3.12. Percentage similarly in the frequency of floods.

Flood class (m^3/s)	Ref	PD (2020)	P1	F0	F1	C0	C1
			PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
100-200	160	65	65	62	60	24	25
200-300	60	22	22	23	22	6	5
300-500	39	16	16	16	16	1	1
500-800	7	4	4	4	4	0	0
>800	1	0	0	0	0	0	0

Table 3.13. Change in hydrology functioning score.

	Wt	Hist. (2010)	PD (2020)	P1	F0	F1	C0	C1
				PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
1a. Present MAR as % of Ref. MAR	60	68	50.3	51.4	47.4	48.0	33.2	34.2
1b. Present Low flows as % of Ref. Low flows			17.3%	22.1%	17.3%	35.9%	9.5%	28.0%
2. % Similarity in mean annual frequency of floods	40	79	40.0%	40.0%	40.2%	39.1%	25.1%	25.1%
Health score = weighted mean of 1 and 2		72	36.3	38.0	35.5	40.8	22.9	28.7

4 HYDRODYNAMICS & SEDIMENTS

4.1 Introduction

The Berg River Estuary is one of three permanently open estuaries on the West Coast of South Africa. The estuary system empties into St Helena Bay, 140 km north of Cape Town and some 285 km downstream from the Berg River source in the Groot Drakenstein and Franschhoek mountains. The lower reaches of the main channel are dominated by the marine tidal influences of St Helena Bay, while the upper reaches are freshwater dominated. The Berg River Estuary is a somewhat unusual South African system because it has a very shallow gradient, rising <1 m in 50 km. The estuarine system itself extends some 70 km inland from the mouth, and includes an extensive floodplain covering an area of 61-69 km^2 that comprises of ephemeral pans, salt marshes and intertidal mud flats (Figure 4.1; Day 1981, Ratcliffe 2007, DWAF 2007, Schuman 2007, Beck & Basson 2007, van Ballegooyen 2010). The main channel of the estuary is 100-200 m wide at Velddrif (4 m deep), becoming narrower and shallower further upstream (150 m wide and 3 m deep on average; van Ballegooyen 2010, Taljaard *et al.* 2010).

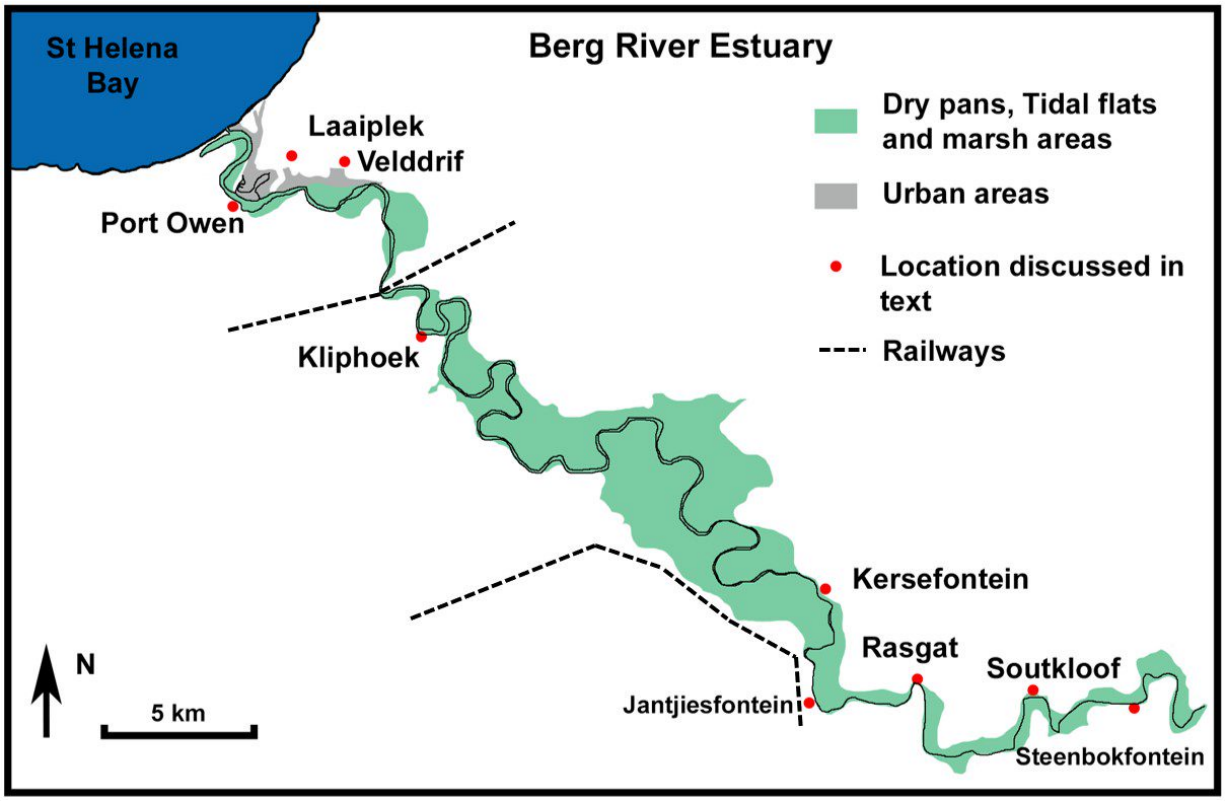

Figure 4.1. Map of the Berg River Estuary (adapted from Schumann 2007 and van Ballegooyen 2010).

The hydrodynamic functioning of the Berg River Estuary is affected by a number of key drivers. The most important of these are:

- Seasonal variation in rainfall and temperature;

- Volume of freshwater entering the system;
- Sea level and tides; and
- Channel morphology.

The relative importance of these drivers in the Berg River Estuary has been examined in a number of flooding and water quality hydrodynamic models, including work by Beck & Basson (2007), van Ballegooyen (2010) and Basson (2014). Other relevant studies on the estuary include the bathymetric surveys and modelling studies by the CSIR (CSIR 1993, Taljaard & Slinger 1992, Taljaard *et al.* 1992, Slinger & Taljaard 1994 and 1996, Slinger *et al.* 1996 and 1998) and those undertaken as part of the Berg River Baseline Monitoring Programme (Schumann & Brink 2009, Schumann 2009).

This chapter describes how these key drivers have shaped the hydrodynamic functioning and sediment dynamics of the Berg River Estuary, past, present and future. The chapter explores the interrelationships between floods, tidal forces and sediment and how these have been affected by changing hydrology and other anthropogenic interventions on and around the estuary.

4.2 Hydrodynamic drivers

The hydrodynamics of the Berg River Estuary are complex, owing to the variation in inflow conditions between winter and summer, the complex bathymetry, the gentle gradient (rising <1 m in 50 km) and the length of the estuary (van Ballegooyen 2010). The efficacy of circulation in the estuary is also influenced by the magnitude and extent of the longitudinal density gradient i.e. how the freshwater inflow from catchment meets and interacts with the saline water from the ocean. In turn, the magnitude of vertical mixing is influenced by river flow, the strength of tidal flows, water depth, the nature of the bottom and the strength of local winds and local air-sea interactions (Largier *et al.* 2000).

4.2.1 Seasonal variation in rainfall and temperature

The Berg River experiences Mediterranean-type climate, with warm dry summers (Nov-Feb) and cool wet winters (May-Aug). The rainfall is mainly of a cyclonic nature originating over the Atlantic Ocean (Taljaard *et al.* 2010). Frontal systems encountering the mountain ranges of the Upper Berg Catchment are forced upwards causing orographic rainfall (i.e. rain that is produced from the lifting of moist air over a mountain). More than 80% of the rain falls in winter and most falls in the upper catchment (Tyson 1986, Taljaard *et al.* 2010, van Ballegooyen 2010).

The Berg River Estuary therefore has a strongly seasonal hydrological regime linked to rainfall. During summer, river inflow to the Berg River Estuary is low (or non-existent) and sea water penetrates up the estuary so that the system becomes marine dominated. The direct influence of flood tide seawater intrusion is up to Kliphoek, 14 km from the mouth. During winter, the river flows strongly and the estuary is river-dominated with low salinity throughout (Schumann 2009, Taljaard *et al.* 2010, van Ballegooyen 2010). River floods also typically occur during the winter months, with the extent of inundation of the floodplains adjacent to the estuary determined by the severity of the flood (Huizinga *et al.* 1994).

Seasonality in air temperature and evaporation also play a role in the hydrodynamic functioning of the estuary. Air temperature shows a strong seasonal pattern, with maximum

daily air temperatures in summer frequently exceeding 35°C, and minimum temperatures consistently above 15°C. In winter, daily maxima are generally below 20°C, while minima are frequently below 10°C (Schumann 2007). Combined with the wind, air temperature is an important determinant of the rate of evaporation. In general, the mean annual potential evaporation (MAPE) in the Berg River exceeds 2000 mm/a, while in the southern parts of the catchment the MAPE is lower at 1500 mm/a (van Ballegooyen 2010). During summer, monthly evaporation losses are approximately 250 mm/a, dropping to 50 mm/a in winter (Ratcliffe 2007, van Ballegooyen 2010). While the effects of evaporation are irrelevant in areas that are drained after a flood event, evaporation is of particular importance to the ephemeral pans, which may be flooded on several occasions during winter (Taljaard *et al.* 2010). When filled, each pan has a characteristic depth and spatial extent that decreases over time due to evaporation (typical evaporation rates of around 200 cm/p.a., with about 40% occurring in the summer) (Taljaard *et al.* 2010). This rate of evaporation and resultant change in water depth and extent of the pan is unique to each pan.

Drought conditions are likely to extend the 'summer' conditions, with low freshwater inflows, saline intrusion extending further upstream, and pans being dryer than normal.

4.2.2 Volume and characteristics of freshwater flow into the estuary

Seasonal variation in freshwater inflows from the catchment strongly influence the hydrodynamic functioning of the estuary. It influences both water level (i.e. high flow rates result in the channel bursting its banks and inundating the surrounding floodplains) and flushing (Largier *et al.* 2000).

Freshwater inflow serves to flush out sea water which penetrates into the lower reaches of the estuary due to tidal influences (van Ballegooyen 2010). The Berg River Estuary is unusual in that there is little stratification in the system (some stratification does occur in lower reaches during higher flows) and as such, shows little classical estuarine circulation (where the difference in density between seawater and freshwater drive the circulation). As a result, saline waters are relatively easily flushed from the estuary, especially in the upper and middle reaches of the system when the river flow is strong enough to push the existing waters out of the estuary and replace these with new river water (Taljaard *et al.* 2010, van Ballegooyen 2010). Freshwater spates exceeding 140 m^3/s are sufficient to fully flush the estuary of saline water (Schumann 2009).

The volume and characteristics of freshwater flow is a significant driver of water level variability in the Berg River Estuary, particularly in the uppermost reaches (Taljaard *et al.* 2010). Here, there is a strong relationship between water level and magnitude of winter baseflow (Beck & Basson 2007). Water levels in the upper reaches of the estuary increase with both increasing baseflows as well as for freshettes and floods (Beck & Basson 2007, Taljaard *et al.* 2010). At freshwater inflows of above 80 m^3/s, the water level rises to such an extent that the estuary breaks its banks in these upper reaches, and flows out onto the adjacent floodplain (Taljaard *et al.* 2010).

Drought conditions would result in a lower freshwater inflow at the estuary head, with a lower 'flushing' efficacy and a lower likelihood of flow rates great enough to result in flooding and inundation of the surrounding floodplains.

4.2.3 Sea level

The marine exchange is driven mostly by changes in water level, with only a limited influence of processes associated with density differences between fresher estuarine waters and the ocean (van Ballegooyen 2010). The tidal effects in the Berg River Estuary are directly linked to the tides of St Helena Bay, which are characterised, as is the rest of South Africa, by a semidiurnal tidal cycle, with two nearly equal high tides and low tides every day. Hughes *et al.* (1991) demonstrated that there had been a sea level rise of roughly 1.2 mm/year, consistent with other tectonically stable world trends at the time, however, more recently, a regional sea level rise of +1.87 mm/year (1959–2006) has been recorded for the West Coast of South Africa (Mather *et al.* 2009). Other mechanisms influencing water circulation in the system include the effects of subtidal water level variations in the adjacent ocean, intrusions of cold upwelled oceanic waters and local wind stress (Taljaard *et al.* 2010).

While water depth varies considerably along the length of the estuary, with wide tidal flats with very shallow areas, water depth in the lower reaches of the estuary is driven by the tide. Towards the mouth of the estuary, the tidal water level variations and flows are strong. Tidal range at the mouth is 0.5-1.5 m, compared to 0.2-0.8 m in the middle of the estuary and less than 0.2 m in the upper estuary (Taljaard *et al.* 2010). Tidal flows in the lower estuary (at the R27 bridge) are 50-100 m^3/s and 200-300 m^3/s during neap and spring tides, respectively (Beck & Basson 2007). Intertidal areas (i.e. areas that are exposed and covered by outgoing and incoming water linked to the tides) occur mainly downstream of the Railway Bridge (Taljaard *et al.* 2010).

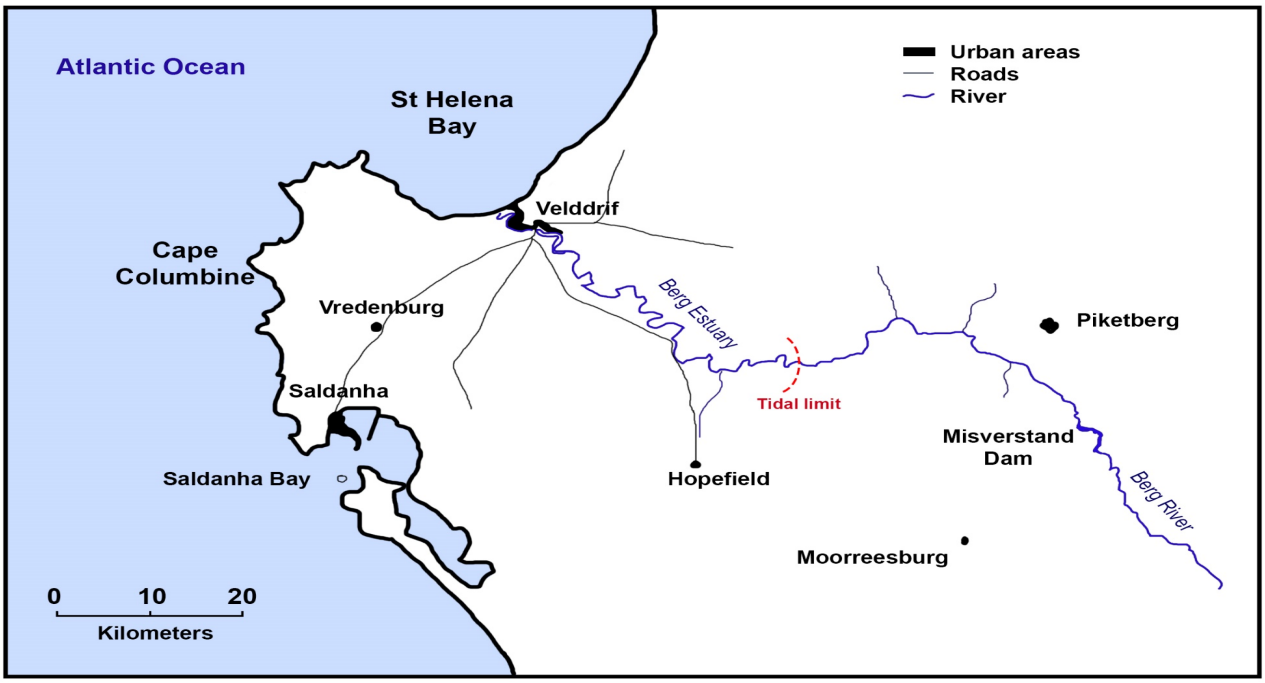

Figure 4.2. The Berg River and Estuary in relation to St Helena Bay and Cape Columbine, showing the approximate tidal limit and the mouth.

Tidal action decreases rapidly upstream, and the lags in tidal variations along the estuary are predominantly due to friction (Taljaard *et al.* 2010, van Ballegooyen 2010). Between the mouth and Kliphoek, approximately 15 km upstream, a reduction in tidal amplitude of between 65% and 56% occurs for spring and neap tides, respectively (Taljaard *et al.* 2010). A slight amplification in tidal variation takes place between Kliphoek and Kersefontein, 46 km upstream of the mouth, and while the mechanism for this is not well understood, such amplifications in tidal variation are known to occur in estuaries and are usually caused by funnelling as the estuary becomes shallower and narrower with distance upstream (Taljaard *et al.* 2010). Just upstream of Kersefontein the estuary narrows, as does the adjacent floodplain, and the tidal amplitude decreases rapidly — here the tidal amplitude ranges between 11% (spring tide) and 17% (neap tide) of that measured at the mouth of the estuary (Taljaard *et al.* 2010). By Steenbokfontein, 70 km upstream, the tidal amplitude drops to between 4 and 10% of that measured at the mouth of the estuary (Taljaard *et al.* 2010). In the upper reaches beyond the tidal influence, water levels are driven by freshwater inflows should they be of any significance (i.e. winter base flows or greater).

The canalised entrance channel ensures a relatively unconstricted exchange of water between the estuary and the adjacent ocean. The tidal levels immediately inside the mouth are the same as the predicted tidal variation at Saldanha Bay, which implies that the present mouth does not reduce water level variation (Taljaard *et al.* 2010). There is evidence, however, that the new mouth has impacted tidal flows at the mouth, disrupting sediment transport processes.

Changes in water level due to tides are not the only water level variations of importance in the Berg River Estuary. Subtidal water level fluctuations are observed along the entire length of the estuary, with water level variability both at tidal periods and at periods longer than the diurnal tidal period. While there is a clear two-week spring-neap cycle in the estuary, these longer-period fluctuations serve to alter the extent of the tidal influence by changing the background water level (Taljaard *et al.* 2010). These subtidal water level fluctuations have typical amplitudes of 15-20 cm, but extend up to 30 cm or more, on occasion (Taljaard *et al.* 2010, van Ballegooyen 2010). These subtidal water level fluctuations are associated with subtidal changes in water level in the adjacent ocean (coastally trapped waves and more local weather effects) that propagate upstream in the Berg River Estuary (Shillington 1984, De Cuevas *et al.* 1986, van Ballegooyen 2010). Negative fluctuations in subtidal water levels in St Helena Bay (i.e. at the mouth of the estuary) are associated with south to south-easterly wind conditions offshore, and associated upwelling of cold waters, while positive fluctuations are associated with north-westerly onshore winds and downwelling (and to a lesser degree, waves). These longer-period oceanic water level variations are often more noticeable in the upper reaches of the Berg River Estuary than tidal variations (Schumann & Brink 2009, van Ballegooyen 2010). These long period waves influence water level variations as equivalent to that produced by neap tides (Taljaard *et al.* 2010).

The significance of the above observations is that past studies have suggested a very strong relationship between the extent of flooding and the magnitude of winter base flows as well as subtidal water level fluctuations at the mouth of the estuary (Ratcliffe 2009, Beck & Basson 2007, van Ballegooyen 2010). In fact, if explicit cognisance is taken of the rapid downstream decrease in water level increases associated with winter base flows, there is only a weak relationship between base flow and the extent of flooding in all but most upstream reaches of the estuary and then only for smaller floods (<200 m^3/s) (van Ballegooyen 2010). Under small flood conditions (100-200 m^3/s), subtidal water level fluctuations are expected to influence

flood extents from the mouth to approximately 30-40 km upstream (van Ballegooyen 2010). Under large flood conditions, it is expected that the influence of subtidal fluctuation will be limited to the region downstream of the Railway Bridge (about 10 km from the mouth, van Ballegooyen 2010). Under non-flood conditions, subtidal water level variability is expected to significantly influence that area of the estuary where normal tidal water level variability dominates over that associated with freshwater inflows, i.e. typically downstream of Kersefontein during winter base flow conditions but as far upstream as Steenbokfontein (approximately 70 km upstream) during the low flow conditions of summer (van Ballegooyen 2010).

Water level data for the upper and lower estuary are presented in Figure 4.3. While the water level is generally deeper in the upper reaches (i.e. at Jantjiesfontein) than in the lower reaches (i.e. at Kliphoek), this can be attributed to channel bathymetry — the channel is deeper at Jantjiesfontein, and wider and shallower at Kliphoek. However, it is also evident that there are greater water level fluctuations further upstream than in the lower reaches, due to the impact of floods. There is a "gap" in these flood peaks in 2016-2019 (Figure 4.3) i.e. during the drought period.

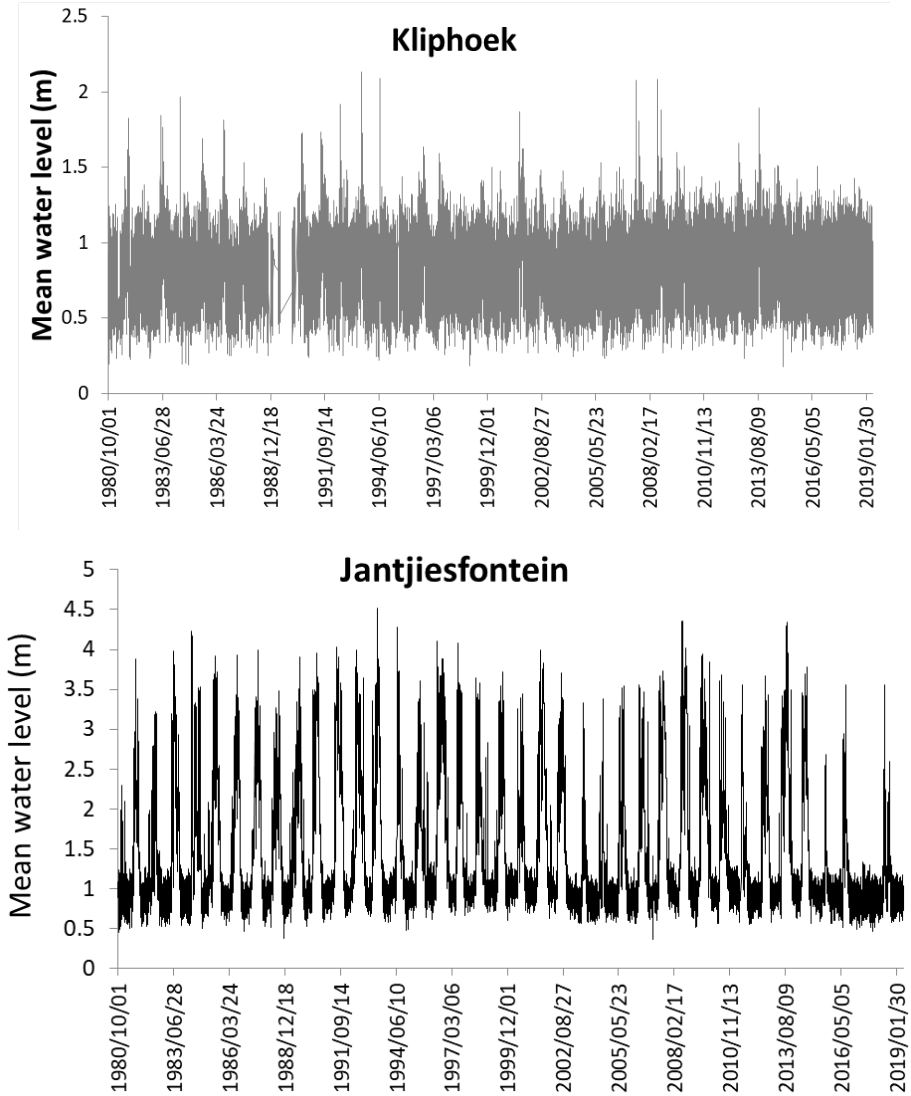

Figure 4.3. Hourly measured water level at Kliphoek (lower reaches) and Jantjiesfontein (mid to upper reaches) from January 1980 to Januray 2019 (DWS 2019).

4.2.4 Channel and floodplain morphology

The main channel at Velddrif is about 100-200 m wide but becomes progressively narrower and shallower upstream. Depth is about 3-5 m on average but extends up to 9 m in places (Beck & Basson 2007, Taljaard *et al.* 2010, van Ballegooyen 2010).

In the uppermost 15 km, the Berg River Estuary is bounded by steep banks covered in riparian woodland (van Ballegooyen 2010). Downstream, the estuary is flanked by a floodplain that varies in width from 1.5 to 4.0 km in the middle reaches, to <1.5 km in the lower reaches (Figure 4.4) (van Ballegooyen 2010). This extensive floodplain covers an area of 61-69 km^2 and is an integral part of the estuarine ecosystem (Beck & Basson 2007). Along with the shallow gradient, this extensive floodplain makes the Berg River Estuary atypical in relation to most South African estuaries (Schuman 2007).

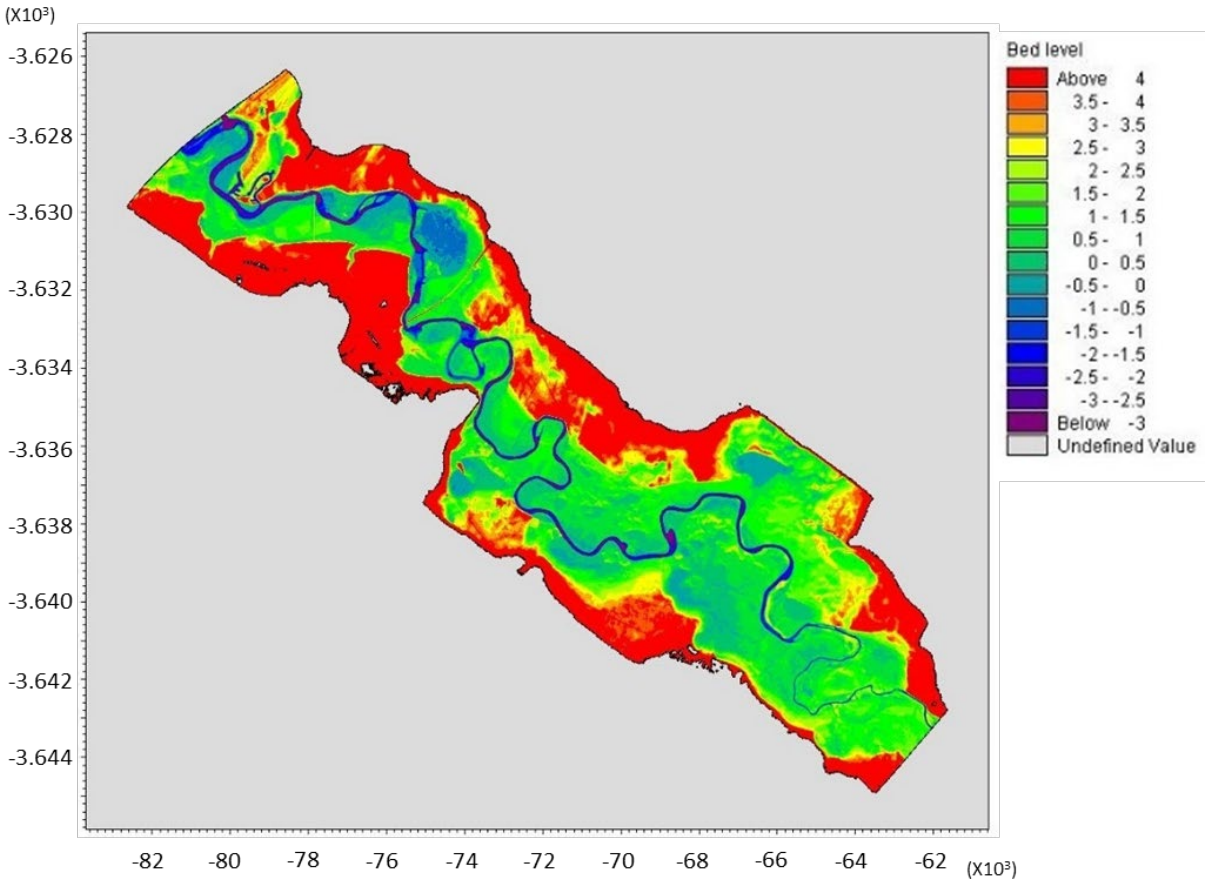

Figure 4.4. Bathymetry of the Berg River Estuary and topography of the Berg River Estuary flood plain (Beck & Basson 2007).

Freshwater inflows of above 80 m^3/s result the estuary breaking its banks in the upper reaches and flowing onto the adjacent floodplain (Taljaard *et al.* 2010). These periodic floods restrict both the build-up of riverine sediment in the estuary and the ingress of marine sediment into the system at the mouth, or at least flush these out on a regular basis (Beck & Basson 2007). The inundation of the estuarine floodplain, the extent of inundation of pans and the duration for which they remain under water are important for the health and abundance of the birds

that use these pans for roosting, feeding and breeding (Velasquez *et al.* 1991, Hockey *et al.* 1992).

Factors that influence the extent of flooding include (Beck & Basson 2007, Schumann 2007):

- the general extent of inundation of the flood plain at the start of a flood;
- the tidal state (which influences the inundation in the mid to lower reaches of the estuary); and
- non-tidal sea level fluctuations entering the estuary.

Prior inundation is influenced by characteristic winter base flow water levels and the magnitude and timing of prior flood events. Under present day conditions, base flows occurring between June and July inundate about 34.7% of the floodplain on average (Basson 2014). Assuming a floodplain area of 60 km^2, this would result in a typical floodplain inundation of 20.8 km^2 between June and July on average.

Flow depths over the floodplain during spring high tide are generally less than 1 m, except on De Plaat mudflat (8 km from estuary mouth), where some areas are deeper, and flow depths can rise to 1.5 m (Beck & Basson 2007). Velocities are typically very low over the floodplain, generally less than 0.3 m/s, with higher velocities (>1 m/s) occurring only in the main channel downstream of the Railway Bridge (~ 11 km from the mouth) during the rising tide (Beck & Basson 2007). Simulations suggest that between 87 and 76% of the total flood peak discharge flows through the fixed mouth for the 1:50 year and 1:100-year floods, respectively (Basson 2014). The rest of the flow is across the berm at the beach (Basson 2014).

Floodplain inundation is seasonal and linked to winter base flows in the upper reaches. The lower reaches of the estuary are tidal dominated, and as such floods have a relatively small impact on the area of inundation (Beck & Basson 2007). Under "low" winter base flows (<12 m^3/s), there is a small reduction (2-3%) in overall flood extent between the same size floods for typical winter base flows (~35 m^3/s) (Figure 4.5) (van Ballegooyen 2010). For small floods (100 m^3/s), the flooded area in the upper reaches of the estuary is approximately 3%, and in the mid reaches approximately 2% greater for a "high" winter base flow than for a "low" winter base flow conditions (van Ballegooyen 2010) (Figure 4.5). For floods of 200 m^3/s, the flooded area in the upper reaches is approximately 1%, in upper mid reaches approximately 3%, and in the lower mid reaches approximately 4% greater under high winter base flow than for the low winter base flow conditions (van Ballegooyen 2010). For floods of 300 m^3/s the flooded area in the upper reaches is approximately 1%, in upper mid reaches approximately 3%, and in lower mid reaches approximately 2%, greater for the high winter base flow than for the low winter base flow simulations (van Ballegooyen 2010). The differences in flood extents for flood sizes greater than this, is less than 2-3% between high and low winter base flow conditions (van Ballegooyen 2010).

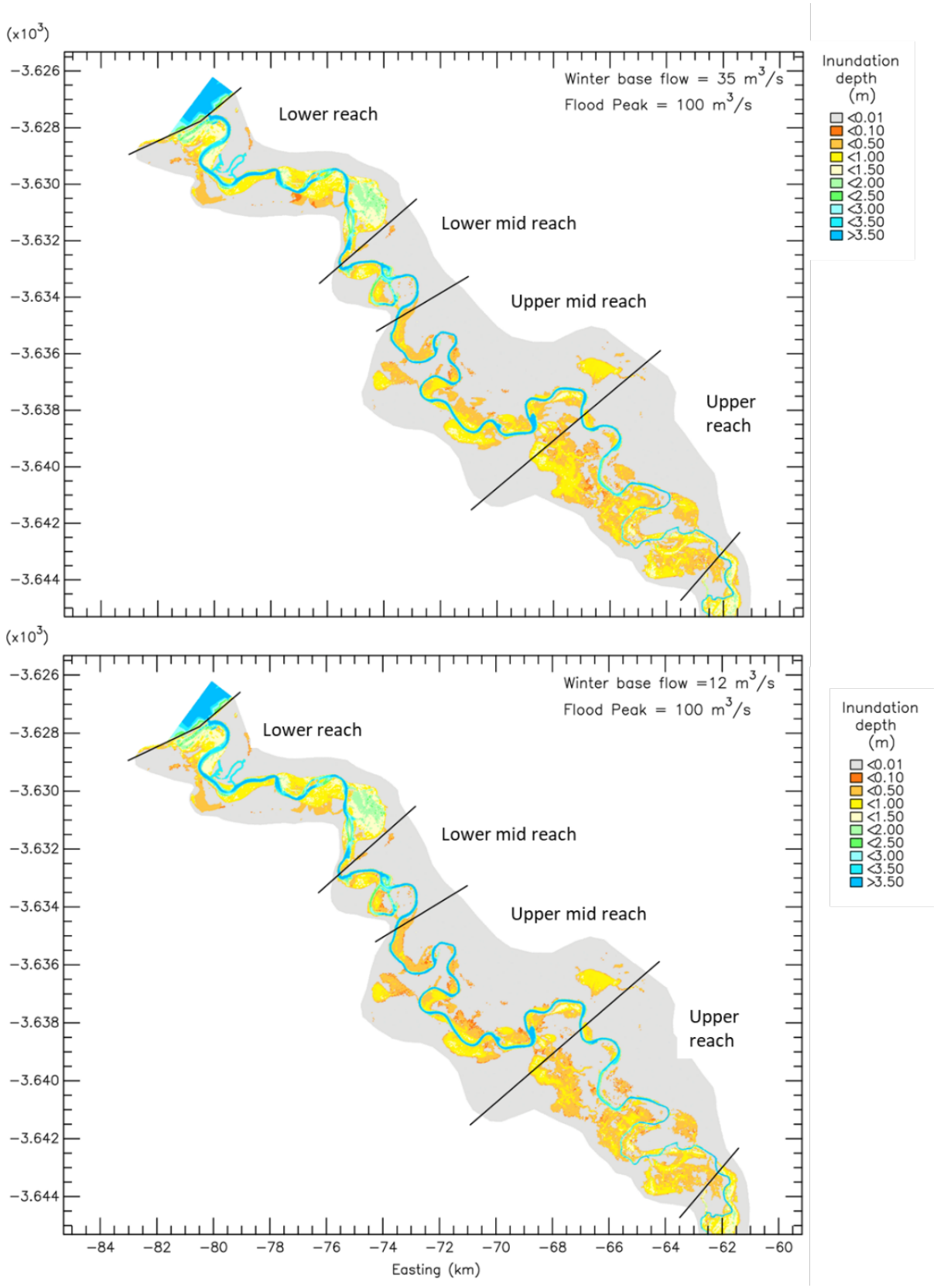

Figure 4.5. Depth of inundation of the Berg River floodplain for a 100 m³/s flood under winter base flow conditions of (top) 35 m³/s and (bottom) 12 m³/s (van Ballegooyen 2010).

The flooding regime is also significantly influenced by upstream water levels at the start of a flooding sequence, as well as subtidal water level fluctuations at the mouth of the estuary (Beck & Basson 2007). The influence of this is, however, not consistent across the length of the estuary (Figure 4.6). For example, under a flood with a peak discharge of 290 m^3/s and an initial water level of 1.8 m, the maximum area of inundation was estimated to be 26.4 km^2 in the upper reaches, while a larger large flood with a peak discharge of 893 m^3/s, at starting water level of 1.45 m, yields a similar 23 km^2 area of inundation (Beck & Basson 2007). Similarly, the influence of subtidal water levels on the flood extents is not the same throughout estuary, although water level changes due to subtidal water variability at the mouth of the estuary are observed to remain unattenuated throughout the estuary.

During floods, sediment is scoured from the banks and floodplains, and deposited towards the middle of the channel. During floods, the main channel tends to become slightly wider and shallower. Therefore, during floods, the river channel tends to widen to accommodate these floods, and afterwards the channel becomes narrower again (Beck & Basson 2007).

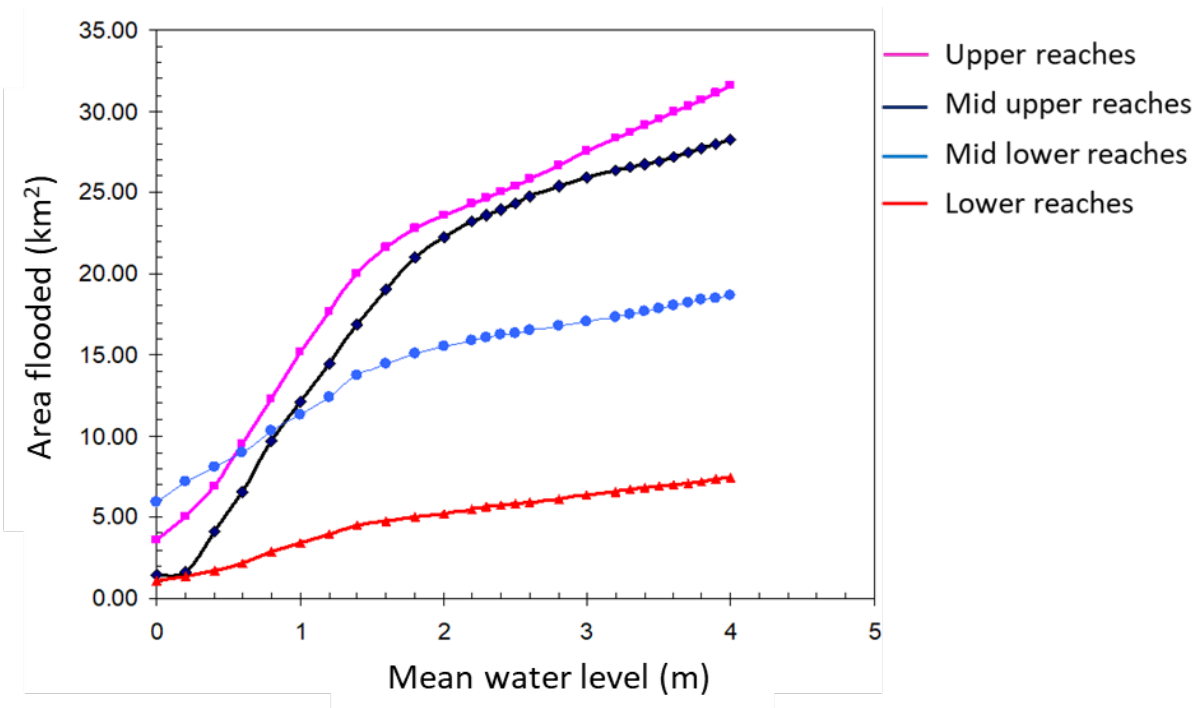

Figure 4.6. Area of flood plain that would be inundated as a function of mean water level in the various reaches of the Berg River Estuary (van Ballegooyen 2010).

Anthropogenic developments such as the modification of the mouth area and the construction of bridges have also altered the geographic extent of the estuary and the floods therein. Historically, the estuary terminus flowed parallel to the coast, with the mouth emptying at the southwestern end of an extended coastal berm (Figure 4.7). There were large floodplains inland of the mouth, which appeared to be regularly inundated by tidal and/or fluvial action (Figure 4.7; Beck & Basson 2007).

Figure 4.7. **Historical photographs of the Berg River Estuary mouth, in 1942 (with the natural mouth) and in 1976 (after canalisation and construction of the artificial mouth). Note the new, permanent mouth and 'blind arm' parallel to the coast in the 1976 image. Source: Chief Surveyor General.**

In 1969, the estuary mouth was artificially shifted one kilometre to the northeast through the coastal dunes to connect the estuary directly to St Helena Bay (Figure 4.7). This was undertaken because gradual sedimentation had caused the development of a sand bar at the natural mouth, which could not be crossed by large boats except at high tide, and could never be crossed during bad weather (Taljaard *et al.* 2010, van Ballegooyen 2010). This new canalised mouth allows a relatively unrestricted exchange of water between the estuary and the adjacent ocean and permits fishing vessel access to Laaiplek and Velddrif. The mouth is dredged to allow purse-seiners to move in and out during any the state of the tide (Schumann 2007, Taljaard *et al.* 2010). The remains of the former mouth channel now form a so-called 'blind arm' to the west of the permanent mouth and parallel to the coast (Figure 4.7). The construction of this new mouth may have reduced tidal flux in and out of the estuary to some degree, disrupting sediment transport processes in the mouth region (Beck & Basson 2007). For example, the changes in tidal flux has resulted in significant sedimentation of the old channel, which has become narrower and shallower, and could be the reason for the need for ongoing dredging that has to be carried out in the area (Beck & Basson 2007).

Several bridges have also been constructed across the estuary (Figure 4.1). The Carinus Bridge (R27) across the estuary at Velddrif was built in the 1940s, giving the people of Velddrif permanent access to the Cape. This bridge appears to have caused local scouring of the bank at the left river bend travelling downstream — the river channel upstream of the Carinus Bridge at Velddrif was around 150 m wide in 1998, compared to 100 m in 1990 (Beck & Basson 2007). By the end of the 1990s flood embankments had been built on the floodplain upstream of the Carinus Bridge and further upstream at the Saldanha-Sishen Railway Bridge where salt mining was being carried out (the latter is part of the railway line that connects iron ore mines near Sishen in the Northern Cape with the port at Saldanha Bay in the Western Cape. The line and bridge were opened in 1975 (Beck & Basson 2007). The Kersefontein Bridge (45 km upstream) is the third bridge spanning the estuary. There is management concern that the estuary channel is migrating southwards in the vicinity of the Carinus bridge, and that it may be washed away in the future. While this may be associated with stabilization of the mouth, it has been suggested that boat wakes as well as natural processes (currents, waves and winds) may also be a contributing factor (DEA&DP 2019a).

These modifications may have influenced the extent of flooding in the floodplain area. to the Under present day conditions, base flows occurring between June and July (i.e. "normal" winter flows excluding flood events) inundate about 34.7% of the floodplain on average (Basson 2014). Under Reference conditions there was an approximate 2% greater inundation, mostly in the upper reaches of the estuary (Basson 2014). Flood modelling studies suggest that while the canalised mouth does not restrict tidal flows in and out of the mouth, this may not be applicable under flood conditions, Water levels at Velddrif during the 1954 flood are expected to have been lower than under current development levels owing to the fact that natural river mouth is wider than the existing canalised mouth -the 1:100-year flood level at Velddrif is predicted to have been 0.3 m lower than at present (Basson 2014).

In addition, both the main channel and floodplain have become narrower in places over time. For example, since construction of the permanent mouth, the river width upstream of Steenboksfontein (Figure 4.1) had decreased from 75 m to 50 m and the floodplain width at Lang Riet Vlei farm (about 21 km from the mouth) had decreased from 5.5 km to 4.8 km by the 1980s (Beck & Basson 2007). Such changes may be the result of reduced inflow from the catchment or developments in the estuary but may also be due to natural dynamics of the estuary (Beck & Basson 2007).

Despite the seemingly large nature of these anthropogenic changes, impacts on the hydrodynamic functioning and sediment transport in the Berg River Estuary as a whole have remained relatively modest, mostly due to its large size (Beck & Basson 2007). In fact, as minimal development has taken place on the floodplains in the upper reaches of the estuary, and this portion of the estuary has retained much the same size and course as it was in 1938 (Beck & Basson 2007). Effects of the anthropogenic developments described above, and subsequent changes to hydrodynamic functioning are, however, felt in lower reaches of the estuary, particularly at the mouth.

4.3 Sediment dynamics

Typically, estuaries contain a mixture of river and marine sediments, the balance of which is determined by the size of the tidal prism (amount of water moving in and out of the estuary during a tidal cycle), and riverine base flows and floods. The size of particles that can be transported from the catchment increases with increased velocity, and larger particles are deposited before small particles as flow is reduced. Where strong currents prevail, the substratum will be coarse (sand or gravel) and where current speeds are low, the substratum is typically comprised of fine sediment (mud and silt). The sediment in both the lower and upper reaches of the Berg River Estuary are characterised by a higher percentage of larger sediment particles (i.e. a mostly sandy environment) than the middle reaches (Figure 4.8). This is due to strong current velocities of the tides in the lower reaches, and fluvial inflow at the head in the upper reaches.

Base flows carry relatively little sediment, mostly fine silts, which is deposited when freshwater flows are slowed by the pushing effect of incoming sea water. Organic matter also flocculates out of the freshwater when it mixes with saltwater. These processes lead to accumulation of fine sediments in the middle estuary, so that the channel and inter-tidal areas become muddier and shallower with time (Figure 4.8). Accordingly, the finer sediment characteristic of the middle reaches of the estuary also typically contain higher levels of organic matter compared with the coarser sediments prevalent in the upper and lower reaches (Figure 4.9).

Floods carry a lot of silt from the catchment, and this is deposited wherever floodwaters slow down significantly, such as on the floodplain. Floods also scour away the sediments that have built up in the channel and in the lower inter-tidal areas. Very large floods may scour the floodplain as well. The area of scouring versus deposition is likely to depend on the size of a particular flood: under low flow conditions, erosion and deposition is also confined to the main channel in the upper reaches of the estuary and the sediment transport is very small, with a maximum scour of about 0.2 m (Beck & Basson 2007). Under flood conditions, model simulations show that erosion and deposition patterns are much more extensive. For example, in the main channel, scouring and deposition of up to 0.3 m occurs (Beck & Basson 2007).

The relationship between flow and sediment load also means that there is a positive relationship between flow and turbidity of inflowing water. Since seawater is comparatively clear, there is a negative relationship between estuarine salinity (i.e. distance to the mouth) and turbidity. Thus, seasonal reductions in freshwater flows lead to decreased turbidity and increased light penetration in the system. Drought conditions are likely to result in fewer flood events, and therefore less scouring of channel and floodplains.

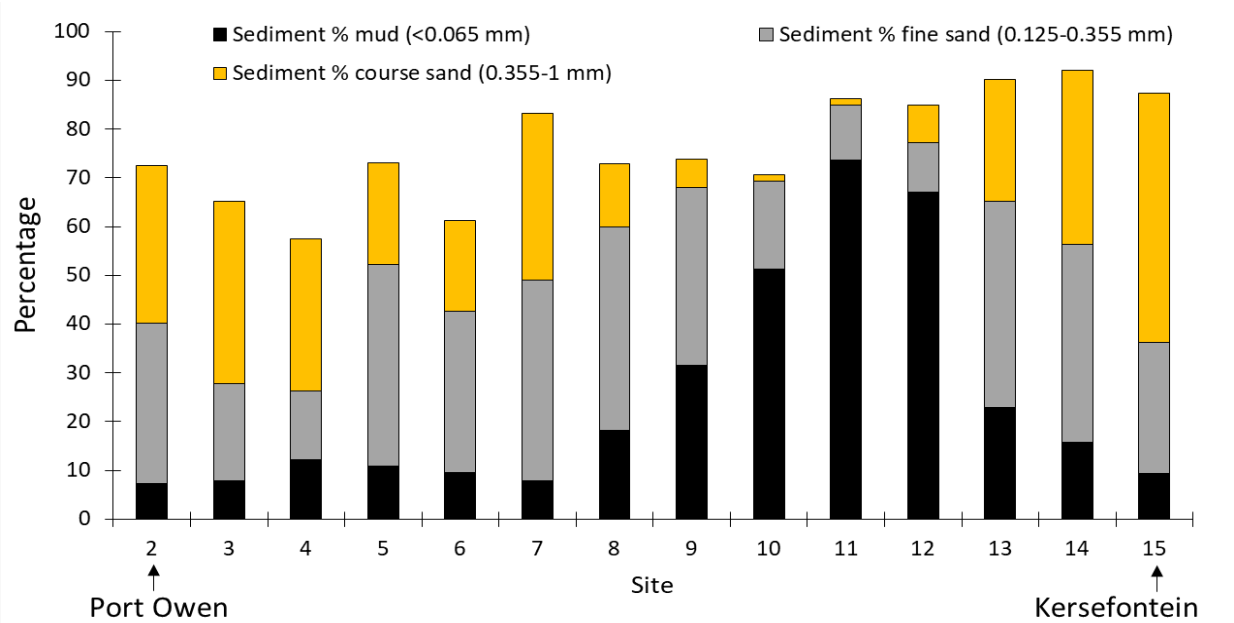

Figure 4.8. Particle size percentage composition (i.e. sediment type) up the Berg River Estuary from Port Owen (3 km from the mouth, site 1) to Kersefontein (47 km, site 15) (based on data from Wooldridge 2007).

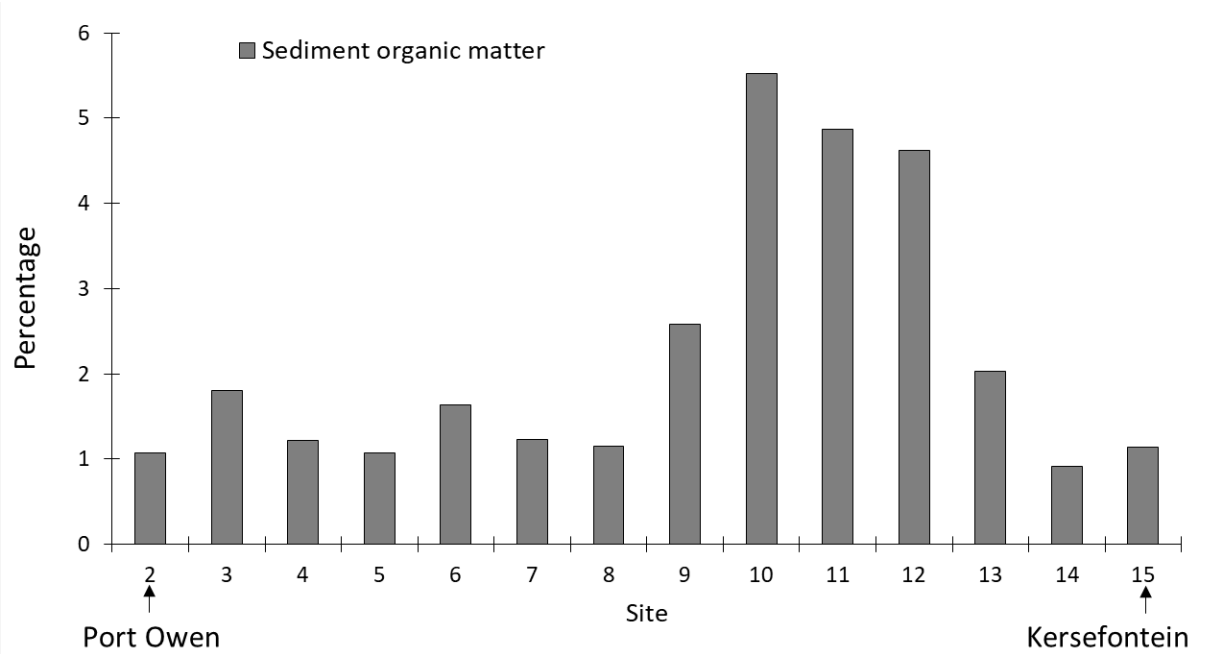

Figure 4.9. Total sediment organic matter up the Berg River Estuary from Port Owen (3 km, site 1) to Kersefontein (47 km from the mouth, site 15) (based on data from Wooldridge 2007).

The construction of the new permanent mouth may have reduced tidal flux in and out of the estuary to some degree, disrupting sediment transport processes, and therefore, influencing erosion in the mouth region. This could be the reason for the ongoing dredging that has had to be carried out in the lower estuary (Beck & Basson 2007). Dredging is also periodically undertaken at the Port Owen Marina (Box 4.1 below).

Box 4.1. History and environmental authorisation for dredging in the Port Owen Marina.

The Port Owen Marina was designed and developed by the Owen Wiggens Trust during the second half of 1988. The waterways and infrastructure were sold to the Velddrif Municipality in July 2000. Responsibility for the management and control of the marina was later transferred to the Port Owen Marina Association (POMA) in 2001. The marina was designed with two entrances to the Berg River Estuary. While this ostensibly allows through flow of estuarine water, it also causes rapid sedimentation of the waterways and a resulting decline in depth. As such, the entrance canals and yacht/boat mooring area of the Port Owen Marina require regular maintenance dredging. The Velddrif Municipality obtained authorisation to dredge the marina in 2000 under the Environmental Conservation Act (ECA) (No. 11927 of 1989). The municipality dredged 71 000 m^3, and this material was discharged directly into the main estuary channel.

POMA undertook an Environmental Impact Assessment (EIA) to undertake additional maintenance dredging in 2009, and Environmental Authorization (EA) was granted by the Department of Environmental Affairs and Development Planning (DEA&DP) on 28 May 2009 (E12/2/3/1-F1/7-0419/08) in accordance with National Environmental Management Act (No. 56 of 2002 (G-24251) (NEMA) (as amended). The conditions of the EA required that coarse dredge material be deposited in settling ponds, while fine sediment may be discharged into the estuary during ebb tides, when the flow rate in the river channel was >200 $m^3.s^{-1}$. The settlement pond sediment was disposed of at the Velddrif waste dump site until its closure in 2011, after which approval was granted for disposal at the Vredenburg landfill in April 2018.

Due to the work conducted on heavy metal contamination of sediments in the Berg River Estuary in 2010, concerns about disposal of dredge spoils led to a November 2015 DEA&DP Pre-Compliance Notice about condition G9 (spoils disposal). In 2017, inspection, sampling and analysis of possible contaminants and meetings with DEA&DP were conducted, and the notice was cleared in 2018.

The 2009 EA further required that the majority of the dredging take place during the winter rainfall period between May and July each year. However, operational inefficiencies, changes in the particle size of spoils (fines now > 97% of total), and increased volumes necessitated year-round dredging. An in-situ assessment of the current velocities in the estuary at Port Owen identified a window of safety either side of the peak of the high tide for different tidal states (spring vs. neap) when it would be acceptable (safe) to discharge the effluent from the settling ponds i.e. when current speed is likely to exceed the 200 $m^3.s^{-1}$ threshold and when there is a reasonable probability that this will be washed out of the estuary mouth and not be re-entrained on the next incoming tide) (Hutchings & Clark 2016). This resulted in an application of amendment to the EA, which was granted by DEA&DP in 2018. Dredging has been undertaken three times since 2015: 5700 m^3 in 2015, 2800 m^3 in 2016 and 15 000 m^3 in 2017.

With the promulgation of the National Environmental Management: Integrated Coastal Management Act (No. 24 of 2008) (ICMA) (as amended), POMA did not have the necessary permit to discharge land-derived effluent (i.e. dredge matter in the settlement ponds) into the estuary. The Department of Environmental Affairs (DEA) would not allow an application for the disposal of the fine sediment dredge material directly into the estuary. As such, POMA and DEA are currently in dispute as to the way forward, and dredging has essentially ceased. This has led to the gradual sedimentation of the Port Owen Marina, which has essentially become impassable for vessels, with associated economic costs.

Erosion of channel banks in the Berg River Estuary is potentially threatening valuable habitat. Potential causes of bank erosion were investigated as part of this study and proposals on options for rehabilitation of the banks have been formulated. Birdlife Africa previously identified five areas where bank erosion threatens important habitat and King (2017) wrote a

report on potential options for protecting these banks from further damage. These are sites at Admiral Island, Carinus Bridge, Cerebos Saltworks, Kuifkopvisvange and Kliphoek.

A review of existing bathymetric data for the estuary was undertaken and cross-sections previously surveyed were found close to all the sites identified by Birdlife Africa. The depth of the channel varied from as much as 9 m near Carnius Bridge to 3.5 m near Admiral Island. Historical aerial photographs of the estuary were also obtained and studied. Except for changes to the Admiral Island salt marsh in the 1980s and Carinus Bridge before 2003, very little change was evident since 1938. The estuary was visited by the study team and the various bank erosion areas inspected and mapped. Erosion at Admiral Island (located mainly on the inside of a bend) appears to be related to wave action, possibly caused by boats and wind, while erosion at the other sites (located on the outside of bends) appears to be due to flow and waves with the possibility of the channel moving slowly as would be expected in this environment. Wind wave erosion was observed occurring at Carinus Bridge. The predominant wind direction according to wind data at Langebaanweg is from the southwest.

Flow velocities have previously been measured in the estuary and were found to generally be less than 1 m/s. Beck and Basson (2004) modelled the hydrodynamics of the estuary and found that velocities for the most part were predicted to be less than 2 m/s during an extreme flood simulation. Velocities up to about 0.6 m/s can be expected not to cause scour. They also modelled the sediment dynamics and found that during extreme flood events the channel can be expected to widen with sediment scoured from the banks and deposited in the channel.

The impact of dredging on bank erosion is thought to be less significant due to the areas that have been dredged in the past and differences in bank erosion in areas close to the harbour where certain banks appear to be stable and others are not. The impact of sea level rise is a potential threat to banks in the future as waves attack areas higher up the banks.

King (2017) proposed three designs be piloted for their effectiveness: regrading of the bank slope and planting with suitable indigenous vegetation, regrading of the bank slope and construction of a toe berm to prevent wash away of material into the channel and planting with suitable indigenous vegetation, and regrading of the bank slope and laying down of geo cells in which soil can be placed and suitable vegetation planted. An existing area of riprap revetment at Carinus Bridge has been in operation for nearly 20 years and is performing well. The riprap has vegetated naturally as well. It appears that the banks on each end of the riprap section may have retreated further in the past 20 years. Quantities were determined for each site and for each option and the costs determined for each. It is recommended that the options recommended by King (2017) are piloted at Admiral Island in small quantities so that no EIA is required and their performance assessed. If they are found to be performing well then, they can be rolled out to larger areas once an EIA has been undertaken. If they are not found to be performing well then it is recommended that riprap be investigated as an option as well-designed riprap can provide excellent services for many years virtually maintenance free.

4.4 Present status and potential changes under different scenarios

4.4.1 Physical habitats and processes (bathymetry and sediments)

The physical habitats of the Berg River Estuary are affected by both the sediment dynamics and the anthropogenic modifications described above. In the EHI, physical habitats are described in terms of sandy/muddiness and organic content of sediments, as well as the major factors influencing the, subtidal, intertidal, supratidal and floodplain areas (DWA 2012).

4.4.1.1 Reference processes

Intertidal sand and mudflats under Reference condition would have occupied approximately 133 ha in the estuary, most of which would have been found in the lower (56% or 74 ha) estuary with the remainder split between the middle (29% or 39 ha) and upper estuary (15% or 20 ha) (DWA 2010). Periodic floods would have restricted build-up of riverine sediment in the estuary and restricted the ingress of marine sediment into the system, or at least would have flushed these out on a regular basis (DWA 2010).

The Berg River Estuary is a river-dominated system and sediment dispersal occurs seaward of the river mouth (DWAF 2007). Because of the very low wave energy and surf zone currents off the mouth, these sediments tend to collect there. The low surf-zone sediment transport potential also means that sedimentation processes that result in the closure of the mouth were minor compared to other South African estuaries i.e. the mouth would have closed only rarely under Refence conditions (DWA 2010).

4.4.1.2 Current processes

The very low gradient of the estuary and its great length means that fluvial sediments entering the head of the estuary have a long residence time in the system before being exported to the sea (DWA 2010). In addition, not all of the fluvial sediment necessarily reaches the sea; with some deposited on the wide floodplains of the system (DWA 2010). Even large river floods cannot easily move sediments right through the estuary and estuarine sediments tend to be moved along in pulses with a range of flood magnitudes (the relative amount of sediment and downstream transportation distance being related to the size and duration of the flood).

The long, low-gradient nature of the Berg River Estuary with its extensive floodplains, also "regulates" the effect of river floods and in general progressively "buffering" the sediments and morphology in a downstream direction, with this effect significant in the middle and especially the lower estuary (DWA 2010). This flood "regulation/buffering" effect together with the large tidal prism of the estuary results in the lower estuary sediment regime and morphology being largely affected by the sea tides, particularly since the mouth was stabilised in a wide-open state (DWA 2010). This results in a sediment distribution of course grains (i.e. sand) at the head and mouth, and fine mud, high in organic content, in the middle reaches.

DWA (2010) attributes much of the changes in physical habitat score away from the Reference condition to development around the estuary (i.e. the Port Owen Marina, salt marsh development and expansion etc.), rather than due to changes in flows.

It must be noted however that DWA (2010) reported that confidence in the quantification of sediment dynamics and morphology and changes therein, is low as there are virtually no sediment or morphology data for the Reference condition, and a paucity of such data for the present. As such, DWA (2010) only scored changes in the resemblance of intertidal sediment structure and distribution to Reference condition. In this study we followed a similar approach for both the present day, and future scenarios (Table 4.2).

4.4.1.3 Future processes

Reduced number and size of floods under the future scenarios will translate into reduced sediment transport and scouring capacity within the estuary. This will result in a less dynamic sediment bottom and greater potential for consolidation of sediments. The reduced sediment transport potential and scouring could also allow more marine sediment intrusion through the mouth, possibly longer residence of such sediments within the estuary before eventual flushing out due to occasional large floods, and further movement of such sediments up the estuary. Again, in alignment with DWA (2010), this study only scored future changes in the resemblance of intertidal sediment structure and distribution to Reference condition.

Modelled "low" winter base flows (<12 m^3/s) result in a varying reduction in floodplain area in the upper (3% reduction), middle (3% reduction) and lower reaches (2% reduction). Under low base flows, there is a small reduction (2-3%) in overall modelled flood extent. However, these models were for low flows, but not the expected further 34% reduction in base flows predicted under Scenarios C0 and C1 (Table 4.1). The changes in score therefore take this further 50% reduction in freshwater flow into account (i.e. a 2-3% reduction under current low winter flows will become a reduction of 4-6% under Scenario 4 and 5).

The overall score (Table 4.1) indicates that the physical habitat and processes health of the Berg River Estuary is the same as the 2010 assessment, and that future scenarios will not have a significant impact on this particular health indicator (down to 59% of Reference in Scenarios C0 and C1, from 66% of Reference in 2020 current state). This is because the primary driver of changes in physical habitat and processes have been anthropogenic in nature (i.e. building of dams, bridges etc.).

Table 4.1. Change in physical habitat and processes score.

	Ref	2010	P0 2020	P1	F0	F1	C0	C1
				PD+EWR	FD	FD+EWR	FD+CC	FD+CC+ EWR
a. % Similarity in intertidal area exposed	100.0	61.0	61.0	61.0	61.0	61.0	60.0	60.0
b. % Similarity in sand fraction relative to total sand and mud	100.0	75.0	75.0	75.0	75.0	75.0	55.0	55.0
c. Resemblance of subtidal estuary to Reference condition: depth, bed or channel morphology	100.0	74.0	64.0	64.0	64.0	64.0	60.0	60.0
Health score = weighted mean of a and b and c	100.0	71.0	66.0	66.0	66.0	66.0	59.0	59.0

Table 4.2. Changes in physical habitat (sediment structure and distribution) under P0, and Scenarios P1, F0 and F1 (from DWA 2010).

Variable	Score	Motivation	Confidence
1. Resemblance in intertidal sediment structure and distribution			
1.1 Intertidal area exposed	61	Lower reaches : 25% (mouth structures, moved and fixed, deeper channel, greater tidal flux thus more marine sediment intrusion; channel dredging; wharfs, jetties, embankments; marina and dredging + dumping; salt works; fewer and smaller floods – thus less sediment flushing and more marine sediment intrusion). Anthropogenic impacts virtually unchanged from present: 75% of 25%. Flow related: (25% of 25%) + 10% increase relative to present (~10% bigger and more floods; less sediment trapped in dams) Middle reaches: 50% (Carinus and Railway bridges and embankments; siltation of channel; bank erosion; livestock trampling of inter and supra tidal sediments; a few wharfs, jetties, embankments; less and smaller floods – thus less sediment flushing, less dynamic bottom (also greater potential for consolidation) and longer time for fluvial sediments to pass through middle reaches). Upper reaches: 75% (one bridge, channel modified – shortcut; livestock trampling of inter and supra tidal sediments; less and smaller floods – thus less sediment flushing, less dynamic bottom (also greater potential for consolidation) and longer time for fluvial sediments to pass through upper reaches; less fluvial coarse sediment deposition due to dam trapping). The 3 zones represent 15%, 29% and 56% of the total subtidal area of the total estuary. Weighted (based on subtidal areas) mathematical average for all 3 zones = 61%	Low (virtually no sediment or morphology data for Reference condition; paucity of such data for present)
1.2 Sand fraction relative to total sand and mud	75	Lower reaches: More marine sediment intrusion through permanently open mouth and greater tidal change (10%) Upper reaches: dams trapping some coarse fluvial sediments (5%) Whole estuary: reduced sediment transport and scouring capacity through reduced floods, thus more marine sediment intrusion; also, less dynamic sediment bottom and greater potential for consolidation (10%). Cumulative impact = 25%, thus score = 75%	Low (virtually no sediment or morphology data for Reference condition; paucity of such data for present)
2. Morphology and bathymetry			
2.1 Resemblance of subtidal estuary to Reference condition: depth, bed or	64	Most of impacts listed in intertidal area exposed above are considered to have effects through intertidal into subtidal area. Thus, practically same total score: Lower reaches: 25% (mouth structures, moved and fixed, deeper channel, greater tidal Δ thus more marine sediment intrusion; channel dredging; wharfs, jetties, embankments; marina and dredging + dumping; salt works; less and smaller floods – thus less sediment flushing and more marine sediment intrusion.) Middle reaches: 50% (Carinus and Railway bridges and embankments; siltation of channel?; bank erosion; a	Low (virtually no sediment or morphology data for Reference condition; paucity of such data for present)

channel morphology		few wharfs, jetties, embankments; less and smaller floods – thus less sediment flushing, less dynamic bottom (also greater potential for consolidation) and longer time for fluvial sediments to pass through middle reaches.) Upper reaches: 75% (one bridge, channel modified – shortcut; less and smaller floods – thus less sediment flushing, less dynamic bottom (also greater potential for consolidation) and longer time for fluvial sediments to pass through upper reaches; less fluvial coarse sediment deposition due to dam trapping.) The 3 zones represent 15%, 29% and 56% of the total subtidal area of the total estuary. Weighted (based on subtidal areas) mathematical average for all 3 zones = 64%	
Physical habitat score	66		

3. Anthropogenic influence			
3.1 Percentage of overall change in intertidal and supratidal habitat caused by anthropogenic activity as opposed to modifications to water flow into estuary	25	Morphology/habitat: the same anthropogenic influences as before, but the relative importance decreases by 5% due to the flow related impacts increasing by 5%. Lower reaches: anthropogenic influence: 71%; middle reaches: 48%; Upper reaches: 24%. Weighted mathematical average for all 3 zones = 37% Sediment composition: More marine sediment intrusion through permanently open mouth and greater tidal flux: 10% of total 26% impact Thus, total cumulative anthropogenic influence = 37% of 41% impact + 10%/26% of 26% impact = 16% + 10% = 25%	
3.2 Percentage of overall change in subtidal habitat caused by anthropogenic modifications (e.g. bridges, weirs, bulkheads, training walls, jetties, marinas) rather than modifications to water flow into estuary	26	Most of impacts listed in 3.1 are considered to have effects through intertidal into subtidal area. Thus, practically same total score. Morphology/habitat: the same anthropogenic influences as before, but the relative importance decreases by 5% due to the flow related impacts increasing by 5% Sediment composition: More marine sediment intrusion through permanently open mouth and greater tidal Δ: 10% of total 26% impact. Thus, total cumulative anthropogenic influence = 38% of 42% impact + 10%/26% of 26% impact = 16% + 10% = 26%	

4.4.2 Hydrodynamics

Hydrodynamics, together with sediment processes and water quality (biochemical) processes (i.e. the abiotic components) in estuaries are, in most instances, the components of an estuary where modification in flow and other human factors manifests influence first. For example, reduced river inflow changes water circulation and salinity distribution patterns in estuaries which in turn affect the biota, while wastewater discharges affect the chemistry of an estuary which in turn affects the biota. Exceptions which may have direct effects on biota, include influences such as living resource exploitation and human disturbance of biota. Knowledge and understanding of these abiotic components, therefore, are crucial in any baseline and health assessment study in estuaries (DWA 2012).

4.4.2.1 Mouth condition and abiotic states

The previous 2010 RDM (DWA 2010) described the change in the mouth conditions for the different flow scenarios (the method scores mouth closure conservatively following the guidelines provided in DWAF 2008). Here, the change in the mouth condition relative to the Reference state was scored as 90% — the system would have been significantly more constricted during the summer months before mouth stabilisation, and there is also anecdotal information that indicates that the Berg might have closed for short periods during drought conditions under the Reference condition (Table 4.6).

The current (2020) state of the mouth conditions is considered the same as those described in DWA (2010), and the rating for the mouth condition relative to Reference has thus been kept the same (90%) for present day and all Scenarios (Table 4.6). While historically the river dominated estuary maintained an open mouth most of the time, occasional closure would occur during the dry summer, this closure can no longer occur due to the canalised and permanently open mouth. Abiotic states were classified based on the abiotic states that can occur in the estuary defined by DWA (2012) (Table 4.3) (Table 4.6).

Table 4.3. Summary of the abiotic states that can occur in the estuary (from DWA 2012).

State	Name	Brief Description	Flow Range ($m^3.s^{-1}$)
1	Severe marine-dominated	Saline intrusion extends further than 45 km upstream of mouth	< 0.5
2	Marine-dominated	Saline intrusion extends up to 45 km from mouth	0.5-1.0
3	Small to medium freshwater inflow	Marine influence evident up to 33 km from mouth	1.0-5.0
4	Medium to high freshwater inflow	Marine influence only evident up to 12 km from mouth	5.0-25.0
5	Freshwater-dominated	Estuary is fresh throughout	>25.0

4.4.2.2 Water column stratification

The Berg River Estuary has historically been a non-stratified system, due to the long length and shallow gradient. However, there is some stratification does occur in lower reaches during higher flows. Under both Reference and Current conditions, the system has been dominated by river flow in winter, and marine flow in summer, but shows little classical estuarine circulation and is therefore still a non-stratified system under both present day (Table 4.6). This stratification in the lower reaches during higher flows will be impacted by future changes in those flows i.e. under future Scenarios, this effect will be lessened, particularly under climate change (Table 4.6).

4.4.2.3 Water retention time

Under Reference conditions, the Berg River Estuary would have been a strongly freshwater dominated system in winter, and marine dominated in summer. The system would have therefore swung between low and high retention times, respectively, but overall retention times would have remained low, being a freshwater dominated system. Under present day conditions and Scenarios P1, F0 and F1, while the artificial mouth ensures constant connection to the sea, lower freshwater input increases residence time during summer (47-50% of natural flow reaching the estuary). Even during winter, less freshwater from upstream is reaching the system, increasing residence times relative to Reference.

Hydrodynamic models of the Berg River Estuary suggest that a freshwater inflow exceeding 140 m^3/s is sufficient to fully flush the estuary of saline water (Schumann 2009). The minimum freshwater water requirement to maintain the system is 0.6 m^3/s. The latter is therefore more important in determining residence time — here, we define a "low" residence time as occurring whenever flow rates exceeded 0.6 m^3/s. The proportion of the months that average flows exceeded this 0.6 m^3/s relative to baseline is shown in Table 4.4 (final score shown in Table 4.6). However, retention time is also demined by flood frequency. The proportion of the months that average flows exceeded 300 m^3/s (a "medium" flood) relative to baseline is shown in Table 4.4 (final score shown in Table 4.6).

Table 4.4. Retention time scoring, based on the % months where average flow > 0.6 m^3/s and >300 m^3/s

	Ref	PD 2020	P1	F0	F1	C0	C1
			PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
% months flow > 0.6 m^3/s	100.0	75.0	96.5	72.4	93.8	64.6	90.3
% months flows > 300 m^3/s	100.0	42.6	42.6	42.6	42.6	21.3	21.3

4.4.2.4 Water levels and flooding

Under Reference conditions, the Berg River Estuary would have been a strongly freshwater dominated system in winter, and marine dominated in summer. Water levels would have therefore shown some variability between seasons. Floods and water levels are connected – the extent of the flood is determined predominately by the starting water level.

Therefore, water levels were scored based on the percentages of "wet" months (May-July) where average flows were below 50 Mm^3 (Table 4.5) (final score shown in Table 4.6). Freshwater inflow exclusively dictates water level in the upper estuary.

Tides influence water level in the lower estuary. The typical water level variation due to tidal forcing is 1.8 m. However, the impacts of sea level rise on this tidal water level need to be included. As such, sea level rise was assessed by examining the change in intertidal habitat available. Intertidal area colonised by the indicator group used here, salt marshes, occupies a tidal range of 0.5 m. Projected sea level rise is 10 cm, leading to a potential change impact of 20% (Table 4.5) (final score shown in Table 4.6).

Table 4.5. Water level scoring, based on the % wet months (May-July) where average flow > 560 Mm^3 and change in intertidal habitat due to increasing mean sea level.

	PD 2020	P1	F0	F1	C0	C1
		PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
% months (May-July) flow < 50 Mm^3	45.5	45.5	41.2	40.7	26.2	26.2
% change in intertidal habitat due to changes in water level	100.0	100.0	100.0	100.0	20.0	20.0

Overall, the hydrodynamics score of the Berg River Estuary is lower under present day (2020) than that assessed in 2010 (DWA 2010). This is in part linked to the fact that only one of the four hydrodynamic parameters were scored in 2010 but also due to real changes in the intervening period (Table 4.6). Scenarios that include climate change (C0 and C1) show a further reduction in hydrodynamic health, especially due to declines in retention time and water level health scores (Table 4.6).

Table 4.6. Hydrodynamic scoring of the system under historical (2010), present day (2020) and for all considered scenarios.

	Wt	2010	P0 2020	P1	F0	F1	C0	C1
				PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
Mouth	50	90.0	90.0	90.0	90.0	90.0	90.0	90.0
Abiotic state		—	69.3	68.4	66.3	73.4	58.1	64.2
Stratification	10	—	90	90.0	90.0	90.0	80.0	80.0
Retention time	20	—	42.6	42.6	42.6	42.6	21.3	21.3
Water level	20	—	45.5	45.5	41.2	40.7	20.0	20.0
Health score		90.0	60.8	60.8	58.9	61.3	46.4	46.8

5 WATER QUALITY

5.1 Introduction

The term "water quality" is often associated the trophic (nutrient) status of an estuary and the presence of heavy metals and pathogens (such as faecal coliforms), but also includes the physical properties such as temperature, salinity and dissolved oxygen. These different aspects of water quality have different drivers. The physical properties such as temperature and salinity are closely linked to hydrodynamics; dissolved and suspended constituents are introduced into the system from the river catchment area, the sea, atmospheric inputs and internal recycling (Stanley & Hobbie 1977, Fisher *et al.* 1982); while pathogens are introduced from urban areas and particularly poorly serviced, growing informal settlements (Gorgens and de Clercq 2005, Taljaard *et al.* 2010, Cullis *et al.* 2019). All of these are influenced by flows, and hence also sensitive to drought.

While conditions in St Helena Bay have changed little over the last 70 years (Hutchings *et al.* 2012), the changes to the estuary mouth and the associated sedimentation and dredging, have affected marine influences on the water quality characteristics of the estuary. Conditions in the Berg River Catchment have changed dramatically over the last 70+ years (Taljaard *et al.* 2010, Cullis *et al.* 2019), with major impacts on estuary water quality. Pollutants have increased due to agricultural return flows and expanding human settlement in the catchment area and along the margins of the estuary, and the effects are exacerbated by a reduction of diluting freshwater inflows (Taljaard *et al.* 2010, Cullis *et al.* 2019).

The relative importance of these various influences on the water quality characteristics of the Berg River Estuary are explored in detail in this chapter, along with an assessment of how these characteristics have changed over time including under "normal" rainfall and drought conditions, and how they are likely to change in future under the range of different scenarios. In addition, the current water quality within the system is compared with the gazetted resource quality objectives (RQOs) for the Berg-Olifants Water Management Area.

The study was based on data from several sources, including recent monitoring data collected by DEA&DP that have not previously been analysed. Two types of data were available for interpretation: (1) longitudinal data sets which include samples taken on a single occasion, during specific tidal states, at multiple stations within the estuary, and (2) long term time series data - represented by readings taken at discrete sites and distinct time intervals over several years. These data were from multiple sources (see Figure 5.1 for sampling locations):

- CSIR Estuaries of the Cape – Synopses of available information on individual systems. Data on Berg River Estuary collected from surveys conducted in 1975, 1976, 1989, 1990 and 1995. Data are available from ~19 sites spanning the length of the estuary.
- Department of Water & Sanitation (DWS): Berg River Baseline Monitoring Programme. Data on the Berg River Estuary collected over the period 2003-2005 (DWAF 2007). Data are available from ~25 sites spanning the length of the estuary.
- Department of Water & Sanitation (DWS) and West Coast District Municipality (WCDM): continuous monitoring of selected parameters (temperature and salinity) at two sites in the Berg River Estuary over the period 2012-2019.

- Berg River Improvement Plan (BRIP) undertaken by Department of Environmental Affairs & Development Planning (DEA&DP), within a transversal Western Cape Government project. Water quality data available for a range of sites in the Berg River (12 sites) and Estuary (10 sites) collected over the period 2013-2019.
- Department of Water & Sanitation (DWS): Resource Quality Information Services (RQIS) – Unpublished data collected as part of national monitoring programmes between 1965 and 2019. Thse include sites above the estuary and also in the estuary itself (Drieheuwels, Misverstand Dam Bridge, Jantjiesfontein, Kliphoek and Carinus Bridge).

5.2 Physical properties – salinity, temperature, oxygen

5.2.1 Salinity and temperature

Salinity and temperature are influenced by both the sea conditions and freshwater inputs. Salinity values for inshore water along the West Coast and St Helena Bay typically vary between 34.6 and 34.9 (Shannon 1966). Therefore, the seawater entering the estuary has values for water temperature (12.3-14.2 °C) and salinity (34.6-34.9) that are typical for the upwelling driven West Coast of South Africa (Laird & Clark 2015, Wright *et al.* 2019). The interplay between the fresh and saline water inflows determine the extent of saline intrusion up the estuary, and the consequences thereof. The lower reaches of the Berg River Estuary remain cool during summer due to upwelling at sea.

Salinity, temperature and oxygen data collected along the length of the estuary during the Berg River Baseline Monitoring Programme in 2003-2005 (DWAF 2007) are shown in Figure 5.2 to Figure 5.7 for low (summer) and high (winter) freshwater flow periods during the spring high tide, when the ocean's tidal influence is the strongest. It is important to note that 2003 had lower than average winter rainfall and 2004 was only slightly above average (Figure 3.7).

The salinity profiles show that, for the most part, the estuary is "well mixed" in that salinity does not change much with depth. The opposite situation, which is a "stratified" system, occurs when less dense, warm or low salinity fresh water, lies on top of more dense, cooler or more saline seawater[4]. The lack of stratification in the Berg River Estuary is linked to strong mixing from tidal exchange during summer months as well as inputs of freshwater during winter months. Based on the data from 2004, during summer (low flow) the saline water from St Helena Bay penetrates up past the middle reaches of the estuary to distances ranging between roughly 45 km during spring high and 43 km during neap low tide (Figure 5.2). However, saline water has been known to penetrate as high as 65km upstream or more. In contrast, during periods of incoming freshwater flow in winter, the salinity intrusion is greatly reduced, extending only 15 km upstream during spring high tide and as little as 3 km during neap low tide (Figure 5.2). During the intermittent flow periods (spring and autumn) the upper limit of the saline intrusion ranges between 20-25 km.

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4 This abrupt change in salinity or temperature with depth is known as a thermocline (in the case of temperature) or pycnocline (for salinity).

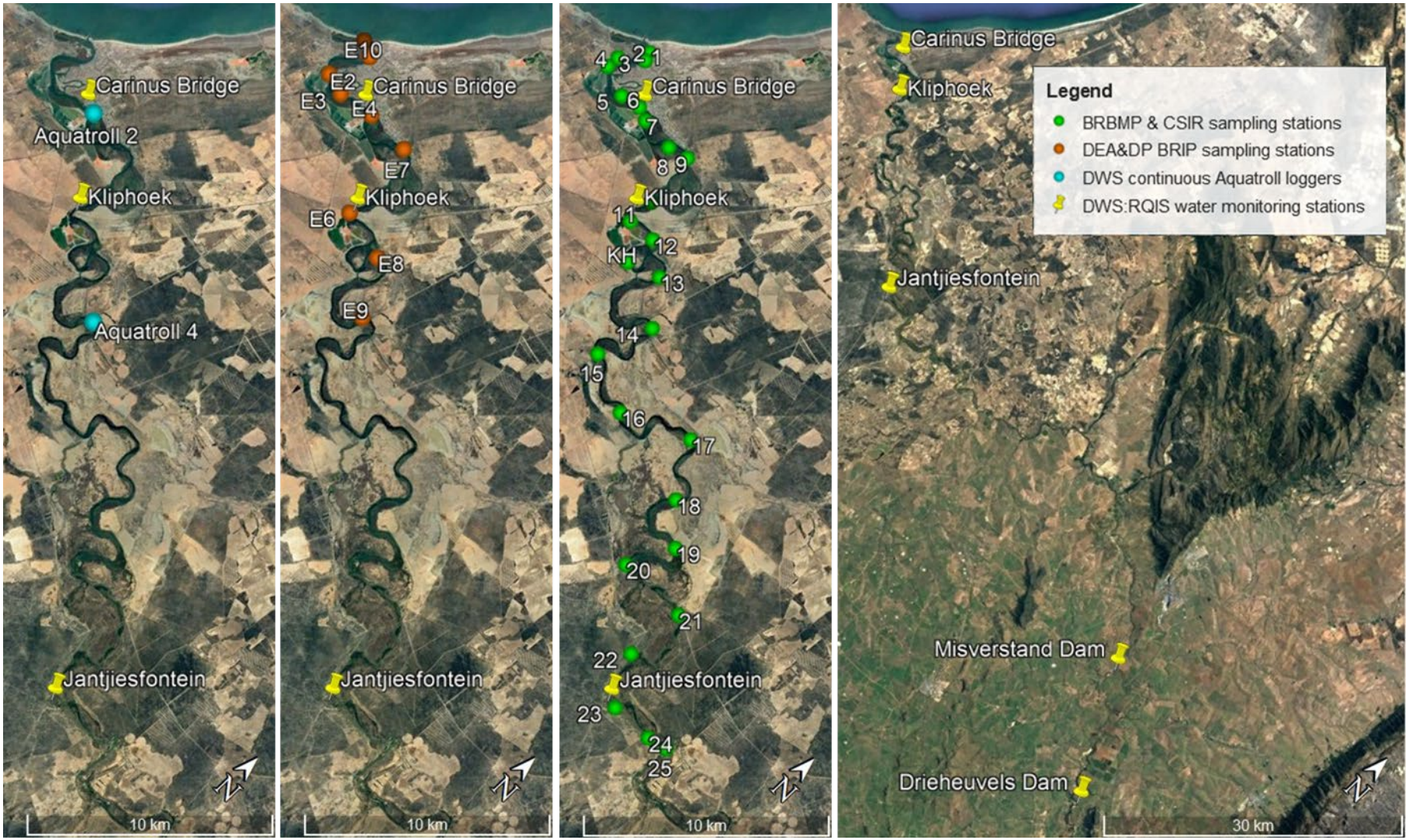

Figure 5.1. **Map showing the locations of the Monitoring stations within the estuary and relative to the Misverstand Dam and Drieheuvels Dam. Stations from the Berg River Baseline Monitoring Project (BRBMP: 2003-2005), Council for Scientific and Industrial Research (CSIR: 1975, 1976, 1989, 1990 and 1995), Berg River Improvement Plan (DEA&DP BRIP:2013-2019), Department of Water & Sanitation and West Coast District Municipality Aquatroll loggers (DWS: 2012-2019) and Department Water & Sanitation: Resource Quality Information Services (DWS:RQIS: 1965-2019).**

Slinger and Taljaard (1974) showed that under normal summer conditions salinity within the Berg River Estuary declines from the mouth to the upper reaches, as seen in Figure 5.2 (middle). However, during the recent drought, more specifically during the first five months of 2018, the lower to middle reaches of the estuary (BRIP stations only reached 23 km upriver) experienced a reverse salinity gradient (Figure 5.3). This means that the salinity **increased** from the mouth to the topmost station.

Temperature varies seasonally (Figure 5.4), with summer water temperatures in the upper and middle reaches of the estuary reaching between 24-27°C during the day. This is as a result of the low level of river inflow and longer water residence time as well as higher ambient air temperature. There is a much greater temperature range in the lower reaches of the estuary because of summer upwelling, when sea water temperatures drop to as low as 8°C. During winter, when freshwater inflow is far greater, water temperatures range from around 14°C at the mouth up to a maximum of around 18°C in the upper reaches.

Continuous monitoring data collected from a single point in the estuary also provide a useful insight into how salinity and temperature in the Berg River Estuary changes in response to short (tidal and diurnal) and longer term (seasonal and interannual) physical forcing. Data collected by the Department of Water and Sanitation (DWS) and the West Coast District Municipality (WCDM) using Aquatroll Data Loggers deployed at two stations in the estuary – Aquatroll 2 located 5.3 km from the mouth, immediately above Carinus Bridge, and Aquatroll 4 some 23.7 km from the mouth in the early middle reaches – demonstrate this very effectively (Figure 5.5). It is possible infer what is happening above and below these two points by examining these two data sets together.

In the lower reaches of the estuary (Aquatroll 2) salinity remained high throughout the summer months (when river flow is lowest), showing very little variability over time (typically remaining in the range of 30-35 for this entire period – close to that of seawater). In winter, salinity at this site was much more variable, ranging between 0 and 35. What this indicates is that seawater forced into the mouth of the estuary on each high tide is flushed out again by freshwater inflow during the winter but not in summer. Importantly though, **during the recent drought** that was experienced between 2017 and 2018, variation in salinity in the lower reaches of the estuary during winter was much reduced, to the extent that **the estuary remained highly saline (>17) for a period of almost 18 months** (see Figure 5.5 second panel). The fact that the estuary at this point (5 km from the mouth) was not properly flushed for such a long period, and also the fact that reduced freshwater inflows also result in reduced flushing of the estuary generally, has important implications for other water quality parameters (dissolved oxygen and nutrients in particular) and biota.

During periods of reduced freshwater inflow, a saline mass of water gets trapped in the estuary that is characteristically warmer than the seawater, but cooler than the river water. At the onset of winter, as freshwater moves into the system, this plug of saline water shifts downstream. If winter flow rates are not high enough to completely flush the plug out of the system, it undergoes some limited renewal near the mouth during flood tidal intrusion, but once winter freshwater flows recede again, this saline plug is pushed back upstream again (Taljaard *et al.* 2010, van Ballegooyen 2010).

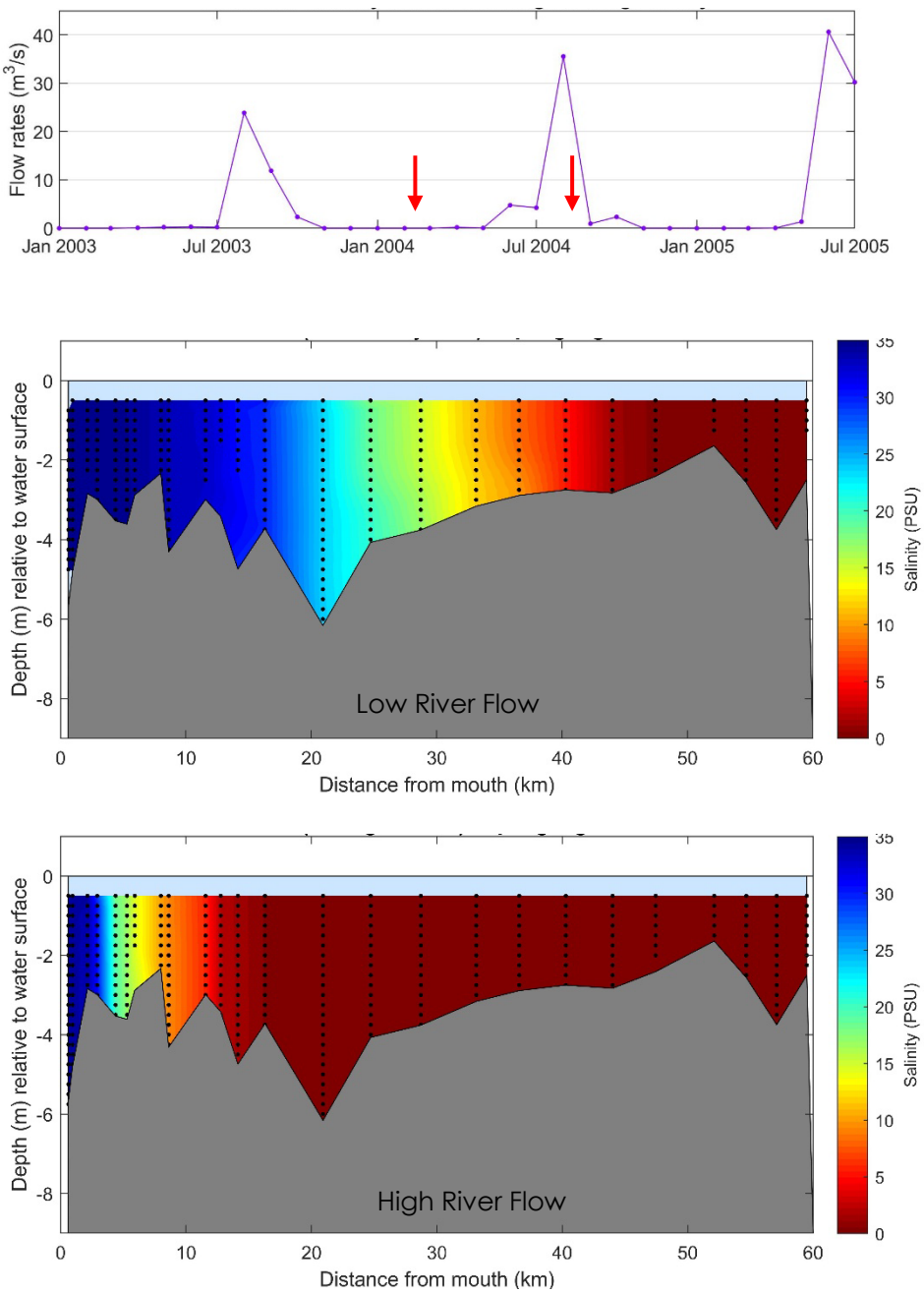

Figure 5.2. Salinity in the Berg River Estuary during spring high tide in summer (low flow, middle) and winter (high flow, bottom) under different freshwater inflows. Variation in freshwater inflows entering at the head of the estuary between 2003 and 2005 with timing of sampling indicated (top), vertical salinity struture during Summer (13 Febuary 2004) – Spring High tide (middle) and Winter (18 August 2004) – Spring high tide (nottom).

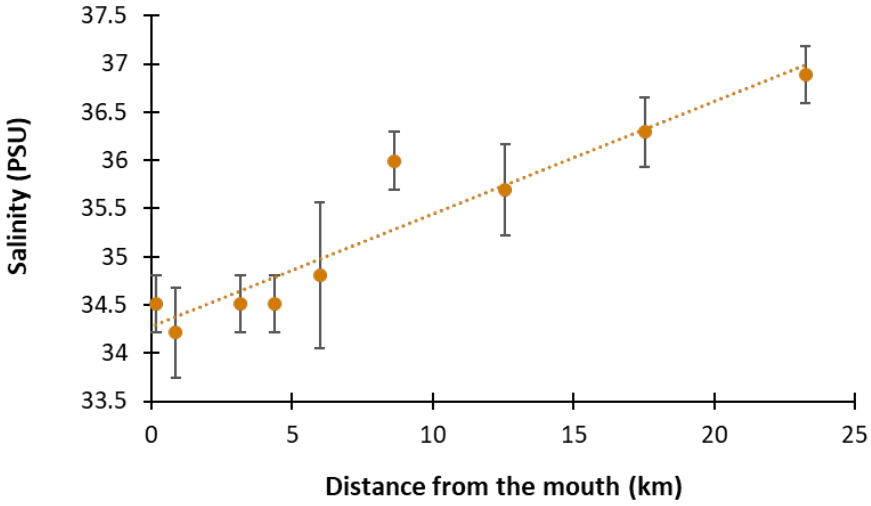

Figure 5.3. **Average salinity (± standard error) in the lower to middle reaches of the Berg River Estuary for the drought period January - May 2018.**

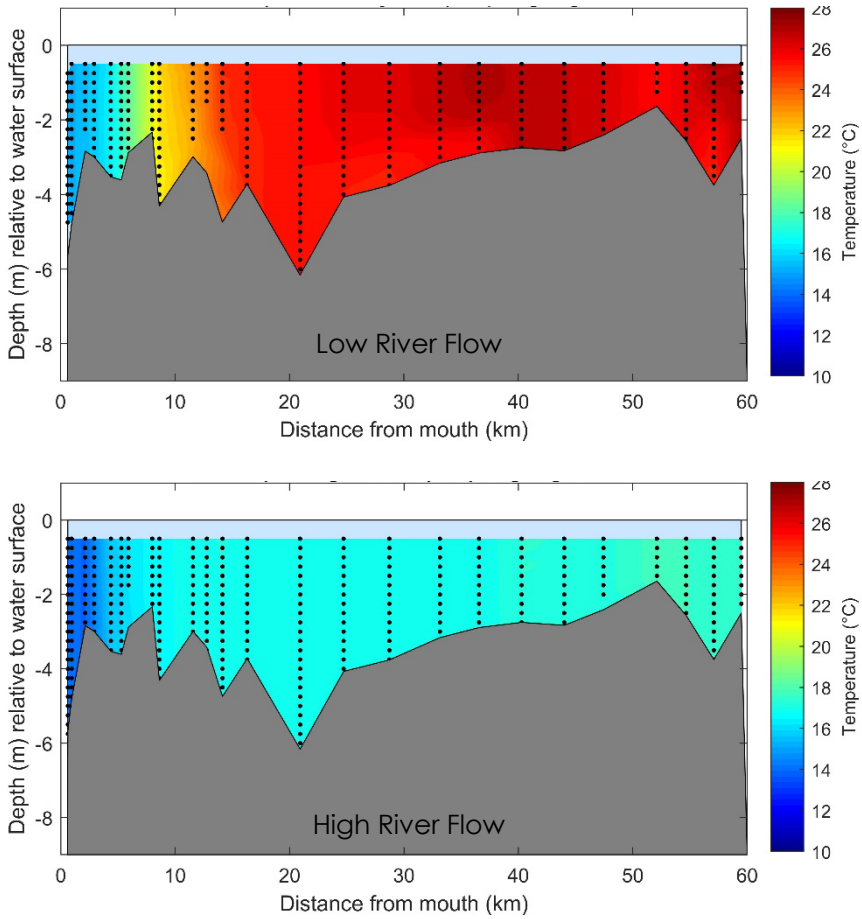

Figure 5.4. **Temperature (°C) within the Berg River Estuary during spring high tide in Summer (Low flow - 13 Febuary 2004) (top) and Winter (High flow - 18 August 2004) (bottom). Times at which these measurements were taken correspond with salinity measurements presented on Figure 5.2.**

Measured variations in salinity in the middle reaches of the estuary (27.3 km from the mouth, Aquatroll 4), were more pronounced than those in the lower reaches. In winter, salinity hardly varied at all (remained below 1 except during the drought), while in summer salinities remained quite variable but did not rise as high as in the lower reaches. During the drought, salinity at this point did not drop below 5 for more than 18 months.

Reduced freshwater inflow during the drought also had a marked impact on maximum salinity levels that were attained in the estuary at both stations. At the upstream site (Aquatroll 4), maximum levels attained each year increased from ~32 in 2016, to ~35 in 2017, and reached ~39 in 2018, which is substantially higher than normal sea water (around 34.5 , Figure 5.5 fourth panel). These increases were a result of evaporation of freshwater from the saline plug of water that was trapped in the estuary over this period. The high salinity readings also aligned with the reverse salinity gradient seen in Figure 5.3, the Aquatroll 4 located 24 km from the mouth (just past the furthest recorded point in Figure 5.3). A once-off reading taken at Kersefontein (roughly 45.5 km from the mouth) on 13 March 2017 recorded a salinity of 34.8, slightly higher than sea water. This suggests that the reverse salinity gradient (salinity increasing with distance upstream) may extend past BRIP station 9. However, the exact extent to which it penetrated upstream is unknown as the next available permanent monitoring station, at which salinity was recorded, is Jantjiesfontein (52 km from the mouth) at which point there was no longer evidence of a reverse salinity gradient.

Long term data records (1970 to 2017) collected by DWS at Jantjiesfontein, located roughly 52 km from the mouth, also highlights changes in the extent of the salinity intrusion (Figure 5.6). Salinity readings at this station are normally <2, however, in summer of 2016/2017 values as high as 4 were recorded. The only other time over the period 1970 to present that salinities this high were recorded at this site was in 1995. These periods of increased salinity may be a point of concern as they exceeded the gazetted RQOs which state that salinity everywhere in the estuary should not exceeded 35 and salinity in the areas above 40 km upstream of the mouth should not exceed 1.

Variations in water temperature at the two Aquatroll stations were not as strongly influenced by freshwater inflow as salinity, and while the overall magnitude of the fluctuations were similar in the lower and middle reaches of the estuary, the short term diurnal variations in temperature were much more pronounced closer to the sea than higher up in the estuary. This is mostly a function of higher tidally induced variations in water temperature in the lower reaches particularly, during summer, caused by the twice daily influx of cool seawater.

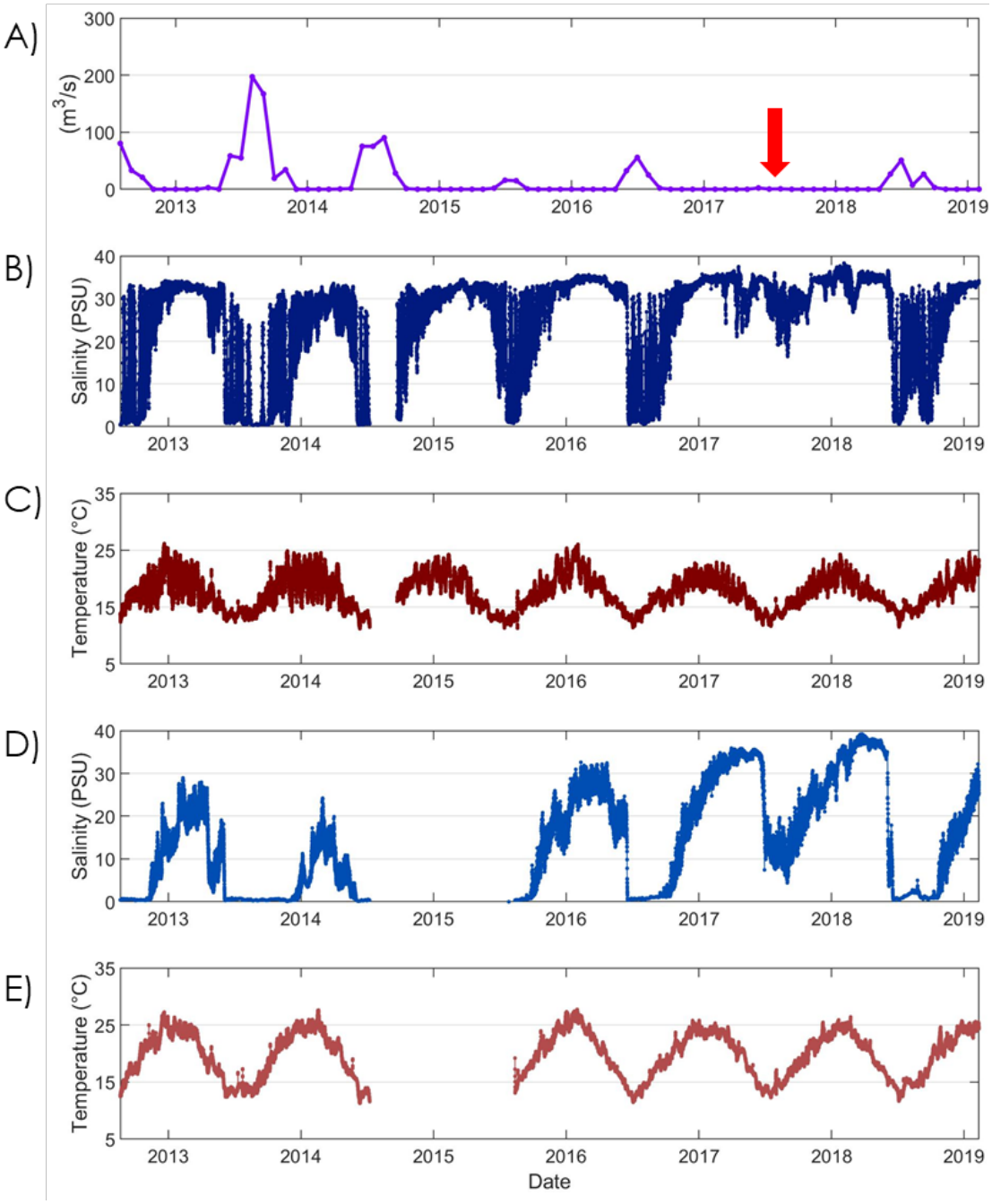

Figure 5.5. **Salinity and Temperature (°C) data from two continuous data loggers moored in the Berg River Estuary for the period 2012-2019. Aquatroll 2 was located at Carinus Bridge, 5.3 km from the mouth and Aquatroll 4 was located 23.7 km from the mouth. A) Variation in estimated freshwater inflows entering at the head of the estuary, B) DWS Aquatroll 2 – Salinity, C) DWS Aquatroll 2 – Temperature, D) DWS Aquatroll 4 – Salinity, E) DWS Aquatroll 4 – Temperature. Red arrow indicates the peak of the recent drought in 2017/2018.**

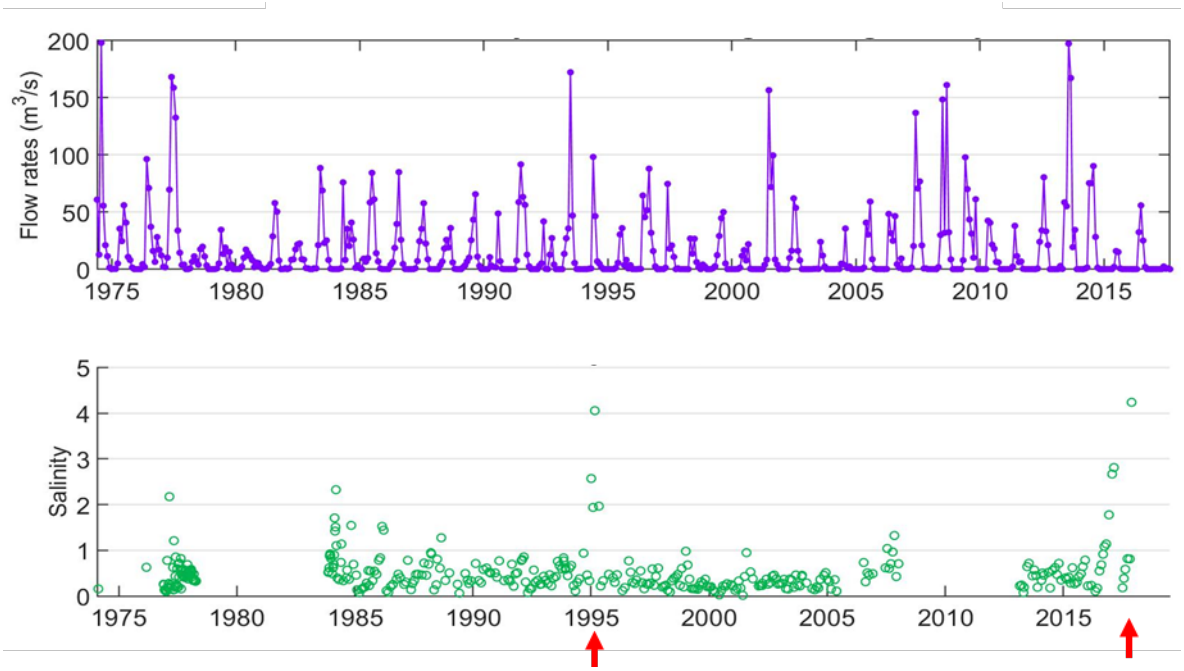

Figure 5.6. Long term DWS:RQIS data series showing variation in estimated freshwater inflows entering at the head of the estuary between Feb 1974 and Nov 2017 (Top) and salinity at Jantjiesfontein roughly 52 km upstream of the mouth (Bottom). Arrows indicate periods of increased salinity in what is normally fresh water.

5.2.2 Dissolved Oxygen

The natural levels of Dissolved oxygen (DO) are governed by temperature and salinity, as well as the organic content of the water. Colder and lower-salinity waters have a higher saturation limit (can hold more oxygen) than warmer, saltier waters. Oxygen is also removed from the water column, by respiration of biota and through decomposition of organic matter. Sufficient dissolved oxygen in sea water is essential for the survival of nearly all marine organisms. The well-known "black tides" and associated mass mortalities of marine species that occasionally occur along the west coast results from the decay of large plankton blooms under calm conditions after intense upwelling cycles (Lamberth *et al.* 2010). Once all the oxygen in the water is depleted, anaerobic bacteria (not requiring oxygen) continue the decay process, causing the characteristic sulphurous smell.

Chapman and Shannon (1985) identified St Helena Bay as a zone for the formation of oxygen-deficient waters, as a result of algal blooms (also known as black tides), which have in the past lead to major mortality events for organisms such as rock lobsters and fish (Bakun 1998, Cockroft *et al.* 2000, Monteiro & Roychoudhury 2005). The Berg River Estuary acts as a refuge for fish from low oxygen events in St Helena Bay (Lamberth *et al.* 2010).

Marine waters entering the mouth of the estuary can range from as low as 1 to as high as 10 mg/l. In addition, low oxygen (or anoxic conditions) can be caused by excessive discharge of organic effluents (from for example, fish factory waste or municipal sewage) as microbial breakdown of this excessive organic matter depletes oxygen in the water. Oxygen levels in the freshwater entering at the head of the estuary are mostly high (in the range of 8-12 mg/l) and vary depending on water temperature and organic content.

DO was lowest (<8 mg/l) in upper reaches of the estuary in summer (Figure 5.7 A), except during periods of intense upwelling at sea, when sea water with very low levels of dissolved oxygen is pushed up into the lower reaches of the estuary (Lamberth *et al.* 2010, Laird & Clark 2015). Summer DO is lower in the upper reaches because water temperatures are higher. Low flows and salinity intrusion may also result in stratification, which isolates bottom water. If organic detritus is present, the DO levels can then be depleted through decomposition, causing pockets of lower oxygen to form within the system. It is also apparent that, roughly 50 km upstream there is a lower DO body of water, especially in early summer (November, Figure 5.7 C). This could be at least partly ascribed to the abundance of rotting vegetation left in the river from the clearing of alien trees (*Eucalyptus* sp.; Schumann 2007).

In winter, highly oxygenated freshwater spates filling the estuary and flushing any oxygen depleted water plugs out of the system. Therefore, the upper estuary DO is higher than 10 mg/l with levels throughout the mid to lower reaches generally greater than 7 mg/l (Figure 5.7 B). Importantly, if winter rainfall is reduced (such as during drought periods), this limits or prevents the expulsion of these oxygen depleted plugs from the system, allowing them to persist for longer periods. When this happens, the plug of low oxygen water can act as a barrier for species that cannot tolerate low levels of DO. In addition, if the concentrations within these oxygen depleted plugs drops below 4 mg/l, juvenile fish species (which use the estuary as a nursery and are less mobile and therefore not able to move away from these plugs) may start to be negatively affected and can be seen gulping air at the surface of the water (Lamberth *et al.* 2010). Mass mortalities of fish can occur if concentrations drop below 2 mg/l (Whitfield 1995, Cyrus & McLean 1996, Lamberth *et al.* 2010).

In addition, dredging within the Port Owen Marina could also affect DO in the short-term. A fair portion of the deeper sediment that is removed and discharged into the estuary may be anoxic, causing oxygen levels within the lower reaches of the estuary, between the marina and the mouth, to decline temporarily until all the sediment has been flushed out the system. The dredging and discharge of the sediment can also result in a temporary decline in water clarity as tests have shown that visibility of secchi disc readings drops from 2.0 m to 0.47 m in the plume which is below the gazetted RQOs.

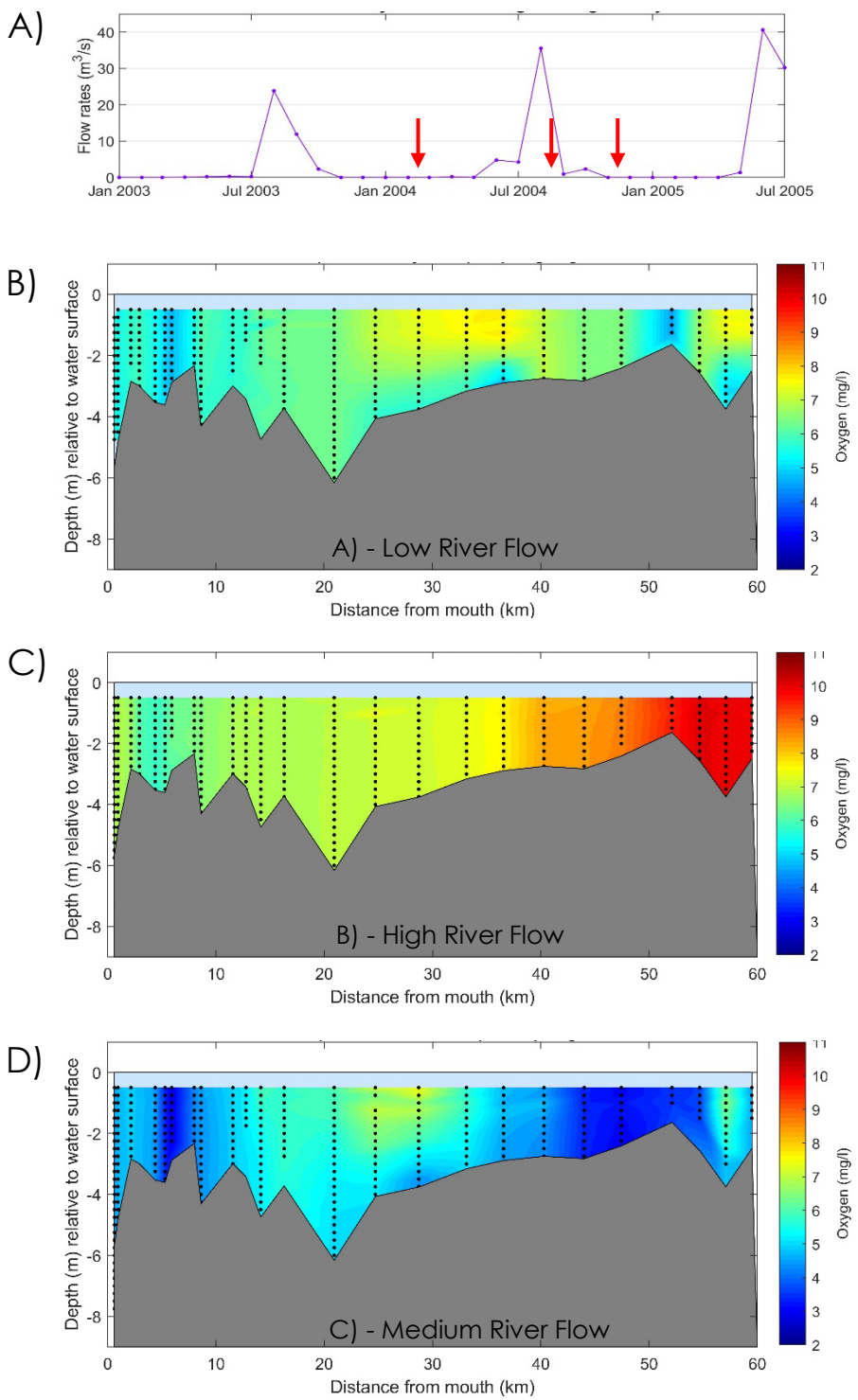

Figure 5.7. **Dissolved oxygen (mg.l^{-1}) within the Berg River Estuary during spring high tide in B) Summer (low flow - 13 Febuary 2004) and C) Winter (high flow – 18 August 2004), as well as in D) early summer (medium flow - November 2004) for neap high tide. Variation in freshwater inflows entering at the head of the estuary between 2003 and 2005 is shown in the top panel (A) as are the times at which sampling was undertaken (red arrows) to place these in context of "normal" variation in flow, outside of drought periods.**

5.3 Nutrients

Nutrients enter the estuary with both the sea and the river, with sea inputs dominating in summer (low flow season), and river inputs dominating in winter (high flow season. Nutrient cycling and transformation are one of the key abiotic components that drive biological production in estuaries. Under natural conditions, estuaries are typically oligotrophic (nutrient poor) systems, with primary production being limited by the availability of nitrogen (N) and in some cases also phosphorus (P; Nixon 1995, Herbert 1999, Flemer & Champ 2006). However, human-induced (anthropogenic) forcing often disrupts the natural balance of nutrient cycling and transformation, leading to changes in the systems' trophic status towards a more eutrophic state (Ferguson *et al.* 2004). Activities that influence nutrient dynamics in estuaries such as waste loading, agricultural runoff and reductions in freshwater inflows, can potentially impact the ecological services provided by these ecosystems.

The introduction of nutrients, such as nitrogen and phosphorus through land clearing, application of fertilizer, discharge of human wastes, animal production, and combustion of fossil fuels and urban runoff, can lead to eutrophication. This has been singled out as one of the major concerns for estuarine health across the globe (Nixon 1995, Kemp *et al.* 2005, Flemer & Champ 2006). One of the key outcomes is the stimulation of undesirable types and amounts of aquatic plant growth, particularly phytoplankton and nuisance macrophyte growth. In eutrophic estuaries, phytoplankton tends to proliferate at the expense of submerged and emergent vegetation as the microscopic phytoplankton species grow faster than the larger macroscopic plant species. As such, phytoplankton reduces the amount of light reaching the larger rooted plant species thereby inhibiting their growth. Other undesirable effects of nutrient enrichment include oxygen depletion or hypoxia which arises from the decomposition of phytoplankton and plant biomass. This can kill fish and invertebrates and cause a general reduction in biodiversity. It is therefore of concern that concentrations of nutrients have increased dramatically in most estuaries in South Africa in recent years.

5.3.1 Dissolved Inorganic Nitrogen (DIN)

Long term BRIP data which recorded the nutrient levels in the estuary were collected predominantly on a monthly basis throughout the year, but this was not standardised with the tidal cycle. Therefore, recordings could have been taken during the high tide when the ocean's influence is at its peak or during low tides when the ocean influence is lowest. The implications of this is that there can be substantial variability in the recorded values. In addition, the available data are limited to the lower to middle reaches of the estuary (up to 23 km upstream). These data were graphed in relation to distance from the mouth to provide an indication of the spatial distribution of the nutrients within the lower to middle reaches. In addition, we also present mixing diagrams (otherwise known as or property-salinity plots).

Mixing diagrams are widely used to assess nutrient cycling in estuaries (Church 1986, Fisher *et al.* 1988, Ball 1994, Eyre 2000, Ferguson *et al.* 2004). It is used where changes in the level of non-conservative parameters (which can be affected by biological activity), such as DIN or DIP, are plotted in relation to a conservative parameter (which is not typically depleted through biological activity), such as salinity. In a plot of nutrients against salinity, one might expect a straight-line relationship, which could have a positive, negative or zero slope, or may have an inflection point, depending the primary source of the element in question (river inflow, seawater or both). Any deviation from the theoretical straight-line relationship can be

interpreted as being a result of some process along the length of the system. Deviation above a straight line (concave curvature) could indicate that there is some addition in the intermediate salinity zone (e.g. through remineralisation or input from an external source) while deviation below the line (convex curvature) suggests some removal in that zone (Figure 5.8).

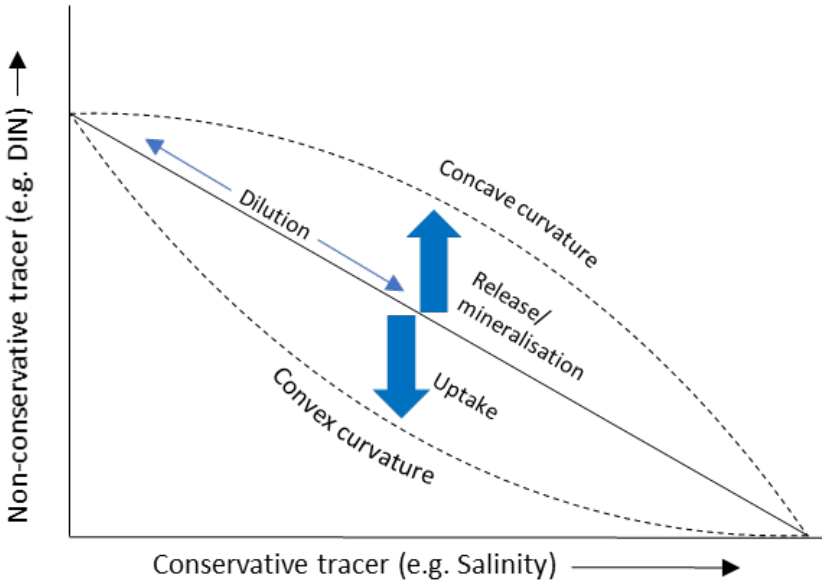

Figure 5.8. Example of a property-salinity plot where variations in a non-conservation tracer (such as Dissolved Inorganic Nitrogen or DIN) is plotted against a conservative tracer (such as salinity) to assess levels of uptake or release in the water body in question. The solid line indicates straight line dilution (no uptake or release) while the dotted lines indicate release (above) and uptake (below) the theoretical straight line relationship.

For this study, the concentrations of dissolved inorganic nitrogen (DIN), ammonia and dissolved inorganic phosphorus were plotted against salinity. Using simulated flow generated in the hydrological analysis, the data were separated into low and high freshwater flow periods. Generally, data points collected from December to the end of May were categorised as low flow, while those collected from June until the end of October were classified as high flow. The 2017 year was an exception in that data from June to October was included in the low flow category because of the drought.

The average annual DIN ((NOx-N or [Nitrite + Nitrate]-N) for the period 2013-2019 increased slightly with increasing distance from the mouth (Figure 5.9) suggesting that the majority of this nutrient is being introduced upstream. There were significantly higher DIN values throughout the estuary during high flow than during low flow periods, which also suggests that DIN was being introduced by freshwater inflow at the head of the estuary. Furthermore, DIN values in long term data did not show a seasonal winter spike during 2017 because freshwater flows were greatly reduced by the drought (Figure 5.10). While the average DIN values recorded during the low flows met the required RQOs (< 300 $\mu g.l^{-1}$), average high flow DIN values starting roughly 10 km from the mouth and further upstream exceeded the gazetted RQOs which state that DIN within the estuary should be less than 800 $\mu g.l^{-1}$ during high flow periods.

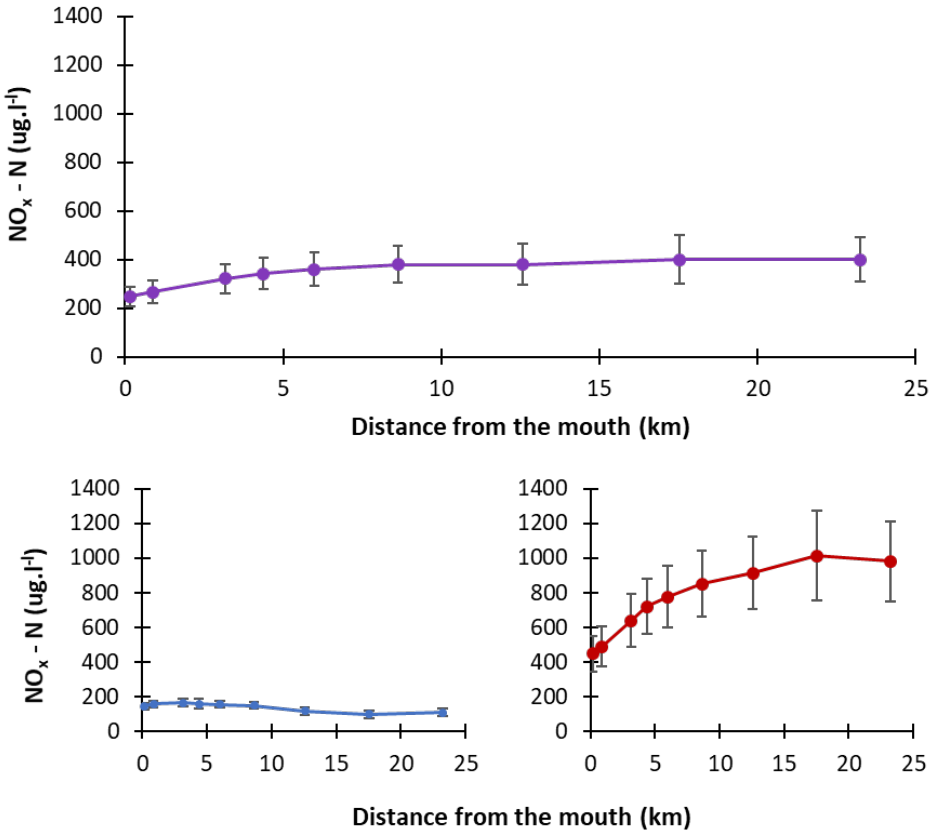

Figure 5.9. Average dissolved inorganic nitrogen (DIN ± Standard error) within the Berg River Estuary with increasing distance from the mouth. Values averaged over the entire sampling period of BRIP data (2013-2019) for the full year (top), during the summer low flow period (bottom left) and the winter high flow period (bottom right).

Typical values for DIN within St Helena Bay are reported as 124 µg.l^{-1}, while DIN values in newly upwelled water (upwelling occurring in summer) can reach 620 µg.l^{-1} (Bailey and Chapman 1991). The average low flow (summer) DIN at the mouth (144 µg.l^{-1}) is slightly higher than in the middle reaches as water from the ocean is forced into the estuary by the tides.

Property-salinity plots for DIN (NOx-N or [Nitrite + Nitrate]-N) for the period 2013 to 2019 indicated marked differences between the high and low flow events (Figure 5.11). Overall, levels of DIN were very much higher in the high flow than the low flow season. Both plots are convex in shape, indicating nutrient inputs both at the mouth and the head of the estuary (the latter being particularly important during the high flow season), and nutrient uptake (depletion) in the middle reaches. The level of depletion was lower during the high flow season probably because of lower water residence time and hence limited time for microalgae and macrophytes to take up these nutrients. In contrast, nutrients are more easily depleted in the plug of saline water that occurs in the estuary during the low flow season. The input of DIN from the catchment is also very much lower at this time of year than in the high flow season. Nutrient inputs from the sea during the low flow period are also higher than the high flow period, as this corresponds with the summer upwelling period.

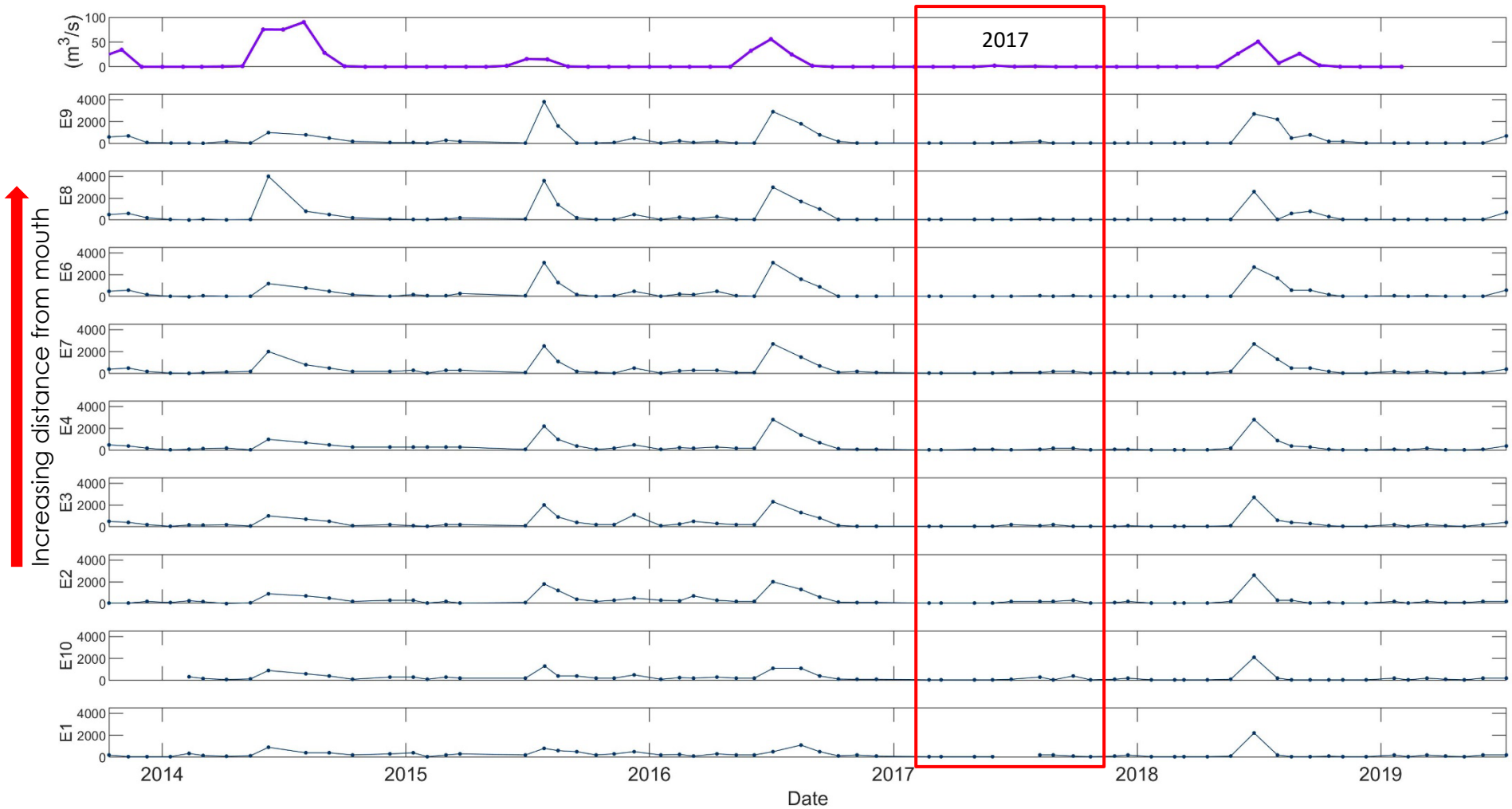

Figure 5.10. Long term BRIP data series showing variation in freshwater inflows entering at the head of the estuary (Top) and dissolved inorganic nitrogen (NO_x: NO_2 + NO_3, µg.l^{-1}) at 9 stations within the Berg River Estuary ranging from the mouth (E1) to roughly 23 km upstream (BRIP E9). Data in 2017 highlights the lack of a spike in NO_x as very little freshwater from the river entered the estuary during the drought.

Collectively these data suggest that DIN plays an important role in controlling primary production in the estuary, especially during the low flow season where it is clearly a major limiting nutrient.

Compared with historical data, which extends back to the 1980s (Figure 5.12), it is clear that levels of DIN in river inflow in the high flow season have increased dramatically over the years, from around 644.1 µg.l^{-1} in 1989 to around 1245.0 µg.l^{-1} measured in the last decade (2013-2019, an almost 2-fold increase, Table 5.1). This is presumably linked to increased agricultural activities and human settlements in the catchment (Cullis *et al.* 2019). DIN levels in seawater entering at the mouth of the estuary have hardly changed at all.

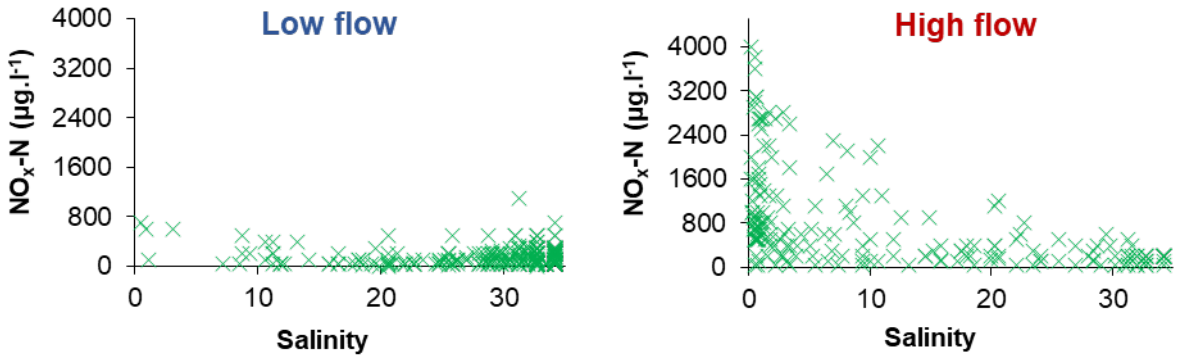

Figure 5.11. Property-salinity plot showing variation in levels of dissolved inorganic nitrogen (NO_x) relative to salinity in the low (left) and high flow seasons (right) in the Berg River Estuary from 2013-2019

Table 5.1. Mean concentration of dissolved inorganic nutrients (dissolved inorganic nitrogen, ammonia and inorganic phosphate) in freshwater (<1) flowing into the Berg River Estuary during high and low flow seasons between 1989 and 2019.

	Year	NOx-N (µg.l^{-1})	NH4-N (µg.l^{-1})	PO4-P (µg.l^{-1})
Low flow (Summer)	1990	70.0	57.0	41.0
	1996	36.6	32.7	13.7
	2005	37.0	21.1	6.4
	2013-2019	650.0	25.0	60
High flow (Winter)	1989	644.1	73.6	27.2
	1995	1193.2	56.3	44.5
	2005	880.0	48.1	38.6
	2013-2019	1245.0	123.9	68.4

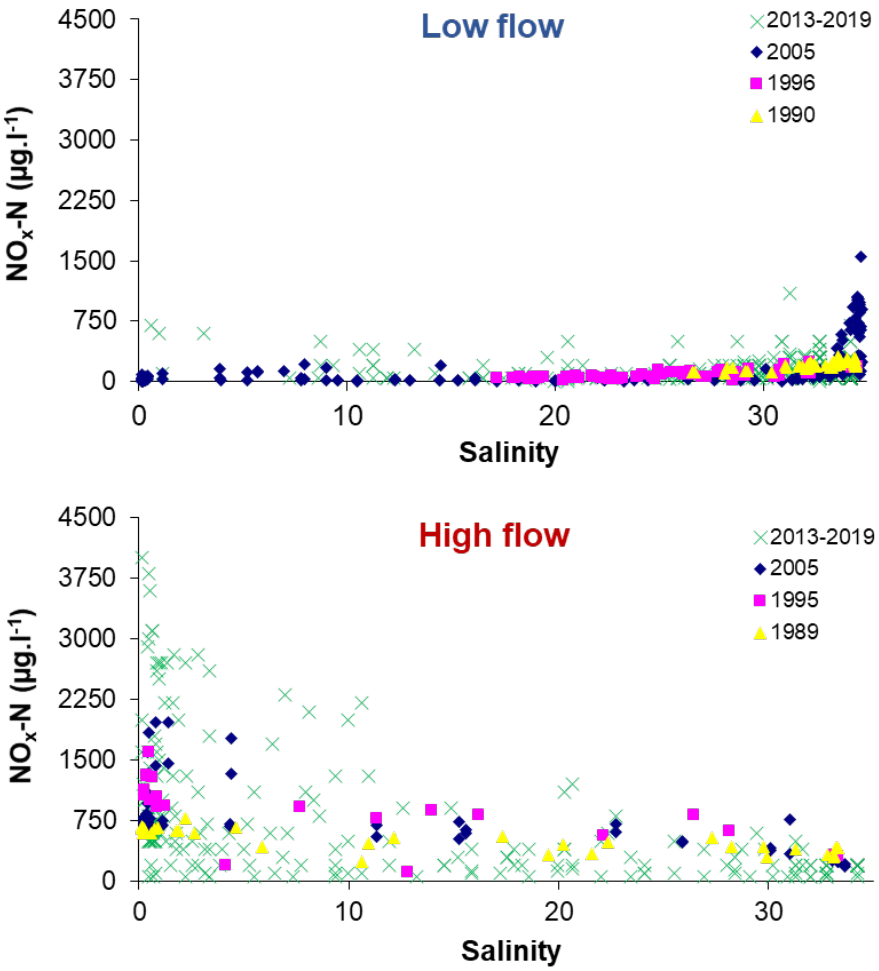

Figure 5.12. Property-salinity plot showing variation in levels of dissolved inorganic nitrogen (NO_2 + NO_3) relative to salinity in the Berg River Estuary during the low (top) and high flow seasons (bottom) between 1989 and 2019.

5.3.2 Dissolved Total Ammonia

The average annual dissolved total ammonia (NH_4-N) showed higher values close to the mouth with values generally decreasing with distance upstream for all periods (Figure 5.13). Typical values for ammonia within St Helena Bay are reported as 36 µg.l^{-1}, while values have been seen to reach 144 µg.l^{-1} in newly upwelled water, during summer (Bailey & Chapman 1991), both of which are significantly lower than values seen in the lower reaches of the estuary. This suggests that although tidal movement may introduce ammonia into the system, the ocean is not the only, or most significant, source of ammonia. It is likely that ammonia is also being introduced into the estuary by discharge from the fish factory located close to the mouth of the estuary (Figure 5.13 -BRIP Station E10 – 0.87 km from the mouth). In addition, ammonia can be introduced into the system during red tides (a large algal bloom during which algal cell numbers become so numerous that the water is discoloured red or orange) such as the one that occurred between December 2014 and March 2015 (Ndhlovu *et al.* 2017). During this time, high levels of ammonia were recorded within the lower reaches of the estuary, with readings reaching as high as 1300 µg.l-1 at BRIP Station E4 (6 km from the mouth)

and 1400 µg.l-1 at BRIP Station E8 (17.5 km from the mouth, Figure 5.14). All the nutrient readings (with one exception) taken during the red tide in March 2015 were higher than the average March values recorded in years during which no red tide occurred (Figure 5.14).

Property-salinity plots for dissolved total ammonia (NH_4-N, Figure 5.15) measured during low flow periods showed renewal only occurring across the seaward boundary, with mid-level readings (~20) slightly elevated (probably due to the tides pushing seawater upriver) and the estuary-river boundary showing near depleted readings. This relationship is attributed to biological uptake and (as for dissolved inorganic nitrogen) supports the suggestion that nitrogen is the limiting nutrient in the Berg River Estuary at these times.

During high flow periods, elevated total ammonia concentrations were maintained throughout the estuary, ascribed to short flushing times with regular renewal across source boundaries and short retention times limiting the effects of biological uptake. However, although the majority of the values were below 500 µg.l^{-1} a few higher readings suggesting that total ammonia was introduced to the water column in the mid-salinity (or middle reaches). This pattern is usually indicative of internal processes releasing N into the water column through remineralisation or anthropogenic inputs containing N-enriched effluents. However, these processes and activities are usually not expected to have such marked effects during high flow periods when river water volumes and flushing rates are relatively high.

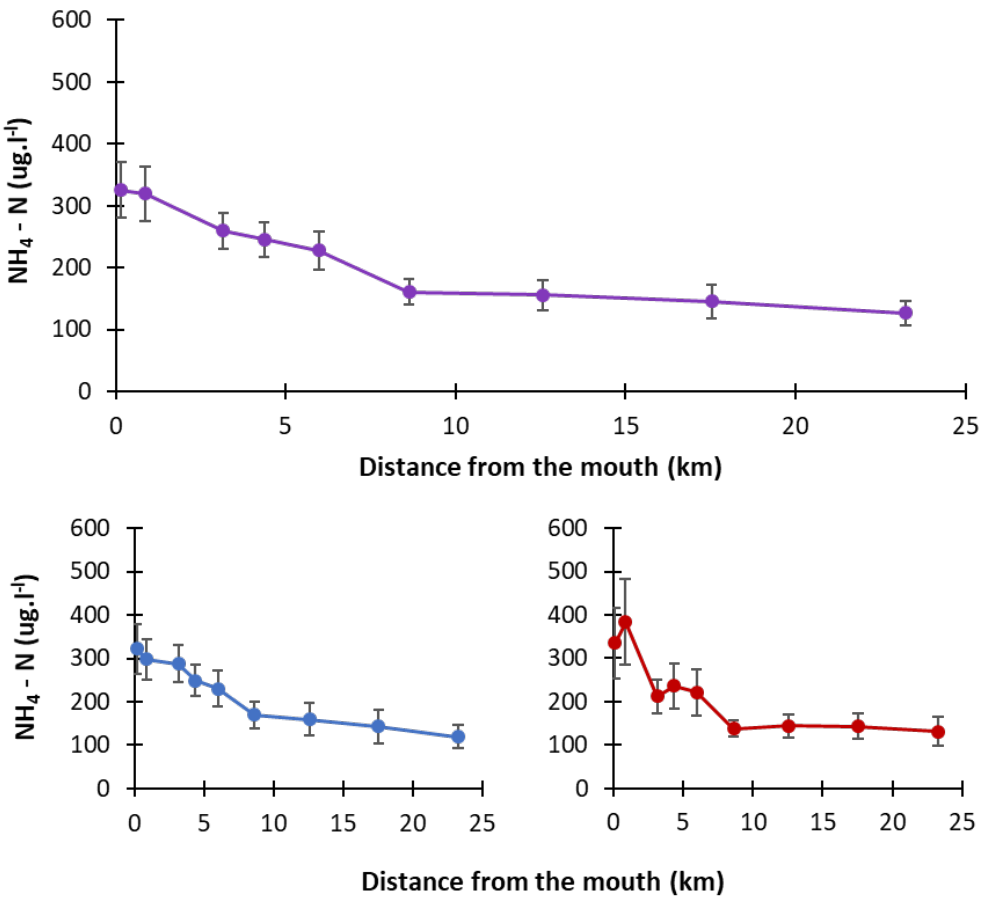

Figure 5.13. Average total Ammonia (± Standard error) within the Berg River Estuary with increasing distance from the mouth. Values averaged over the entire sampling period of BRIP data (2013-2019) for the full year (top), during the summer low flow period (bottom left) and the winter high flow period (bottom right).

A comparison with historic data is provided in Figure 5.16 where property-salinity plots of total ammonia suggest a similar trend for low and high flow periods, respectively, over the years. However, in recent years there appears to be a marked increase in concentrations in the lower and middle reaches (high and mid-level salinity range) of the estuary during low flow periods which suggests that nutrients entering at the mouth and from the fish factory are possibly now being advected further upstream than they were in the past (which corresponds with increased salinity levels throughout the estuary associated with reduced freshwater inflow) and also that levels of mineralisation (or release of ammonia) in the middle reaches of the estuary may have increased in recent years. This is most likely linked with increased residence time of the plug of warm saline water that is present in the estuary during the low flow periods, and death and decomposition of living organisms in this zone. Anecdotal reports indicate a mass mortality and subsequent decomposition of cuttlefish *Sepia officinalis* occurred in the middle reaches of the estuary (Kliphoek) during the drought (2017-2018). This mass mortality may have been a result of a sudden salinity decrease to below tolerable levels for this marine cephalopod that had penetrated further upstream than normal due to the prolonged high salinity. Whatever the cause, this observation does give credence to the hypothesis that death and decomposition of organisms in the middle reaches may have contributed to the observed elevated ammonia levels. Levels of total ammonia entering the estuary during the high flow season from the catchment have also increased markedly in the last decade relative to historic times from 74 µg.l^{-1} in 1989 to levels 124 µg.l^{-1} in the last decade (Table 5.1).

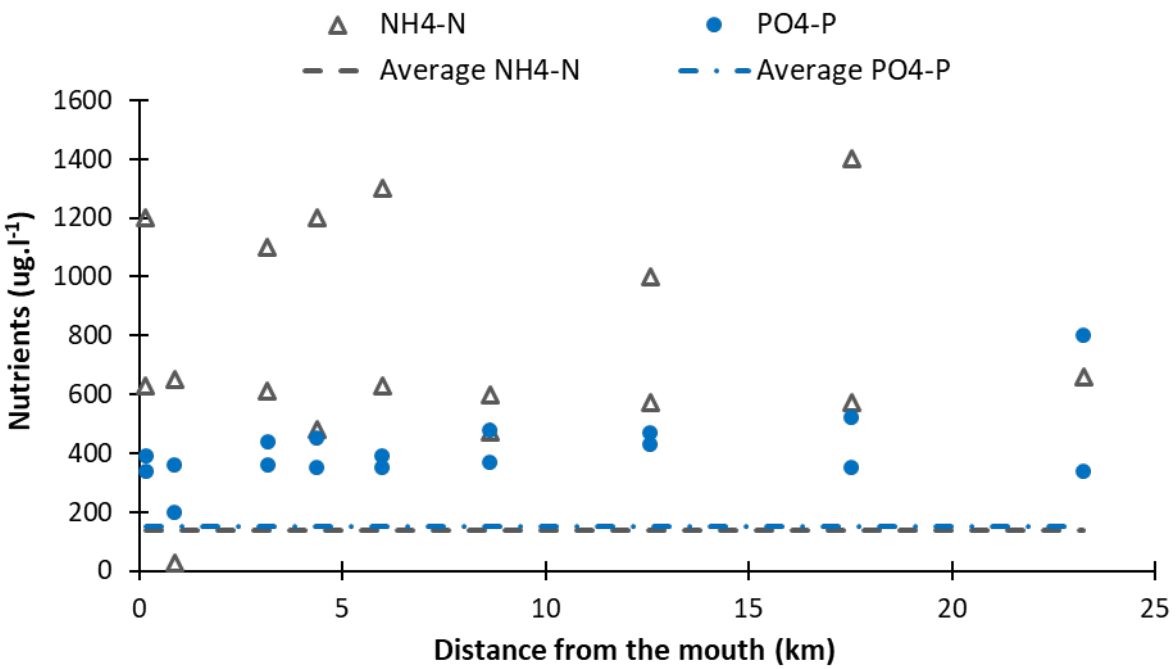

Figure 5.14. **Total Ammonia (NH4-N) and total dissolved inorganic phosphate (PO4-P) measured in March 2015 during a red tide. Lines show the average values for nutrients recorded in March of years when no red tide occured (2014, 2016-2019).**

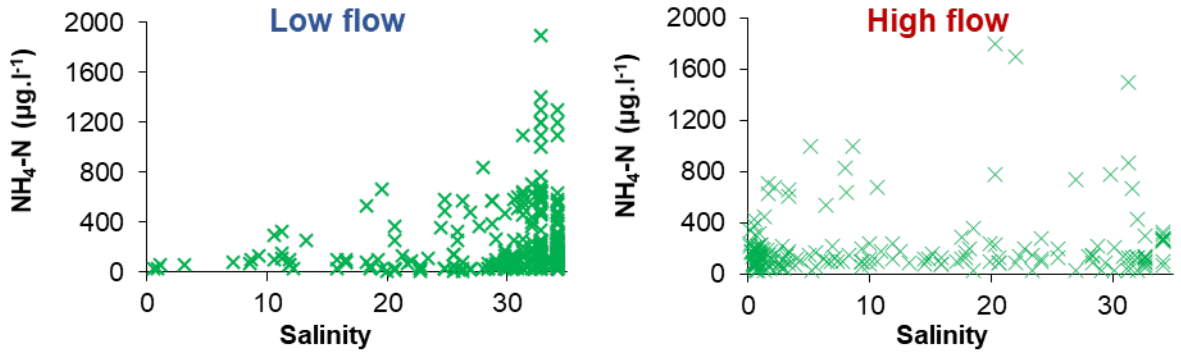

Figure 5.15. Property-salinity plot showing variation in levels of total ammonium (NH_4-N) relative to salinity in the Berg River Estuary in the low (left) and high flow seasons (right) from 2013-2019 .

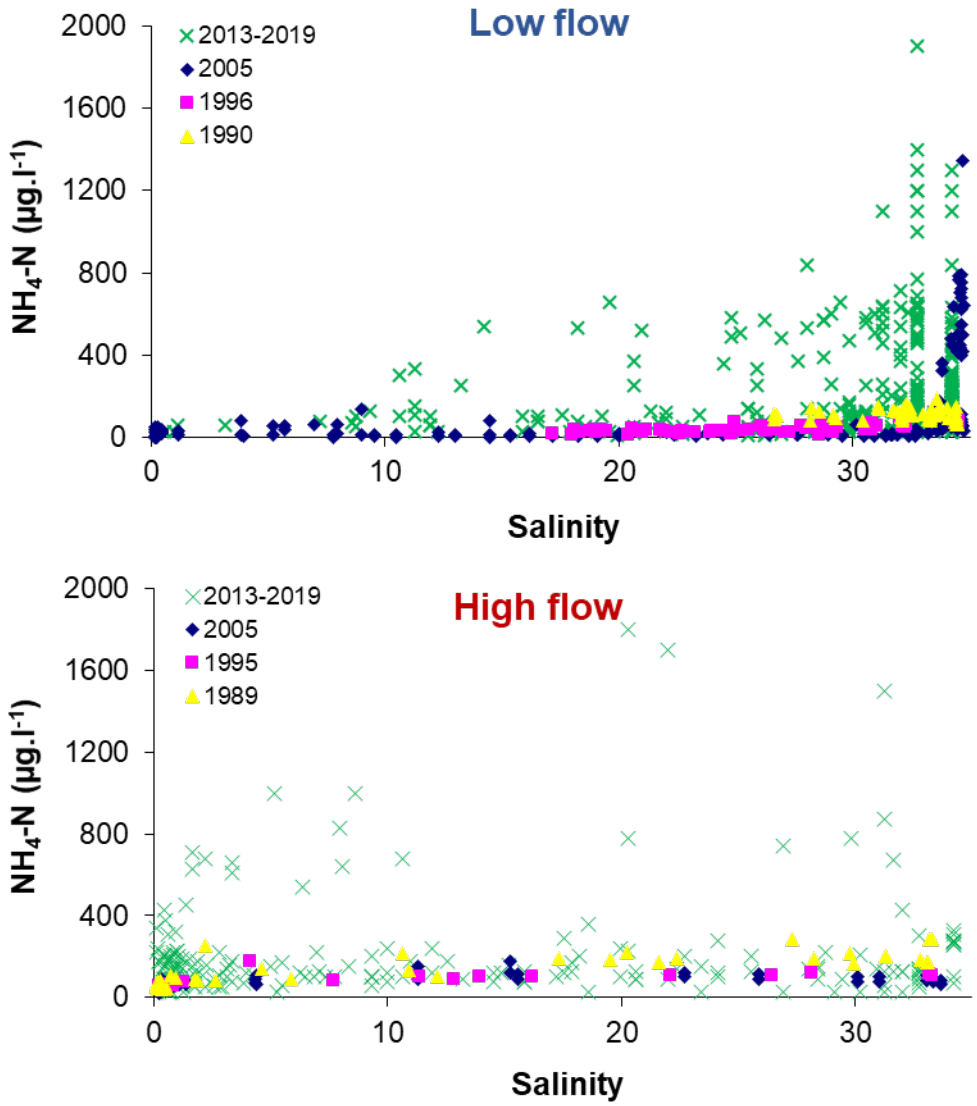

Figure 5.16. Property-salinity plot showing variation in levels of total ammonium (NH_4-N) relative to salinity in the Berg River Estuary measured in the low and high flow seasons between 1989 and 2019.

5.3.3 Dissolved Inorganic Phosphate (DIP)

The average annual and seasonal dissolved inorganic phosphate (DIP) showed similar patterns during all periods with levels not varying greatly from the mouth to middle reaches of the estuary, however, the summer DIP was higher than that of winter values, when river flow was high. This is indicative of the introduction of phosphates from the ocean by tidal exchange as newly upwelled water within St Helena Bay is known to reach a maximum of 171 $\mu g.l^{-1}$while typical and winter values within the bay are 94 $\mu g.l^{-1}$ (Bailey & Chapman 1991). As above, readings during the red tide recorded between December 2014 to March 2015 increased, reaching as high as 530-800 $\mu g.l^{-1}$. It is important to note that at current levels seen in Figure 5.17 average DIP exceeds the gazetted RQOs in both the low flow, exceeding the 100 $\mu g.l^{-1}$ limit, and during high flow as they exceed the 60 $\mu g.l^{-1}$ limit.

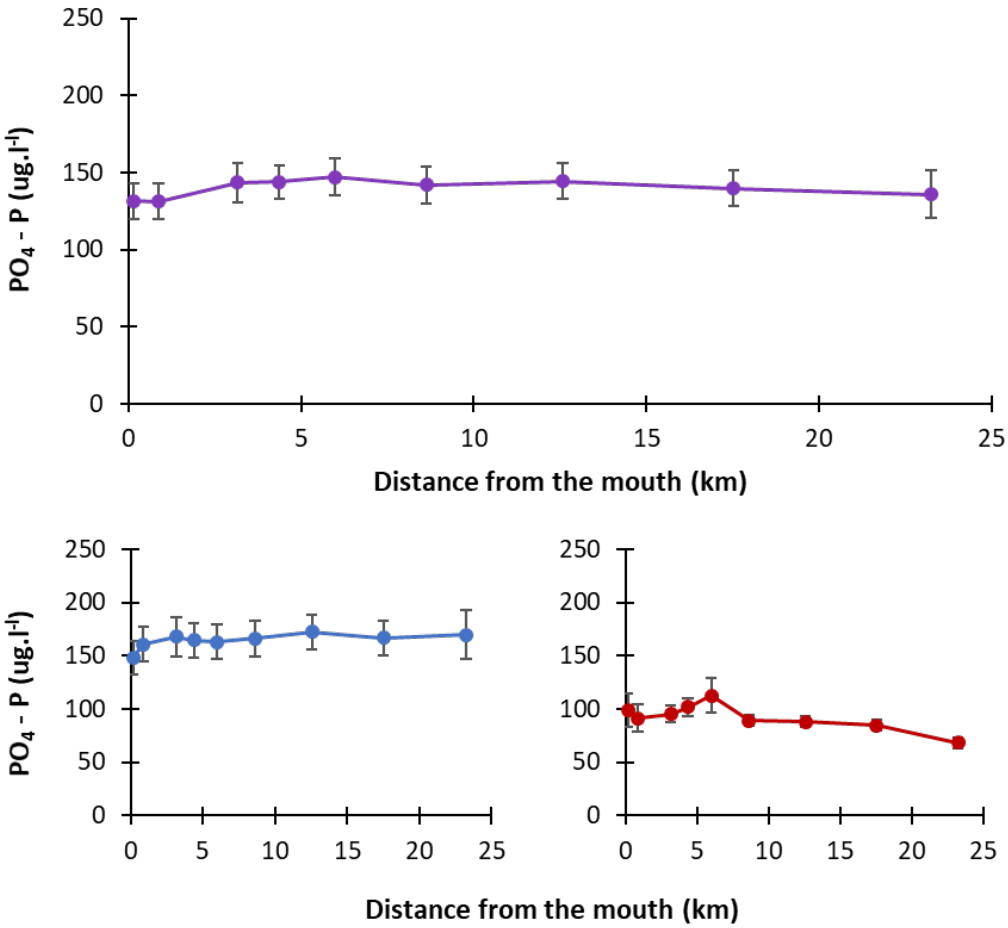

Figure 5.17. Average total dissolved inorganic phosphate (DIP ± Standard error) within the Berg River Estuary with increasing distance from the mouth. Values averaged over the entire sampling period of BRIP data (2013-2019) for the full year (top), during the summer low flow period (bottom left) and the winter high flow period (bottom right).

Property-salinity plots for dissolved inorganic phosphate (DIP or PO_4-P) measured during low and high flow periods showed distinctly different distribution patterns (Figure 5.18). During low flow periods, DIP generally declined up to a salinity of about 10 from which point concentrations remained low throughout. This suggests that fairly high levels of DIP are being

introduced at the mouth of the estuary at times (most likely coinciding with periods of intense upwelling, levels exceed 500 µg.l^{-1} at times) and introduced by the fish factory and flooding of local septic tanks), that DIP is being diluted with distance upstream (suggests that levels of DIP in the inflowing freshwater is low), and that DIP is being sequestered from the water column by primary producers in the upper reaches of the estuary where it may be a limiting nutrient (<200 µg.l-1). An exception to this was in March 2015 during the aforementioned red tide, when high levels of DIP were recorded upstream with readings of 800 µg.l-1 at BRIP Station E9, 20km upstream of mouth, 520 µg.l-1 at BRIP Station E8 just downstream and 440 µg.l-1 at the mouth (Figure 5.14).

In the high flow season, DIP concentrations were generally in the range of 75-200 µg.l^{-1} with no clear pattern emerging. Levels of DIP at the head of the estuary were slightly higher in the high flow season (50-150 µg.l^{-1}) than the low flow season (<100 µg.l^{-1}), consistent with measurements from the catchment (Cullis *et al.* 2019). DIP was lower at the mouth in the high flow season than the low flow season (mostly < 150 µg.l^{-1}), which is also consistent with the fact that upwelling in the sea is much less intense at this time of year (winter).

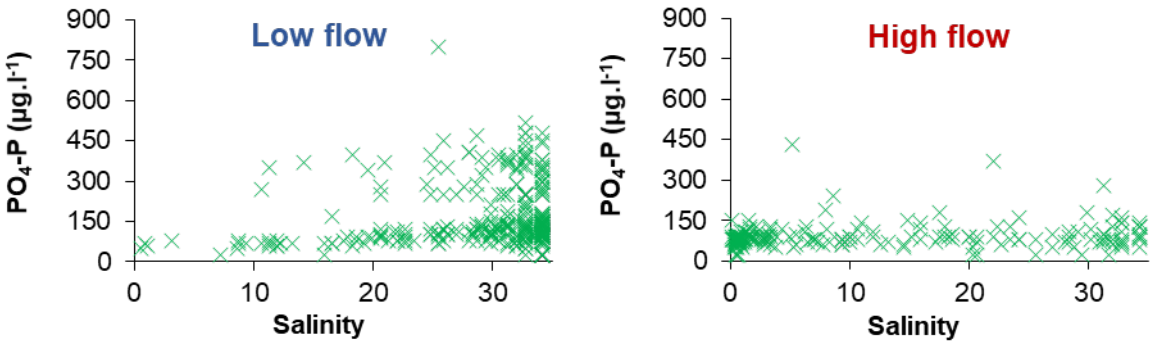

Figure 5.18. Concentration of dissolved inorganic phosphate (PO4-P) in the Berg River Estuary from 2013-2019 measured in the low and high flow seasons.

A comparison with historic data is provided in Figure 5.19, where property-salinity plots for DIP show similar trends for low and high flow periods, respectively, over the years. However, there has been a marked increase in DIP concentrations in the lower portions of the estuary during low flow periods (summer), where concentrations increased from around 69 µg.l^{-1} in 1990 to around 170 µg.l^{-1} in 2013-2019 (Table 5.2). It is assumed that this is linked with a reduction in freshwater inflows and an associated increase in the intrusion of saline water upstream as well as pollutants from the fish factory and local septic tanks. Levels of DIP in the estuary during winter have also increased generally in the last decade relative to the historic data 60 µg.l^{-1} in 1975 to 98 µg.l^{-1} in recent years (Table 5.2). Very little of this DIP is used up on its passage through the estuary, again suggesting that this is not likely to be a limiting nutrient in this system.

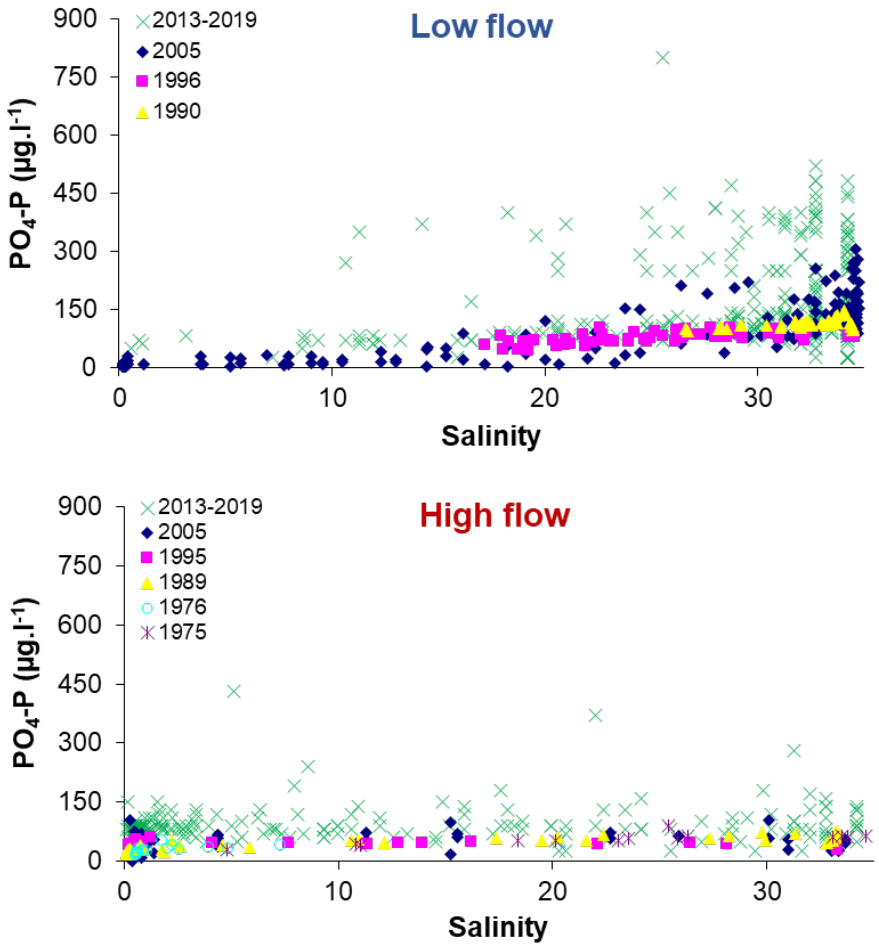

Figure 5.19. Concentration of dissolved inorganic phosphate (PO4-P) in the Berg River Estuary measured in the low and high flow seasons between 1975 and 2019.

Table 5.2. Mean concentration of dissolved inorganic nutrients (dissolved inorganic nitrogen, ammonia and inorganic phosphate) in Seawater (>32) entering the Berg River Estuary during high and low flow seasons between 1975 and 20019.

	Year	NO_x-N (µg.l^{-1})	NH_4-N (µg.l^{-1})	PO_4-P (µg.l^{-1})
Low flow (Summer)	1990	439.3	80.3	68.6
	1996	70.7	31.5	106.3
	2005	544.6	358.8	155.6
	2013-2019	130.1	243.0	169.6
High flow (Winter)	1975	-	-	59.3
	1976	-	-	33.4
	1989	287.6	149.5	90.4
	1995	314.2	113.4	34.9
	2005	241.3	81.0	32.4
	2013-2019	170.0	235.3	98.3

5.4 Turbidity

Seasonal changes to freshwater flows also typically result in changes in turbidity (i.e. the clarity of the water). Seawater tends to be comparatively clear relative to freshwater inflows, which tend to accumulate particulate matter on their way to and down the main channel through scouring. Floods in particular carry a lot of silt from the catchment, and generally occur during the high rainfall winter season. Therefore, during the summer, a reduction in freshwater flows typically lead to decreased turbidity and increased light penetration in the system.

5.5 Microbial indicators

Untreated sewage or storm water runoff may introduce disease-causing micro-organisms into coastal waters through faecal pollution. These pathogenic micro-organisms constitute a threat to recreational water users and consumers of seafood. Faecal coliforms and *Escherichia coli* (*E. coli*) because of their association with mammalian faecal matter as well as decaying vegetable matter, and the ease with which they can be enumerated, they are often used as indicators of the possible presence of other water borne pathogens in water bodies that are used for recreational purposes.

The Hazen non-parametric statistical method is recommended for dealing with long-term microbiological data that do not typically fit a normal (bell shaped) distribution. Data are ranked into ascending order and percentile values are calculated using formulae incorporated in the Hazen Percentile Calculator (McBride and Payne 2009). In order to calculate 95th percentiles, a minimum of ten data points is required, while the calculation of the 90th percentile estimates require only five data points. Rather than using a measure of actual bacterial concentrations, a compliance index is used to determine deviation from a fixed limit (DEA 2012). This method is being increasingly used across Europe to determine compliance in meeting stringent water quality targets within specified time frames (Carr & Rickwood 2008). Compliance data are usually grouped into broad categories, indicating the relative acceptability of different levels of compliance. For example, a low count of bacteria yields a rating of 'Excellent', while a 'Poor' rating would indicate presence of high levels of bacteria. Target limits, based on counts of *E. coli*, for recreational water use in South Africa are indicated in Table 5.3.

Table 5.3. Target limits for *E. coli* based on the revised guidelines for recreational waters of South Africa's coastal marine environment (DEA 2012). The probability of contracting a gastrointestinal illness (GI) is also listed.

Category	Estimated risk per exposure	*E. coli*. (count/100ml)
Excellent	2.9% GI risk	≤ 250 (95th percentile)
Good	5% GI risk	≤ 500 (95th percentile)
Sufficient/Fair (min. requirement)	8.5% GI risk	≤ 500 (90th percentile)
Poor (unacceptable)	>8.5 % GI risk	>500 (90th percentile)

E. coli data collected at high risk sites in the lower reaches of the estuary by the St Helena Bay Water Quality Trust (SHBWQT) for the annual State of the Bay report (Laird *et al.* 2018), show that since data capture started in 2008, sites close to the mouth are generally categorised as having "poor" overall annual water quality with regards to *E. coli* counts, while the Port Owen Slipway/Yacht Club and site located at Carinus bridge ranged from "poor" to "excellent" between 2008-2018 with later years showing generally improved categories (Table 5.4).

Table 5.4. St Helena Bay Water Quality Trust: Sampling site compliance for recreational use based on *E. coli* counts. Ratings were calculated using Hazen percentiles with the 90th and 95th percentile results grouped together to give an overall rating per annum. Modified from Laird *et al.* 2018.

Location	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Berg River Fishing Quay	Poor	Poor	Fair	Poor	Poor	Fair	Poor	Poor	Poor	Poor	Poor
Amawandle (Laaiplek)	Poor	Poor	Poor	Poor	Poor	Poor	Poor	Poor	Poor	Poor	Poor
Berg River Fishing Quay2	Fair	Fair	Poor	Poor	Poor	Fair	Poor	Poor	Fair	Poor	Poor
Port Owen Yacht Club	Fair	Poor	Good	Good	Good	Excellent	Excellent	Excellent	Good	Excellent	Good
Main Road (R27 Bridge)	Fair	Fair	Good	Poor	Poor	Fair	Good	Fair	Excellent	Good	Fair

Data collected by the BRIP over the period 2013 to present (Figure 5.20) from the lower and middle reaches of the estuary (up to 23 km upstream) suggest that average *E. coli* levels fluctuate throughout the year with a peak count occurring in both the high and low flow periods, this suggests that the *E. coli* count is not seasonal or dependant on freshwater flow rates. Averaging the data at each station shows overwhelmingly that the major source of contamination for *E. coli* is at the mouth of the estuary at BRIP Station E10 (Figure 5.20 bottom) which is located next to the Laaiplek Fishing Harbour and Amawandle Pelagic (Pty) Ltd. *E. coli* levels at this site exceeded the prescribed limit above which a water body is considered "poor" or "unfit" for recreational purposes in 8 of the 70 sampling events (11%). This limit is the same as the gazetted Resource Quality Objective (RQO) for *E. coli* within the berg (≤500 *E. coli*/100 ml (90th percentile, Hazen System). The Amawandle fish processing plant was issued a Coastal Waters Discharge Permit (CWDP) in May 2018 for continuous discharge of fish processing effluent, cooling water and brine effluent from a desalination plant into the Berg River Estuary. However, as of late 2018, no monitoring report had been issued and it is unknown whether this organisation is compliant with the CWDP (Laird *et al.* 2018), results seen here would suggest that the fish factory may be of major concern contributing to reduced water quality within the Berg River Estuary as these elevated *E. coli* count readings exceed the limits for safe recreational use and might pose a risk to the tourism industry, reducing the

suitability of the area for recreational use such as swimming, fishing and boating. The high *E.coli* count may also pose a threat to the fish factory itself as the majority of the water used within the plant is taken directly from the estuary.

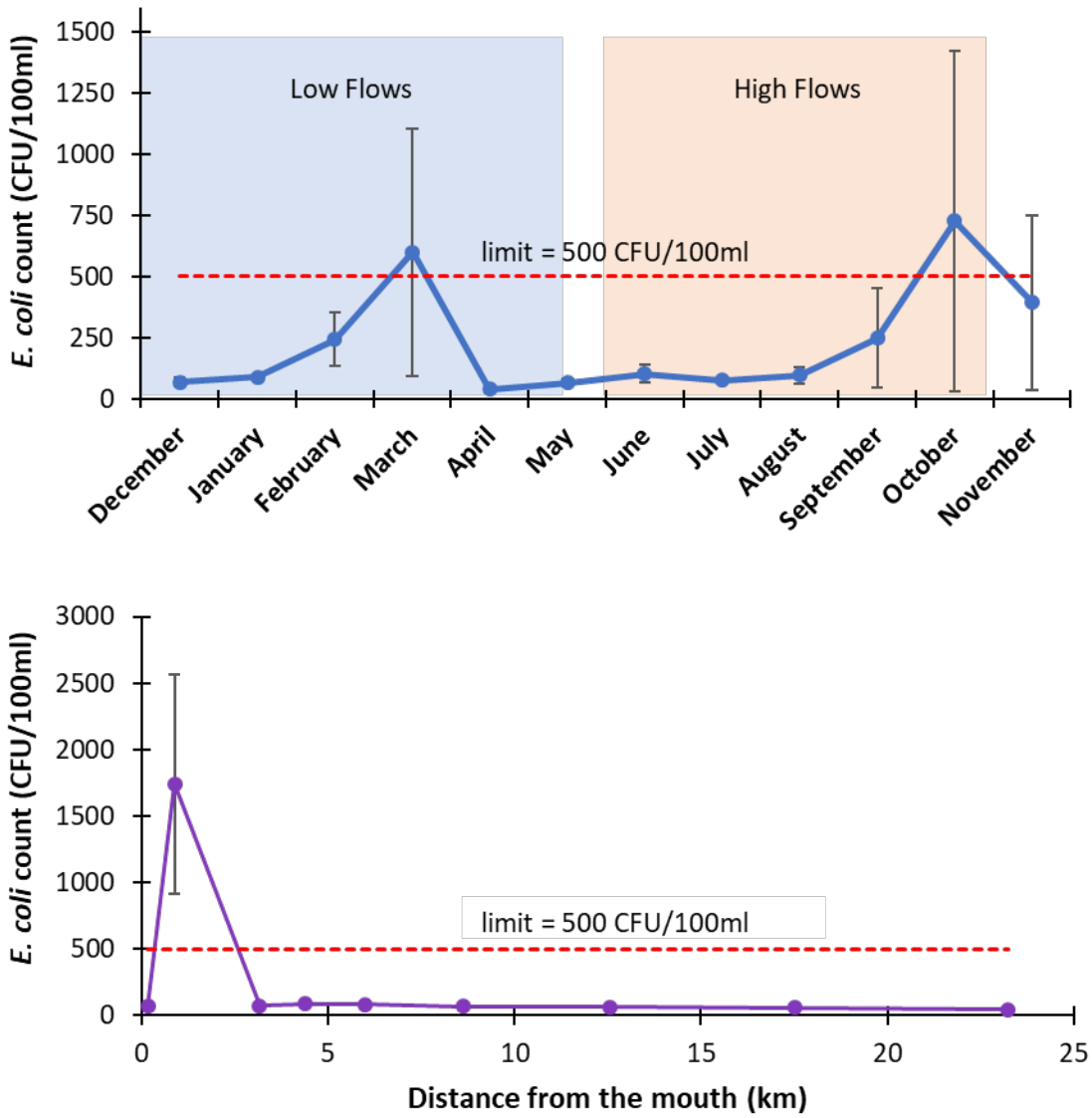

Figure 5.20. Average monthly (top) and average annual (bottom) *E. coli* counts (per 100 ml water, ± Standard error) for the period 2013-2019 at nine stations within the Berg River Estuary ranging from the mouth (Estuary 1) to roughly 23 km upstream (Estuary 9). Red lines indicate the prescribed limit above which *E. coli* counts per 100 ml become 'poor' or unacceptable for recreational purposes.

In addition, there is anecdotal evidence that during extreme high-tides and spring tides, water levels rise high enough to intrude into old septic tanks along the estuary banks, as well as soakaways on the fish factory and all along the quay wall. This can cause a spike in the pollutants within the estuary, especially in the form of added *E. coli* from the septic tanks. The intrusion of estuarine water into septic tanks has been confirmed by the presence of above average salinities within the septic tanks adjacent to the estuary. Until recently all facilities

along the Laaiplek quay wall, including the Laaiplek hotel, were still on this old septic tank and/or soakaway system. These septic tanks were frequently pumped out however, intrusion of saline water still occurred periodically, especially when water levels were high. The houses/businesses to the east of Port Owen up to Carinus bridge were also on the old system of septic tanks and/or soakaways, which may have increased the likelihood of the occurrence of *E. coli* levels exceeding the prescribed limit above which a water body is considered "poor" or "unfit" for recreational purposes. As part of the upgrade underway to the Velddrif WWTW these areas have been linked to the commercial sewage, and a new pump station will be constructed between the Amawandle fish factory and Laaiplek Hotel to link both with the waterborne sewage network and to reduce the risk of septic tanks being flooded and discharging waste water into the estuary. However, the area surrounding Pelican Harbour still remains on the old septic tank system and may continue to contribute to the elevated levels of *E. coli* in this area. Other possible contributors of pollution to the estuary include urban stormwater drains which are flooded doing storms or extreme tidal events, the addition of debris and waste from boat docking areas and yacht mooring sites, as well as waste from fishing and recreation which occurs in areas without localised sanitation facilities.

5.6 Heavy metals

While heavy metals are linked to influent water quality, they are best detected in sediments. Measured trace metal concentrations in the estuary sediments at some stations exceeded South African and international sediment quality guidelines certain metals in 2010 including Arsenic (As), Cadmium (Cd), Chromium (Cr), Copper (Cu), Mercury (Hg), Nickel (Ni) and Lead (Pb) (Figure 5.21) (Hutchings & Clark 2010).

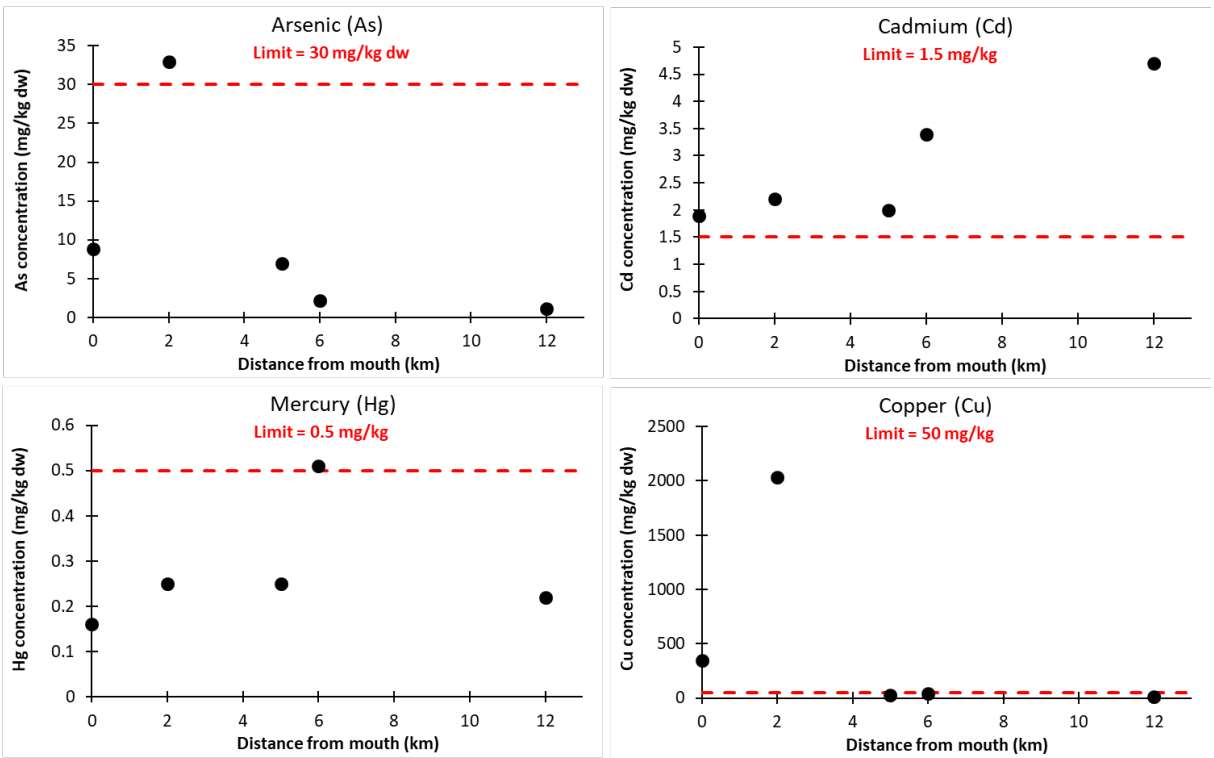

Figure 5.21. Sediment trace metal concentrations at five sites up the Berg River Estuary, measured from the mouth (km). The sediment quality guidelines are illustrated in red (South African special care concentrations for Arsenic, Chromium and Mercury and NOAA ELR limits for the remainder) (Hutchings & Clark 2010).

There is little evidence for significant or widespread metal contamination of sediment in the Berg River and estuary (CSIR 2016). The CSIR (2016) However, reported concentrations of cadmium and manganese at some stations in the Berg River Estuary were high, potentially as result of natural processes (i.e. upwelling), and a focused study of the estuary and Port Owen Marina has been recommended to confirm this. The report also strongly recommended that ongoing monitoring take place to assess metal concentrations in the estuary and broader catchment (at 4-5 year intervals, at a once off basis at end of dry season), and that these results be applied to the sediment baseline models that have been developed. Organic contaminants are also a concern, given the reported poor water quality in the system (i.e. excessive faecal indicator bacteria counts and high nutrient concentrations). It is recommended that polycyclic aromatic hydrocarbons (PAHs) be monitored as a proxy measure for these organic compounds, because PAHs have both anthropogenic (e.g. oil) and natural (e.g. bush fires) origins, behave in a similar manner to other contaminates such as pesticides, and are easier and more cost effective to measure and analyse than organic compounds.

5.7 Present status and potential changes under different scenarios

Salinity is often used to describe the abiotic state within estuaries. In the Berg River Estuary freshwater flow is very seasonal, resulting from winter rainfall in the mountainous catchment to the southeast. These winter spates of freshwater flush out sea water which has penetrated into the lower to mid-reaches of the estuary during the summer, when little rain falls. Due to the combination of tides and freshwater flow the Berg River Estuary has been characterised into 5 typical abiotic states, summarised in Table 5.5. In characterising the hydrodynamic and water quality characteristics within each abiotic state the estuary was sub-divided into 4 distinct zones (A to D), as illustrated in Figure 5.22. These zones were largely derived from typical salinity distributions and channel bathymetry.

The water quality scoring was based on the Resource Directed Measures (RDM) methodology (DWA 2012). The change in salinity was evaluated based on the change in average salinity which was calculated as the average salinity per state for a zone, multiplied by the percentage occurrence of the state. The latter was determined using the monthly flow range characterised by each state in Table 5.5 and calculating how often these ranges occur in the modelled monthly inflows reaching the estuary for each hydrological scenario.

Average salinity has increased relative to the reference condition. Present salinities for Zone A and D are similar to 2010, but higher for Zone B and Zone C (Table 5.6). Decreased winter freshwater flows prevent the complete flushing of these zones. Salinity values for scenario 1-3 will not differ greatly from present as flows will not be dramatically altered, scenarios in which EWRs are included show slightly healthier (lower) salinities than those without EWRs. In contrast, under climate change conditions, salinities increase across all zones because of two factors. Firstly, sea level rise (estimated at 1.87 mm/yr by Mather *et al.* 2009) will result in an increased water level within the estuary and push sea water further upstream. Secondly, reduced flushing of the system will increase the likelihood of salinity plugs occurring as well as the occurrence of reverse salinity gradients (similar to that seen in the start of 2018, Figure 5.3) within the system.

Table 5.5. Typical abiotic states of the Berg River Estuary, from Taljaard *et al.* 2010.

STATE	Brief Description	Monthly Flow Range (m^3/s)
1	Severe marine-dominated - saline intrusion extends further than 45 km upstream of mouth (into Zone D, see Figure 5.22)	<0.5
2	Marine-dominated - saline intrusion extends up to 45 km from mouth (downstream of Zone D, see Figure 5.22)	0.5 - 1
3	Small to medium freshwater inflow – marine influence evident up to 33 km from mouth (i.e. downstream of Zone C), with strong freshwater influence in upper ~40 km (in Zones C and D)	1 - 5
4	Medium to high freshwater inflow – marine influence only evident up to 12 km from mouth (downstream of Zone B), with strong freshwater influence in upper ~60 km (in Zones B-D)	5 - 25
5	Freshwater-dominated – estuary is fresh throughout (Zones A-D), except during spring tides when seawater intrusion may extend up to 6 km from mouth into Zone A during high tides	>25

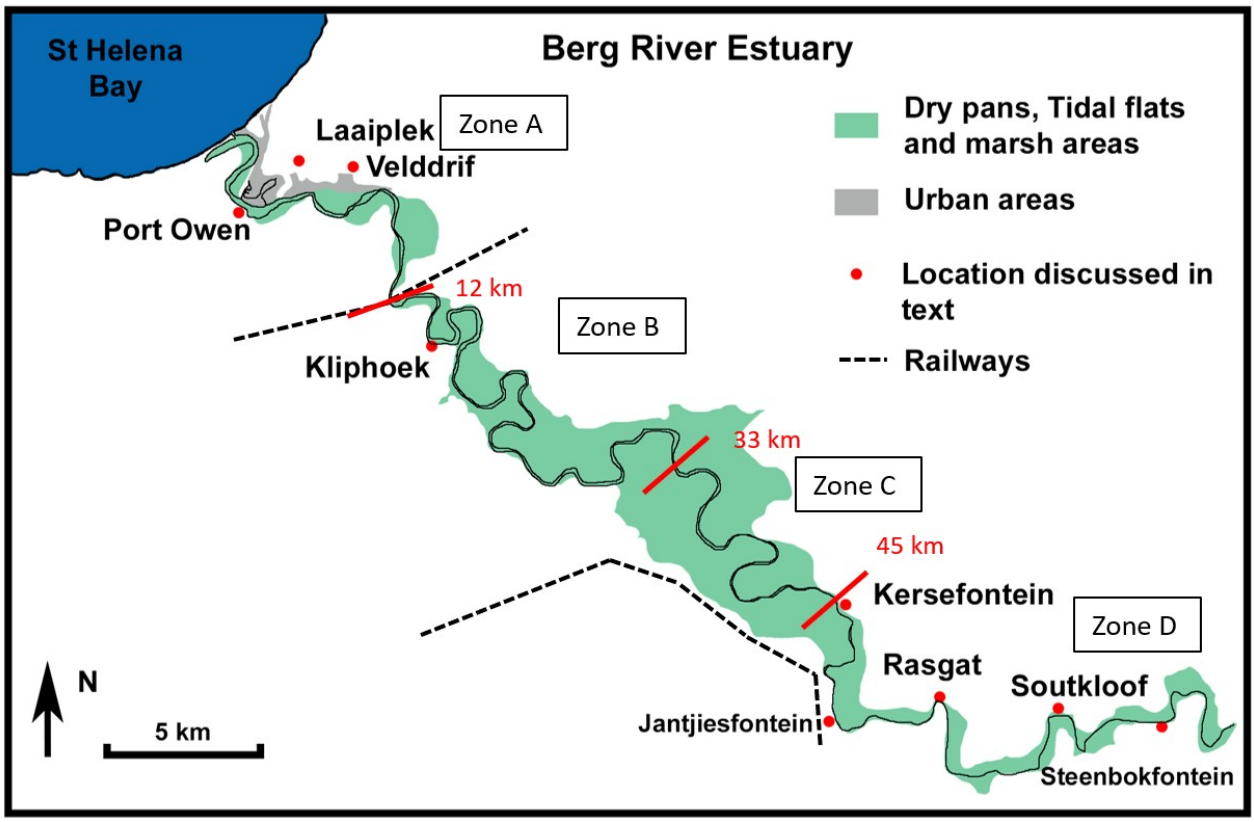

Figure 5.22. Different abiotic zones identified for the Great Berg River Estuary (map adapted from Schumann, 2007).

Table 5.6. Calculated Average salinity within each zone under different scenarios (method of calculation given above).

	Ref	2010	P0 2020	P1	F0	F1	C0	C1
				PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
Zone A	18	24	24	23	25	24	30	30
Zone B	5	12	17	15	18	16	25	23
Zone C	0	5	7	5	7	5	11	9
Zone D	0	1	1	0	1	0	2	1

The average levels (classified into groups: high, medium or low) for water quality parameters were calculated for each abiotic state within each zone and then used in conjunction with the percentage occurrence of the respective abiotic state under various scenarios to calculate a score for nutrient concentration (2a in Table 5.7). Recalling from Section 0 that elevated levels of dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphate (DIP) are introduced with freshwater entering the system, and that levels of these nutrients have increased within the Berg River in the past 10 years (Cullis *et al.* 2019), it is not surprising that the present day (2020) score for water quality has dropped relative to 2010. The scores under future development and climate change are expected to drop slightly relative to present day, as although the nutrient levels within the Berg River may increase, the volumes of nutrient-rich water entering the estuary is expected to decrease. Similarly, water clarity (2b in Table 5.7) has dropped under present day conditions relative to 2010, due to increased algal blooms. The dissolved oxygen is a function of the salinity, temperature and nutrient concentrations within the estuary, with warmer, more saline water holding less oxygen than cooler, fresher water. In addition, increased nutrient levels often result in decreased oxygen levels as algal blooms deplete oxygen concentrations. Levels of trace metals are known to be elevated in the estuary (Hutchings & Clark 2010, CSIR 2016), and are likely to increase in future as a result of increasing development in the catchment. Therefore, scores are lower under the future development and climate change scenarios.

Overall, the water quality health score for the Berg River Estuary has decreased from 40.2% in the 2010 RDM to 31.1% for present day (2020). Water quality health scores for scenario 1-3 do not differ greatly from present as flows will not be dramatically altered in these scenarios, however as above, scenarios in which EWRs are included show slightly better scores than those without EWRs Under the future development scenario (Sc 2) the water quality health score drops further to 28.7% but is slightly better when low flow EWRs are supplied (score increases to 30.8%). Scores drop still further when the effects of climate change are included (21.5 and 22.1% for Scenarios 8 and 9, respectively).

It is important to note that under present conditions the average high flow values for dissolved inorganic nitrogen, average high and low flow values for inorganic phosphates and periodically, the E.coli count within the lower reaches of the estuary all exceed the gazetted RQOs. This raises the concern that the occurrence of these exceedances and/or the extent to which the RQOs are exceeded may increase under future scenarios, especially under Climate change scenarios 8 and 9.

Table 5.7. Expected changes in water quality health scores in the Berg River Estuary under the various future flow scenarios. *2010 health scores based on 2010 RDM study.

	Weight	2010*	P0 2020	P1	F0	F1	C0	C1
				PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
1. Salinity	40	63.0	50.8	55.9	48.9	53.0	31.6	33.0
2 - General water quality								
a. N and P concentrations		25.0	18.0	16.9	15.3	16.0	14.8	14.9
b. Water clarity (suspended solids/ turbidity/transparency)		85.0	65.1	63.4	64.9	62.7	57.5	56.1
c. Dissolved oxygen (mg/l) concentrations		85.0	77.3	79.1	72.1	74.0	62.7	64.4
d. Toxic substances		80.0	80.0	80.0	70.0	70.0	60.0	60.0
General water quality = min (a to d)	60	25.0	18.0	16.9	15.3	16.0	14.8	14.9
Water quality health score = weighted mean (1,2)		40.2	31.1	32.5	28.7	30.8	21.5	22.1

6 BIODIVERSITY

6.1 Introduction

This chapter provides a description of the various biotic groups of the Berg River Estuary, with emphasis on fish and birds, for which more recent data have been added to the information available in the literature. These groups include:

- **Microalgae** – single-celled algae which are found in the water column (phytoplankton), as well as settled on bottom sediments and plants;
- **Macrophytes** – the plants growing in and alongside the estuary channel, and in the floodplain, including the extensive salt marshes of the system;
- **Invertebrates** – found burrowing in the mud and swimming in the water column, and ranging in size from tiny zooplankton to large crabs;
- **Fish** – including those that breed in the estuary and those that range between the estuary and marine or freshwater environments;
- **Birds** – including those that are resident, those that move between the estuary and inland habitats, and those that migrate long distances to spend the summer.

The focus is on improved understanding of how these biotic communities respond to physical drivers related to changes in freshwater flows and predicting biotic responses under future, potential freshwater flow regimes in the Berg River Estuary relative to Reference, or Natural conditions (i.e. prior to any anthropogenic impacts). Data from earlier studies are reviewed to infer how biota may have changed from the earlier surveys, particularly in response to the severe drought experienced over the 2015-2018 period. Responses of the higher trophic level groups, namely fish and birds, to the recent drought, are investigated through analysis of recent data. The findings are considered in conjunction with those of the abiotic elements in the overall assessment of estuary health and implications of the scenarios in the final synthesis chapter.

We then provide an assessment of the current status (Present Day 2020) of each taxonomic group, which is compared with the findings of the Resource Directed Measures (RDM) study in 2010. Following this, the impacts of the hydrological scenarios described above are also assessed for each of the taxonomic groups. All the scores are relative to a natural "Reference" condition.

6.2 Microalgae

Microalgae are generally defined as unicellular algae that either live suspended in the water column (phytoplankton), on rock or sediments in the estuary (benthic microalgae) or on plants (epiphytic microalgae). Estuarine phytoplankton communities are typically comprised of flagellates and diatoms, while benthic microalgae communities are dominated by blue-green algae and diatoms. Microalgal productivity is the most important determinant of overall biomass of estuarine biota, with most trophic pathways originating in microalgal rather than macrophyte (plant) productivity (DWAF 2007, DWA 2010). As such, microalgae are an important component of the estuary biota.

The microalgae community composition of the Berg River Estuary is typical of South African estuaries and reflects the physical conditions of the estuary (DWAF 2007). The main physical drivers that influence phytoplankton communities in the system include salinity, water residence time and nutrients.

The biomass of phytoplankton in South African estuarine systems varies widely, from 0-210 µg Chl *a*/l (measured as Chlorophyll *a* concentration) and is strongly linked to nutrient concentrations (Adams *et al.* 1999). If nutrient concentrations in an estuary are high, particularly in the case of nitrogen, then phytoplankton biomass in the estuary is generally high too. Under extreme conditions, when nutrient levels are very high, certain toxic dinoflagellate species may form dense blooms known as red tides (DEA&DP 2019). Nutrient concentrations in the estuary may also not be distributed equally, and therefore may affect where certain microalgae communities occur. For example, while nitrogen concentrations in the Berg River Estuary are low overall under low flow conditions, higher concentrations occur in the lower reaches of the estuary and decrease upstream. Because microalgae biomass is linked to nutrients, this means that a higher microalgae biomass would occur in the lower reaches during low flow periods. In contrast, during high flow periods (i.e. winter), nitrogen concentrations in the estuary are higher overall than during low flow, and highest in the upper reaches of the system, which means that microalgae abundance would be higher in the upper reaches during high flow periods (Bate & Snow in DWA 2007). The upper, middle and lower reaches of the Berg River Estuary are shown in in Figure 5.22 above, and follow the divisions of typical salinity profiles and bathymetry outlined by Taljaard *et al.* in DWA (2010).

Salinity also influences phytoplankton community structure and biomass. Upstream, where river flow dominates and the water is fresh, the phytoplankton community generally consists of flagellates. In contrast, towards the mouth, where marine conditions dominate, the community is mostly made up of diatoms (DEA&DP 2019). It is at the interface of fresh and marine waters, where the salinity is in the region of 10-15, often referred to as the River-Estuary Interface zone, that overall phytoplankton biomass tends to be highest (Snow 2000, DEA&DP 2019). During low flow periods, such as during peak summer or in a drought, this River-Estuary Interface occurs some 40 km upstream of the mouth, depending on the tide, compared to about 15 km upstream during periods of high fluvial input. During winter 2005, phytoplankton chlorophyll a in the Berg River Estuary was highest in the lower reaches, while in summer the highest phytoplankton chlorophyll a was in the upper reaches (Figure 6.1, Snow 2010). Both these high chlorophyll *a* concentrations coincide with low nutrient concentrations, a distribution pattern associated with significant biological uptake i.e. phytoplankton blooms.

While low flow is unlikely to lead to a change in the phytoplankton biomass peak, the resultant change in the location of the river-estuary interface during low fluvial input would influence where the highest phytoplankton biomass occurs in the estuary, with repercussions for the distribution and abundance of higher trophic level biota (Snow in DWA 2010). As phytoplankton form the basis of the food chain in estuarine systems, and play a crucial role in the production of oxygen and cycling of nutrients, changes in where the highest phytoplankton biomass occurs will affect the other biota that depend on them. For example, microalgae are food for various invertebrate and fish species, and shifts in the spatial distribution of the microalgae biomass would also result in changes in the spatial distribution of these invertebrate and fish species as they "follow the food". A reduction in flow may also lead to hypoxic conditions (low water oxygen concentrations) in certain areas of the estuary, potentially releasing nutrients (such as phosphates and ammonia) from the sediment though

anaerobic bacterial processes and resulting in an increased occurrence of phytoplankton blooms (Snow in DWA 2010).

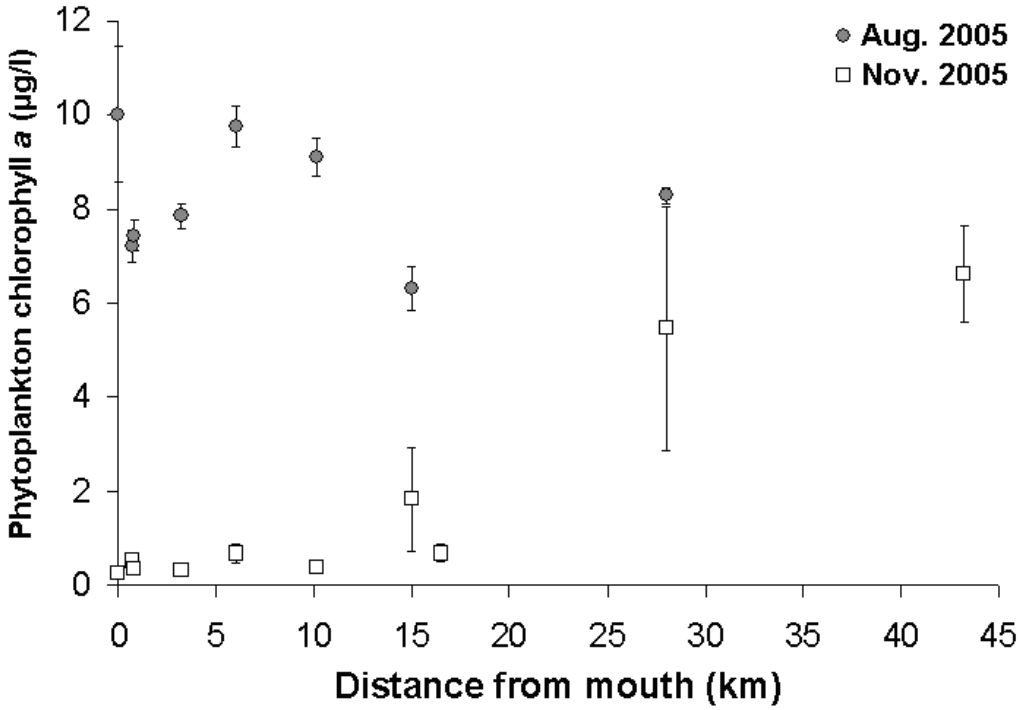

Figure 6.1. Vertically averaged water column chlorophyll a measured along the longitudinal axis in the Berg River Estuary in August and November 2005. Vertical bars represent standard error of the means. Source: Snow (2010).

Water residence time, the time that water remains in the system before being flushed out, also positively influences phytoplankton biomass (Hilmer & Bate 1990). Despite residence time and temperatures being highest in the summer dry season, microalgal abundance in the Berg River Estuary is higher in winter (Slinger & Taljaard 1994, Snow & Bate in DWA 2007), probably due to higher nutrient input during the winter (De Villiers 2007, Cullis *et al.* 2019). Nevertheless, it has been hypothesised that the summer populations make a greater contribution to estuary productivity, being more available to higher trophic levels while in slower-moving water. Most of the winter production is likely exported to St Helena Bay.

There was a dramatic increase in phytoplankton biomass in the Berg River Estuary between 1989 and 2005, with the average biomass ofc phytoplankton increasing from 1.8 to 8.2 µg/l in winter and from 0.2 to 1.2 µg/l in summer (Figure 6.2). This increase may have been in response to the corresponding increases in concentrations of inorganic nutrients in the estuary. This increase is presumably linked to increased agricultural activities and human settlements in the catchment (Cullis *et al.* 2019). Recent, 2015-2019 nutrient data (BRIP – water quality data 2013 to present) indicate that there has been an up to two-fold increase in nitrogen concentrations in river inflow in the high flow season. This suggests that phytoplankton biomass could have increased even more in the last 15 years (if nitrogen availability was the sole limiting factor, and not e.g. light penetration, day length or other requirements for algal growth). This moves

the health of the estuary in terms of microalgae abundance further from the Reference state because of the high level of anthropogenic nutrient input.

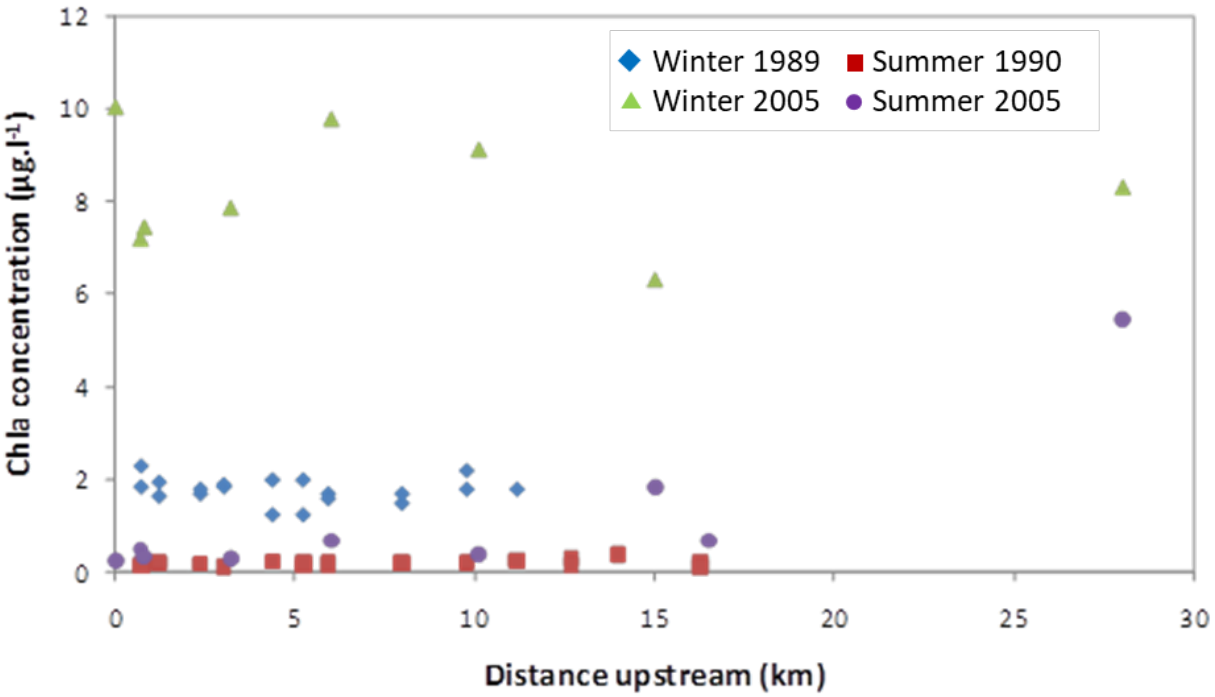

Figure 6.2. Phytoplankton biomass (measured as Chl *a* concentration) in the Berg River Estuary in 1989/1990 and 2005. Data from Slinger & Taljaard 1994 and Snow & Bate in DWA 2007.

Less is known about benthic microalgae communities in estuaries than phytoplankton. However, available data from five different estuaries in South Africa report benthic microalgae biomass ranges from 9.5-21.8 µg Chl *a*/g dry mass of sediment (microalgae that lives on sediments is measured as the Chl a present in one gram of the collected sediment after the sample has been dried; Bate in DWAF 2006). Benthic microalgae community structure is strongly influenced by salinity. In the lower reaches, high salinity and a stable water column support more marine adapted species, while the upper reaches support species better adapted to fine sediment, higher organic content and lower salinity (Snow in DWA 2010).

Benthic microalgae biomass is also reduced by turbulence. Thus with low inflows, there may be higher benthic microalgae biomass (by up to 2 mg/m^2 Chl *a*) in the upper reaches of the estuary where water is relatively calm (Snow 2010). In contrast, tidal currents limit benthic microalgal biomass in the lower reaches of the estuary (Snow in DWA 2010).

In general, during high flow periods when nutrients are plentiful, micro-algal production is limited by turbulence and turbidity, whereas during low flow periods, when water is clear, but nutrients could become limiting (DWAF 2007). This potential nutrient limitation may be countered by the high levels of anthropogenic nutrient input into the system. Peak phytoplankton biomass levels follow the interface between the marine and freshwater systems,

which during low flow periods like a drought, move further upstream from mouth and are likely to be dominated by marine and diatom species.

A dominance of one phytoplankton species over another has potential effects on biogeochemical cycling of key elements such as C, N, and P, and implications for higher trophic levels. Diatoms and flagellates have very different sedimentation patterns after they bloom — while the heavy silica covered diatoms sink quickly out of the euphotic zone, flagellates sink as slowly settling phyto-detritus/ "marine snow" (Spilling *et al.* 2018). The dominance by either phytoplankton group thus directly affects both the summertime nutrient pools of the water column and the input of organic matter to the sediment (Spilling *et al.* 2018).

By 2010, microalgae species richness in the Berg River Estuary were not expected to have changed relative to the Reference state, because the full salinity gradient within the estuary still existed (Table 6.1). Phytoplankton biomass was estimated to have increased by 20% because of anthropogenic nutrient inputs. Benthic microalgal abundance was estimated to have decreased because of a decline in available habitat. The phytoplankton community structure was thought to have changed from being dominated by flagellates to diatoms due to reduced freshwater inputs. The benthic microalgal community composition would have changed from epipelic to episammic diatom taxa because of shifts in the sand and mud content of the system.

Under Present Day (2020) conditions, species richness is expected to have remained the same. However, abundance/biomass of both phytoplankton and benthic microalgae will have increased. The health of the microalgae assemblages has declined markedly since 2010 (down from 75 to 68.3%; Table 6.1) and is linked to increased nutrient levels and salinity.

Projected changes in species richness, biomass and community composition of phytoplankton and benthic microalgae under the various future flow scenarios are summarised in Table 6.1. Again, the microalgae health score increases slightly when summer low flow EWRs are supplied (increases from 62.7 to 64.2% from F0 to F1), but as discussed above, these EWRs are not sufficient to reverse the microalgal changes that have occurred due to historical changes in the estuary structure and function, instead preventing further changes from occurring. Microalgal health drops still further under the future development scenario (62.7%) and again when the effects of climate change are considered (44.5%), with minimal change depending on whether summer low flow EWRs are supplied or not (score changes by no more than 1.7%).

Table 6.1. EHI scores for species richness, biomass and community composition of phytoplankton and benthic microalgae under the various future flow scenarios (Hist. = Historical, P0 = Status Quo, EWR = Environmental Water Requirements , FD = Future Development, CC = Climate Change).

	Hist. 2010	P0 2020	P1	F0	F1	C0	C1
			PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
Species richness	100.0	100.0	100.0	100.0	100.0	100.0	100.0
Abundance/ Biomass	81.0	68.3	67.5	62.7	64.2	51.5	51.6
Community composition	75.0	68.9	68.8	66.5	66.3	44.5	42.9
Health score	75.0	68.3	67.5	62.7	64.2	44.5	42.9

6.3 Plant communities

The Berg River Estuary has by far the most extensive and diverse vegetation of all permanently open estuaries in South Africa (DEA&DP 2019a). The species richness, growth, cover, distribution and community composition of these habitats is linked to tidal and riverine flooding and salinity regimes.

The main plant communities associated with the Berg River Estuary are macroalgae, submerged macrophytes, intertidal and supratidal salt marsh, reeds and sedges (Table 6.2 and Figure 6.3). These are largely distinguishable as subtidal, intertidal and floodplain communities, and include both supratidal and emergent vegetation, as well as intertidal and submerged vegetation.

Macroalgae are generally concentrated in the lower reaches of the estuary, forming mats that cover the intertidal sand and mudflats (DEA&DP 2019a). These algal mats comprise the indigenous *Enteromorpha prolifera* and the European invasive *E. flexuosa* (Clark *et al.* 2007; DEA&DP 2019a). There is some concern that their abundance is increasing, and that they are smothering the mudflats and reducing the availability of invertebrate prey for birds (DEA&DP 2019a).

Subtidal vegetation in the lower reaches is dominated by eelgrass *Zostera capensis*. This is replaced upstream by pond weed *Potamogeton pectinatus* in the fresher upper reaches (DEA&DP 2019a). Both eelgrass and pond weed also extend into intertidal areas, collapsing into dense mats at low tide. Structurally and functionally similar, these communities are separated by their salinity requirements (DEA&DP 2019a). The extent of these two communities is strongly linked to flow, in that they readily replace one another when salinity distribution changes for any significant period (DEA&DP 2019a).

In the lower estuary, intertidal eelgrass competes for space on mudflats with the various filamentous algae, such as *Cladophora* (DEA&DP 2019a). The presence of *Cladophora* is highly seasonal, proliferating in spring and drying up over summer before being washed out of the estuary in winter (Kaletja & Hockey 1991, Clark *et al.* 2009).

As one moves out of the channel and more regularly-submerged intertidal areas, eelgrass gives way to low-growing intertidal salt marsh. Salt marsh composition is strongly zoned by the degree of tidal inundation, with characteristic species progressing from cord grass *Spartina maritima* at lower elevations to soutbos *Bassia diffusa*, daisies *Cotula* spp and brakbos *Sarcocornia perennis* at higher elevations (DEA&DP 2019a). Above the intertidal area of the lower estuary, the intertidal salt marsh gives way to supratidal salt marsh, dominated by brakbos and interspersed with bare patches. These areas are typically flooded in winter. The area of intertidal and supratidal salt marsh on the Berg River Estuary is larger than that on any other estuary in the country (Adams *et al.* 2019).

Table 6.2. Summary of Berg River Estuary riparian vegetation, salinity and flooding requirements as well asc area coverage (within the 5 m contour). Source: Boucher & Jones (2007) and Anchor (2008).

Vegetation type	Habitat type	Dominant species	Optimal salinity	Cover (ha)	
				Boucher & Jones (2007) and Anchor (2008)	Adams *et al.* (2016)
Macroalgae	Intertidal sand and mudflats	Green filamentous algae *Enteromorpha prolifera* and *E. flexuosa*, brown filamentous algae *Ectocarpus siliculosus* and red algae *Caloglossa leprieuri*	35	~ 200.0	
Submerged macrophytes	Intertidal sand and mudflats	Eelgrass *Zostera capensis*, spiral ditchgrass *Ruppia cirrhosa* and pond weed *Potamogeton pectinatus*	35	206.0	206
Intertidal salt marsh	Halophytic (saline tolerant) salt marsh	Brakbos *Sarcocornia perennis*, cord grass *Spartina maritima*, streaked arrow grass *Triglochin striata*, sedge *Salicornia meyeriana*, soutbos *Bassia diffusa*, daisies *Cotula coronopifolia* and swamp grass/kuilgras *Leptochloa fusca*	35	123.9	1667
	Sedge marsh	Salt marsh rush *Juncus kraussii*	25	375.0	
	Open pan	Streaked arrow grass *Triglochin striata* and sedge *Salicornia meyeriana*	<10 (winter); 45 (summer)	1158.6	
Supratidal salt marsh	Halophytic floodplain	Brakbos *Sarcocornia pillansii* and *S. perennis*	45	1520.7	2545
	Xeric (dry) floodplain	Grysbietou *Osteospermum incanum*	45	919.1	
Reeds and sedges	Normal tall reed marsh	Common reed *Phragmites australis*	15	513.5	4588
	Short reed marsh	Invasive sage *Schoenoplectus triqueter*, papgras *S. scirpoides* and majiesgoed *Cyperus textilis*	15	73.1	
	Sedge pan	Sareegras *Juncus maritimus* and waterblommetjie *Aponogeton distachyos*	<5	975.1	

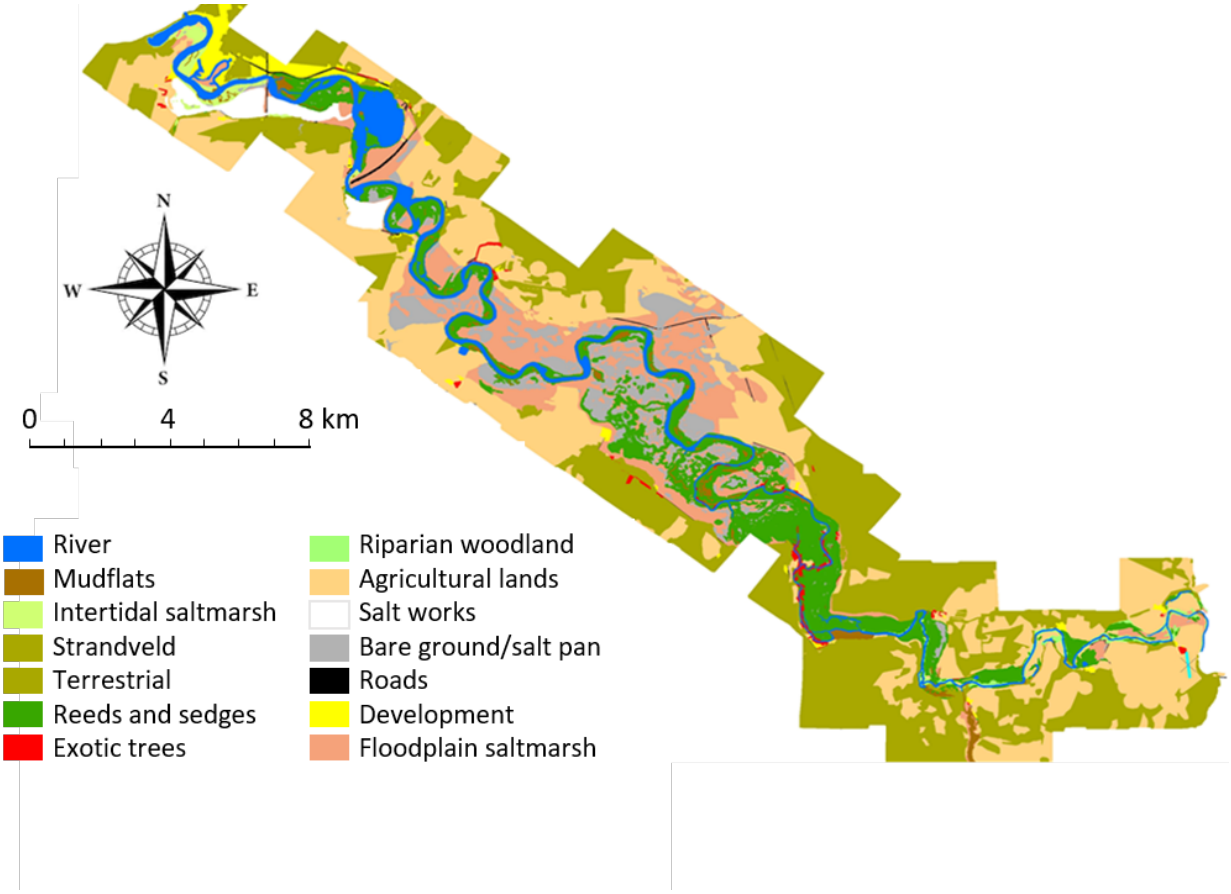

Figure 6.3. Distribution of plant communities along the length of the Berg River Estuary. Source: Anchor 2008.

In the fresher reaches further upstream, the narrow intertidal and adjoining floodplain areas are mainly occupied by sedge marsh, dominated by *Schoenoplectus* spp. and *Cyperus textilis* or by taller reed marsh, mainly monospecific stands of reeds *Phragmites australis*. Reed marsh tends to replace sedge marsh on the silt-rich soils which are deposited on inner river bends. Behind the reed marsh, "many of the inner river bends also contain extensive lower-lying sedge marshes that are flooded during winter, creating sheltered water bodies" (DEA&DP 2019a).

The floodplain also contains numerous pans. These are either open pans, with scattered saltpan sedges such as *Salicornia meyeriana*, or sedge pans characterised by monospecific stands of sareegras *Juncus maritimus* and waterblommetjie *Aponogeton distachyos* (DEA&DP 2019a). The latter are usually at a lower elevation and tend to be linked to the river channel by drainage lines. At higher elevations, the floodplain is occupied by a xeric ('dry') floodplain community which contains elements of terrestrial strandveld and is dominated by succulent families, such as *Aizoaceae* and *Asparagaceae*, as well as other drought-adapted species. This community depends on annual floods to reduce salinity levels and replenish soils. In the uppermost reaches of the estuary, the riverbanks are lined by riparian woodland, with species such as *Salix mucronata*, *Searsia tomentosa*, *Olea* spp. and *Metrosideros angustifolia*. Alien *Eucalyptus, Acacia* and poplar *Populus* trees are also common (DEA&DP 2019a).

These alien trees have may affect water quality in the system. In the past, the *Eucalyptus* trees have been implicated in contributing to the development of low oxygen water near the top of the estuary which occurs during periods of very low flows. While most of the trees have been

removed, current water quality impacts could be potentially ascribed to the abundance of rotting vegetation left in the river from the clearing of these alien trees (Schumann 2007). In addition, such alien trees may also impact the system by further reducing water levels (Schumann 2007).

The vegetation of the Berg River Estuary has been shaped by a strongly seasonal flow and flood regime which maintains the floodplain systems, and will therefore be affected by reduction in flows and floods. Under summer low flow conditions, the salt water moves further upstream before being flushed out in winter. However, under extended low flow conditions, when higher salinities persist, there is dieback of reeds and sedges that grow best in a salinity of less than 15 (DWA 2010). Under drought conditions, freshwater brackish wetlands change to halophytic ("salt marsh") floodplain vegetation, and sedge pans to open saline pans. At higher elevations, the xeric vegetation is well adapted to a lower frequency of inundation, but even this vegetation will suffer dieback if it does not experience enough inundation.

The Western Cape drought of 2015-2018 provided an opportunity to investigate the effects of extended freshwater starvation, when very little flow and no floods were experienced in the estuary. We analysed remote sensing (satellite) enhanced vegetation index (EVI) data on vegetation productivity for the period 2000-2020. The EVI provides an indication of vegetation cover. We used data for 2000 to 2011 to define the "baseline" state (or mean EVI) against which to compare the EVI of the drought years. The "baseline" period included both drier than average years (2003-2004) and wetter than average years (2007-2009). The EVI data showed a clear decrease in vegetation during the 2015-2018 drought (Figure 6.4). EVI levels were at their lowest just after the drought in winter 2019, and remained low at the beginning of 2020 (Figure 6.4). Interestingly, the highest EVI value during 2018 was well below the lowest EVI value during 2011-2017 (Figure 6.4).

Vegetation dieback occurred predominately in the floodplains (Figure 6.5). There was also a decrease in EVI along some parts of the main channel, indicating dieback of fringing reeds and sedges due to changes in salinity (Figure 6.5). This dieback was clearly visible in the field, where reeds were starting to regrow by early 2020. Areas of "increase in EVI vegetation cover" appear to represent an increase in halophytic, floodplain saltmarsh vegetation, or increased vegetation on bare ground or saltpans (Figure 6.5).

Other changes in community structure due to extensive periods of low flow and increased nutrients could include the potential growth of the highly invasive water hyacinth *Eichhornia crassipes* which may displace pond weed in the upper reaches of the estuary (DWA 2010). Under extended low flow conditions, macroalgae, particularly the filamentous species, may also form extensive mats in the lower reaches that displace eelgrass beds. It is unfortunately not possible to ascertain from satellite data if these predicted changes in aquatic vegetation did take place during the 2015-2018 drought (DWA 2010).

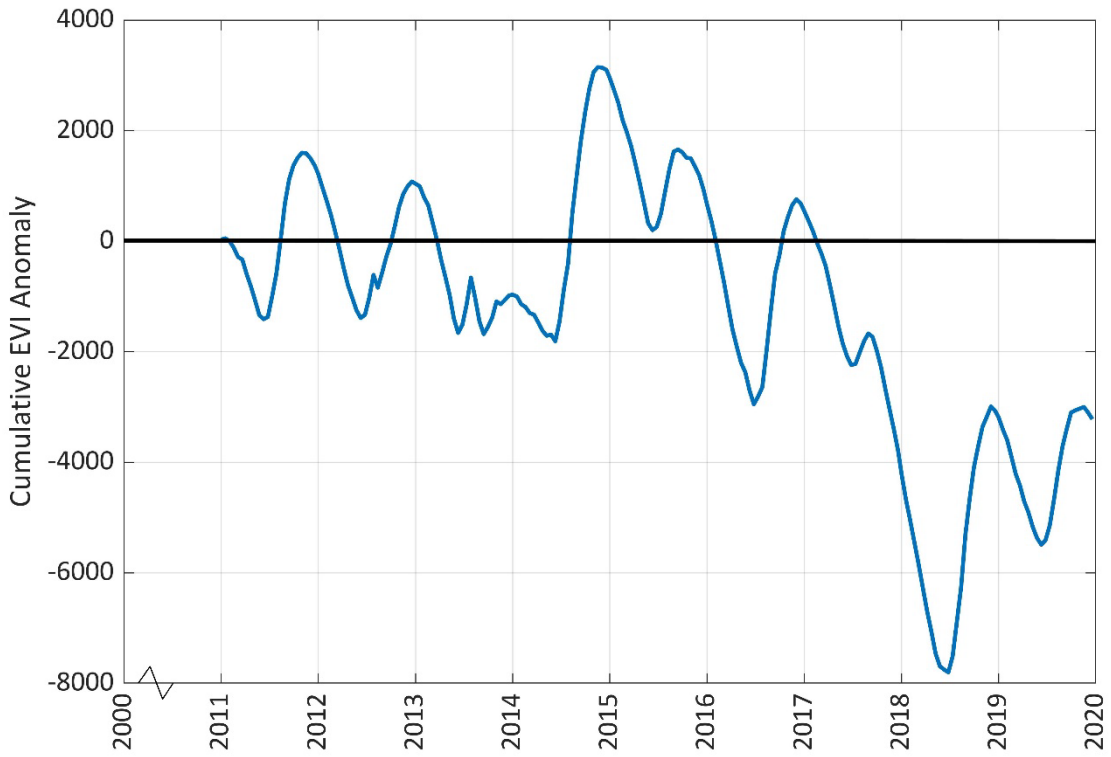

Figure 6.4. Cumulative Enhanced vegetation index (EVI) anomaly for the Berg River Estuary from 2011 to 2020. The 2001-2011 conditions were used to create a baseline mean against which the latter years were compared. With respect to the baseline period; negative values indicate a decrease in canopy structure, while a positive number indicates an increase in canopy structure. By using 10 years as a proxy to establish the baseline EVI, we can decouple annual, intra-annual, and decadal variability in EVI and observe a minimum that occurred during the 2017-2019 Western Cape drought.

The impacts of a reduction in quantity and quality of freshwater inflows are compounded by the historical and current anthropogenic impacts on the vegetation and habitats of the estuary and floodplain. Some 26% of total estuarine area has been lost due to agricultural, urban and other activities (DEA&DP 2019a). These impacts include total transformation of the habitat (with near complete loss of vegetation) such as found at Cerebos salt works and Port Owen, farming activities, including grazing and cropping, which have reduced vegetation cover to less than 5% in some areas, and associated erosion of the riverbanks (Boucher & Jones in DWA 2007). Approximately 40% of the original estuarine vegetation has been lost, including intertidal mudflats (5% lost), open pans (73% lost) and other vegetated areas (approximately 50% lost) (DWA 2010).

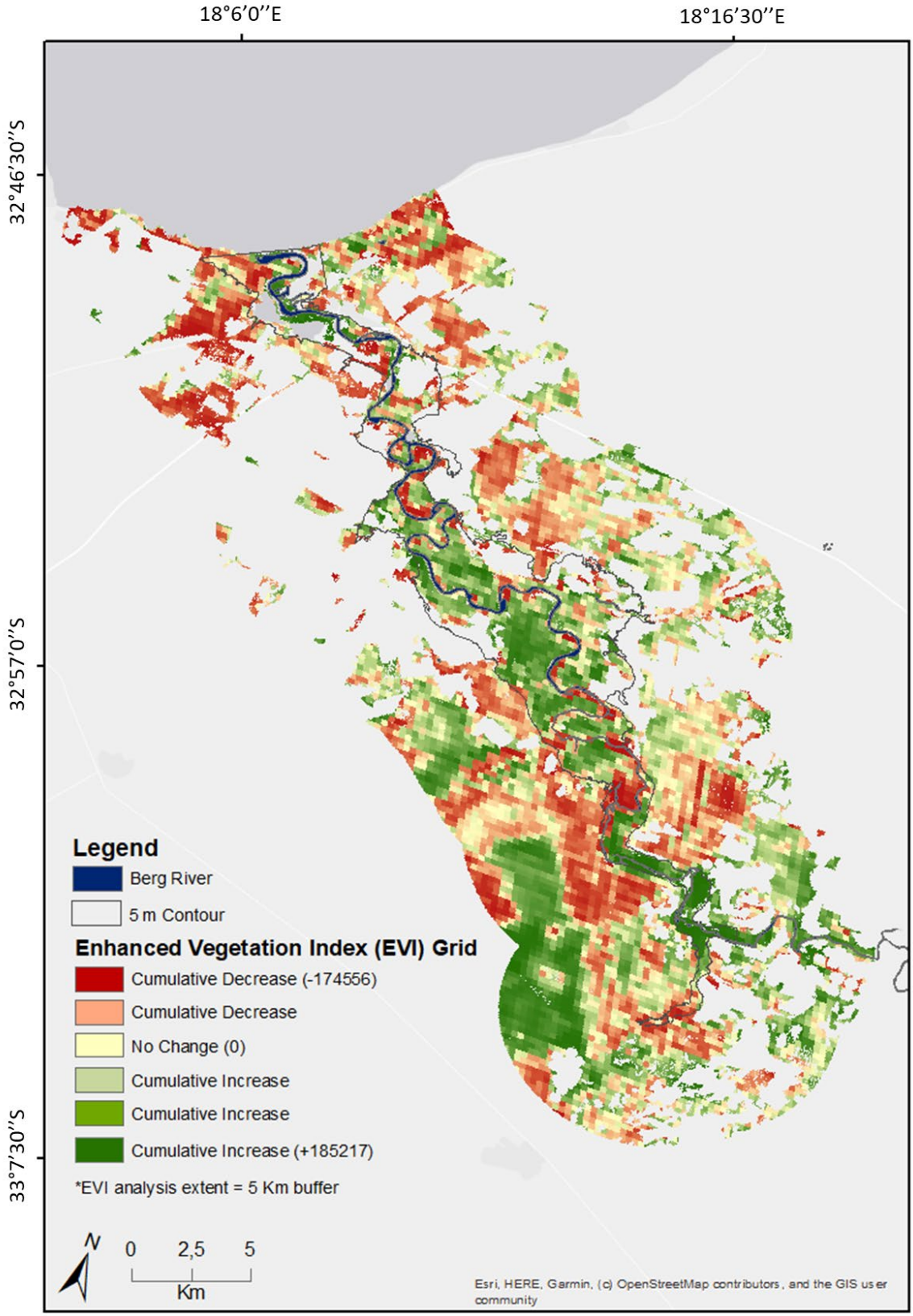

Figure 6.5. Cumulative Enhanced vegetation index (EVI) anomaly for the Berg River Estuary from 2011 to 2020. The red colours indicate a cumulative decrease in vegetation over time, while white colours indicate no change and green colours indicate a cumulative increase in vegetation over time. The 5m contour is shown.

There has been a marked decline in the health of the vegetation community of the Berg River Estuary from 54% similarity to Reference condition in 2010 to 43% in 2020, mostly linked to changes in salinity and nutrient levels in the estuary (Table 6.3). In both the 2010 and Present Day (2020) assessment, low flow conditions and reduced flooding compared to Reference have influenced species richness, biomass and community composition. There has been a loss of brackish sedge pans *Juncus maritimus,* and waterblommetjies *Aponogeton distachyos* because of increases in salinity and reduced flooding compared to Reference conditions.

Projected changes in species richness, biomass and community composition of phytoplankton and benthic microalgae under the various future flow scenarios are summarised in Table 6.3. Given the expectation of a progressively greater frequency of extreme drought and reduced flooding under Scenarios F0, F1, C0 and C1, there would be increased dieback of floodplain habitat, with open pans, halophytic and xeric floodplains becoming drier with less biomass and vegetation cover. Some plant communities that thrive at intermediate salinities could still do well under particularly F1 and C1, when dry season minimum flows are met. Overall, these scenarios could result in a change in community composition from freshwater brackish wetlands to halophytic floodplain and saltmarsh, and from sedge pans to open saline pans. Community composition may also change in response to increases in nutrients and salinity. Health of the vegetation community is expected to decline even further under the future development scenario (down to 42.0%) and further still if climate change projections are factored in (down to 34.9%). Respecting the summer low flow EWRs does little to alleviate this trend (score change by no more than 1%). In fact, due to increased nutrients, the EWR in Scenario P1 results in a decline over the present situation (P0 with no EWR) because of the increased nutrient runoff and decline in water clarity.

Table 6.3. Summary of projected changes in species richness, biomass and community composition of riparian vegetation under the various future flow scenarios (Hist. = Historical, P0 = Status Quo, EWR = Environmental Water Requirements , FD = Future Development, CC = Climate Change).

	Hist. 2010	PD 2020	P1	F0	F1	C0	C1
			PD+EWR	FD	FD+EWR	FD+CC	FD+CC+ Low EWR
Species richness	80.0	71.8	72.1	68.7	68.8	48.2	47.4
Abundance/ Biomass	54.0	45.7	45.2	42.0	43.0	34.9	35.0
Community composition	60.0	53.8	54.1	51.5	51.6	36.2	35.5
Health score	54.0	45.7	45.2	42.0	43.0	34.9	35.0

6.4 Invertebrates

The invertebrate fauna in the Berg River Estuary has amongst the highest biomass of South African estuaries (DWA 2010), both in terms of density and overall biomass. These invertebrates provide rich foraging opportunities for fish and birds within the water column as well as in the subtidal and intertidal mudflat areas.

Estuarine invertebrates can grouped based on where they reside. Zooplankton live mostly in the water column, benthic organisms live on and in the sediments on the bottom and sides of the estuary channel, and hyperbenthic organisms live just above the sediment surface. Benthic organisms can be further subdivided into intertidal and subtidal communities. These communities are briefly described below:

- The zooplankton of the Berg River Estuary are numerically dominated by copepods (~98%), the remainder being made up of fish larvae, brachyuran larvae, mysid shrimps, amphipods and other organisms (Wooldridge 2007). *Pseudodiaptomus hessei* is the most abundant zooplankton species and is a major component in the diet of zooplanktivorous fish. Its distribution follows the typical pattern found in freshwater rich estuaries, being most abundant in the upper-middle reaches. The species has a wide salinity tolerance and does not show any correlation with salinity patterns. However, it tends to be flushed out of the estuary during high winter flows (DEA&DP 2019).
- The hyperbenthos is dominated by larvae and post larvae of the crab *Hymenosoma orbiculare*, mysid shrimps and fish larvae. Mysid shrimps (mainly *Mesopodopsis wooldridgei*) and amphipods (mainly *Corophium triaenonyx*) dominate the hyperbenthos, especially in winter (DEA&DP 2019).
- The subtidal benthos is numerically dominated by amphipods (mainly *Grandidierella lutosa* and *Corophium triaenonyx*) and polychaetes (mainly *Capitella capitata*) in both summer and winter. Amphipods make up half to three quarters of the overall numbers of benthic invertebrates, with polychaetes dominating the remainder. Polychaetes tend to be more abundant in the lower half of the estuary, but both polychaetes and amphipods are most abundant in the middle reaches. Mudprawns *Upogebia africana* are also abundant in the lower reaches and make up a significant proportion of subtidal invertebrate biomass (Anchor 2010, DEA&DP 2019). Sand prawns *Callianassa kraussi* are re-establishing in the blind arm and are also now further than 12 km from the mouth, probably because salinity rose above 16 for much of the drought allowing them to reproduce. These crustaceans may be responsible for the less hypoxic sediment present within the blind arm than in the past. than previously.
- The intertidal benthos is numericaly dominated by polychaetes (82%) (mainly *Ceratonereis erythraensis*), amphipods (11%) and isopods (4%). Density of intertidal invertebrates is highest in the lower estuary (Anchor 2010, DEA&DP 2019). Mud and sand prawns are also abundant in intertidal areas.

While invertebrate distribution patterns are shaped by many physical characteristics, such as sediment grain size, temperature, chlorophyll *a* and nutrients, salinity is the most influential (Wooldridge & Deyzel in DWA 2010). There are generally fewer species in estuaries than marine ecosystems because few organisms can tolerate the full estuarine salinity range. The organisms that are able to survive in estuarine systems are broadly divided into the following groups:

- Marine species (dominant in terms of number of species), which include two subgroups:

- Stenohaline animals with low or no tolerance of salinity changes, which tend to be restricted to lower estuary where salinity is close to that of seawater; and
- Euryhaline animals, which can tolerate varying degrees of salinity reduction below 30 (down to 15) and can penetrate some distance up the estuary. Some very hardy species can tolerate water that is almost fresh (down to a salinity of 3).

- Brackish water or true estuarine species (often endemic), that are found in the middle reaches of estuaries where salinity varies between 5 and 20.
- Freshwater species, usually restricted to a zone where salinity does not exceed 5 in the upper reaches.

Due to their relatively short life cycles and rapid responses to changes in their environment, estuarine invertebrate assemblages are known to shift up or down an estuary in response to shifts in environmental conditions, particularly freshwater inflow (Kalke & Montagna 1991, Attrill *et al.* 1996, MacKay & Cyrus 2001, Rutger & Wing 2006). For example, the amphipod *Grandidierella lutosa* is dominant in the lower estuary and is replaced by the highly abundant *Corophium triaenonyx* in the middle reaches (Wooldridge & Deyzel 2010; Figure 6.6 and Figure 6.7). During high winter flows, the first biomass peak of *C. triaenonyx* occurs closer to the mouth (~12 km) than in summer (~20 km; Figure 6.7). The biomass of the invertebrate community is higher in the summer dry season than in winter (Wooldridge in DWA 2007).

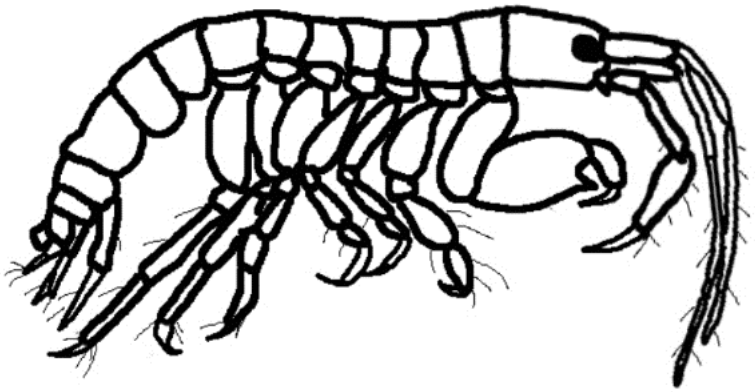

Figure 6.6. A typical amphipod species in the genus *Grandidierella*. Drawing by J Dawson

The number of taxa in the Berg River Estuary increases up to about 10km from the mouth, before decreasing again (Wooldridge & Deyzel in DWA 2010), and is linked to salinity distribution. There is a diverse marine community in the higher salinity parts of the estuary, up to about 5 km from the mouth in winter and up to 20 km in summer. In the upper reaches 30-40 km upstream, where salinity does not exceed 3-5, there is a brackish water assemblage, which has a lower diversity than the marine community (Wooldridge & Deyzel in DWA 2010). In the middle reaches, there is a typical estuarine community which, while the lowest in species diversity, attains a high biomass and as such, forms an important linkage between primary producers and secondary consumers in the water column (Wooldridge & Deyzel in DWA 2010). The link between salinity and invertebrate community structure is particularly strong during the dry summer months, but not during winter (Wooldridge & Deyzel in DWA 2010).

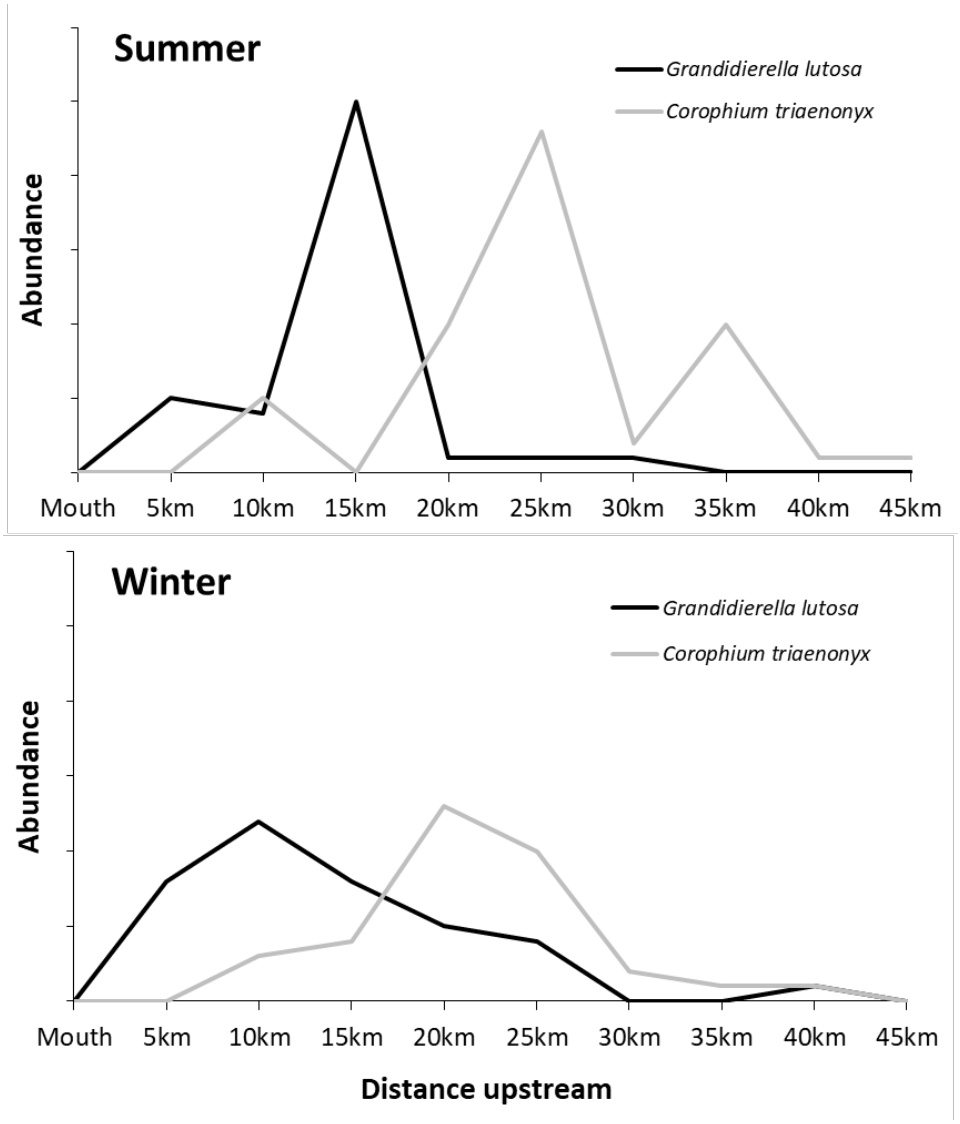

Figure 6.7. Hypothetical distribution of the amphipods *Grandidierella lutosa* (black line) and *Corophium triaenonyx* (grey line) at 15 stations in the Berg River Estuary during the winter (wet) and summer (dry) seasons. Adapted from Wooldridge & Deyzel in DWA 2010.

Sediment particle size is also an important indicator of invertebrate community structure and biomass, influencing both food and oxygen availability, as well as living space for infauna (animals that live on or within the sediment/benthos). For example, finer sediment, like mud, only allows smaller infauna to live between the sediment particles and can only be inhabited by species able to tolerate low oxygen conditions. Fine sediments tend to accumulate in the middle estuary, which may also influence the number of taxa that live there. In this vein, the lower number of species at sites closest to the mouth can be attributed to differences in sediment grain size as well as temperature, as the tidal ebb and flow results in sharp water temperature changes near the mouth, especially during summer upwelling, which cannot be tolerated by many species. The impacts of dredging and sediment disposal on invertebrate species has been considered in various applications for permits to undertake dredging activities in the lower estuary (for the Port Owen marina). Assessments (1993-2006, 2009-2010

and 2016) have shown no discernible decline in invertebrate diversity and abundance from historical baseline levels.

In summary, salinity is the most influential physical parameter for invertebrate community structure in the Berg River Estuary. The invertebrate communities follow the salinity profile of the system: there is a highly diverse marine community in the lower reaches, linked to a salinity of >30; a true estuarine community in the middle reaches, which while lower in species richness is higher in abundance; and a brackish water assemblage in the upper reaches, where salinity does not exceed 3-5. These communities shift with changes in salinity profile of the system. Marine species move further upstream during periods of low freshwater inflow with a parallel upstream shift of the euryhaline community as the brackish zone is pushed back, and vice versa during periods of high fluvial input. Therefore, a change in salinity structure, which is linked to a change in habitat, as well as less frequent flooding will result in a change in the invertebrate community abundance and species richness of the Berg River Estuary. Saline intrusion further up the system for longer periods will result in a marine dominated system, and a subsequent increase in marine species biomass and diversity, especially as the canalised estuary mouth maintains a permanent link to the ocean. In contrast, freshwater-dependent invertebrate communities will retreat upstream, where decreasing habitat due to the narrowing estuary channel will lead to declines in abundance of this component.

By 2010, the invertebrate community had already changed markedly from the Reference condition. As a result of reduced freshwater inflows and increased salinity, zooplankton subtidal invertebrate biomass had already increased by 30-40%, and there would also have been reduced habitat for carid shrimps such as *Palaemon peringueyi* as a result of the retraction of reedbeds from the lower estuary (DWA 2010). Habitat loss has also played an important role, reducing the biomass of intertidal invertebrates and completely changing the invertebrate community in which is now the blind arm near the mouth (DWA 2010). Intertidal and subtidal sandbanks in the former channel near the mouth, running south and parallel to the coast for about 1 km, that is the present-day blind arm, would have provided extensive habitat for the sand infauna. Currently, the mouth and lower estuary are dredged to maintain a deep channel. Under present-day mouth conditions, tidal currents are stronger, and result in coarser sediments in the lower estuary, while under Reference conditions, in contrast, the choked mouth during summer would have resulted in finer sediments in the lower estuary. The present blind arm at the mouth has now become fine mud, compared to its previous sandy character, leading to a complete switch in species composition (DWA 2010).

Further reductions in river inputs will lead to further upstream shifts of the euryhaline community as the brackish zone is pushed back (Wooldridge & Deyzel 2010). Increased marine dominance is likely to lead to a change in invertebrate community structure, causing an upstream shift of a more marine associated fauna, especially with the canalisation of the mouth maintaining a permanent link with the marine environment and allowing tidal penetration further up the estuary (Wooldridge & Deyzel 2010). The level of saline intrusion upstream and the duration thereof will determine the extent of this change in the fauna. In addition, increased marine influence upstream may lead to water temperatures in summer which could also impact community structure and estuarine processes. It is likely that a more marine dominated system due to low freshwater input will result in an increased invertebrate biomass, although this would be counteracted to some extent by increases in algal growth and losses of eelgrass habitat in the lower estuary, and by reduced biomass of freshwater loving species in the upper estuary.

Projected changes in species richness, biomass and community composition of invertebrates under the various future flow scenarios are summarised in Table 6.4. It is estimated that the health of the invertebrate community has declined markedly from 54.0% of Reference in 2010 to 49.1% of Reference in 2020. This would decline slightly to 47.1% under the future development conditions if there were no change in climate (F0), but markedly to about 31.3% in a future with climate change (C0). Respecting low flow EWRs makes a very small improvement.

Table 6.4. Summary of projected changes in species richness, biomass and community composition of invertebrates under the various future flow scenarios (Hist. = Historical, P0 = Status Quo, EWR = Environmental Water Requirements , FD = Future Development, CC = Climate Change).

	Hist. 2010	PD 2020	P1	F0	F1	C0	C1
			PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
Species richness	80.0	72.8	73.4	69.8	69.8	46.3	45.7
Abundance/ Biomass	54.0	49.1	49.5	47.1	47.1	31.3	30.9
Community composition	60.0	54.6	55.0	52.4	52.4	34.8	34.3
Health score	54.0	49.1	49.5	47.1	47.1	31.3	30.9

6.5 Fish

6.5.1 Importance of the Berg River Estuary for fish

Estuaries offer a markedly different type of habitat compared to adjacent South African coastal waters in that they constitute the bulk of the sheltered, shallow water inshore habitat. Benefits provided by estuaries for fish are not restricted to shelter from wave action and also include improved access to food resources associated with elevated nutrient concentrations, high primary productivity, elevated water temperatures that are conducive to rapid somatic growth, and expansive areas of shallow and turbid water that offer some protection from predators (Wallace 1975, Claridge *et al.* 1986, Potter *et al.* 1990, Whitfield & Kok 1992). The Berg River Estuary is one of only three permanently open systems on the west coast of South Africa and is therefore of particular importance to fish (and other biota) that utilize marine and estuarine habitats during their lifecycles. Fish fauna inhabiting this system have been studied by a number of authors in the past (Day *et al.* 1981, Bennett 1993, Harrison 1997, Clark *et al.* 2009 Lamberth *et al.* 2010), all of whom have highlighted the importance of this system to the inshore marine fish fauna of the region. In all, 17 out of the 35 fish species (48%) recorded from the Berg River Estuary can be regarded as either partially or completely dependent on the estuary for their survival. Similarly, high dependence ratios have been reported for other west coast estuaries such as the Olifants and Orange, (Lamberth 2003, Lamberth *et al.* 2008). Bennett (1994) when studying the Berg River Estuary, made the observation that dependence ratios for west coast estuaries are much higher than that observed in estuaries on the south, east and KwaZulu-Natal coasts and concluded that estuaries on the west coast must be disproportionately important from a fish conservation perspective than those elsewhere in the country. Bennett (1994) further argues that the high reliance of the local fish on west coast

estuaries means that any degradation of the estuarine habitat will have worse consequences for fish on the west coast than elsewhere in South Africa. These arguments are strongly supported by the work of Lamberth *et al.* (2008) who highlights the fact that there are only nine functional estuaries on the west coast, and that of these, only the Berg and Olifants are permanently open, with the Orange now being intermittently closed. Consequently, each of these estuaries is crucial in maintaining the range and stock integrity of estuarine and estuarine dependent species along the entire west coast. In turn, the Berg River Estuary is an important nursery area for exploited marine and estuarine species before they move out of the estuary to recruit into the marine environment. The Berg Estuary has also been shown to serve as an important refuge for fish when low oxygen "black tide" events occur in St Helena Bay, with abundance of both marine and estuarine species increasing in the estuary during these events (Lamberth *et al.* 2010).

This section presents a summary of the findings of Clark *et al.* (2009), who provided an analysis of all available quantitative data on the Berg River Estuary fish fauna spanning the period 1992-2006 (14 surveys), and compares these to the results of a recent (February 2020) survey by DEFF that sampled the same sites using the same methods as the earlier surveys. A description of the current state of the Berg River Estuary fish community is provided, as is an assessment of changes and/or recovery post the unprecedented drought experienced over the 2015-2018 period. The response, (or lack thereof), of the Berg River Estuary fish community to changes in freshwater flows is investigated with a view to improved understanding of the likely impacts of future changes in freshwater flow.

6.5.2 Estuarine fish categories

Many marine fish species have acquired the necessary adaptations to enable them to utilise estuaries for at least part of their life cycles. Whitfield (2019) identified at least 100 species that show a clear association with estuaries in South Africa. Most of these are juveniles of marine species that enter estuaries as post-flexion larvae, where they remain for up to a year or more before returning to the marine environment as adults or sub-adults to spawn and hence complete their life cycles. Other fish species that use estuaries in southern Africa include some that are able to complete their entire life cycles in these systems, as well as a limited range of salt tolerant freshwater species, and a wider group of euryhaline marine species. This classification system was employed to subdivide the fish fauna of the Berg River Estuary into functional groups to enhance understanding of observed changes in community structure and abundance in response to changes in physical drivers).

Five major categories of estuary associated fish species and several subcategories are recognized under this system (Whitfield 1994):

- I. Estuarine species breeding in estuaries, further divided into:
    - Ia: Resident species not recorded spawning in marine or freshwater environment
    - Ib: Resident species also having marine and/or freshwater breeding populations
- II. Euryhaline marine species usually breeding at sea with juveniles showing varying degrees of dependence on estuaries, further divided into:
    - IIa: Juveniles dependent on estuaries as nursery areas
    - IIb: Juveniles occurring mainly in estuaries, but also found at sea

    - IIc: Juveniles occur mainly at sea, but also found in estuaries
- III. Marine species that occur in estuaries in small numbers but are not dependent on estuaries
- IV. Euryhaline freshwater species whose penetration into estuaries is determined primarily by salinity tolerance. Includes some species which may breed in both freshwater and estuaries
- V. Catadromous species which use estuaries as transit routes between the marine and freshwater environments but may also occupy estuaries in certain regions, further divided into:
    - Va: Obligate catadromous species which require a freshwater phase in their development
    - Vb: Facultative catadromous species which do not require a freshwater phase in their development but use estuaries as nursery areas.

Fish species in categories I, IIa and b, and V as defined by Whitfield (1994) are all wholly or largely dependent on estuaries for their survival and are hence the most important from an estuary conservation perspective, whilst many species in category IIc support nearshore marine and estuarine fisheries and are of socio-economic importance (Turpie & Lamberth 2003).

6.5.3 Description of the Berg River Estuary fish fauna

Between 14 and 23 species were caught in each of the historical surveys. The total number of fish collected varied by more than 10-fold, from a low of 20 402 (Summer 1996) following a severe drought and several low oxygen "black tide" events in St Helena Bay, to 243 226 (Summer 2006) (Table 6.5). The 2020 survey recorded 68 154 fish from 22 species, comparable to the diversity of catches in historical summer surveys, and very close to the average catch and abundance for all previous surveys (62 952) (Table 6.5). The same six species have dominated catches in all surveys, together contributing 93-99% of the catches in each year (Table 6.5). Southern mullet *Chelon richardsonii* is usually the most common species with the estuarine round-herring *Gilchristella aestuaria* being second most abundant and occasionally dominant (as in 2020). These, together with barehead goby *Caffrogobius nudiceps*, Cape silverside *Atherina breviceps*, Knysna sand goby *Psammogobius knysnaensis* and Mozambique tilapia *Oreochromis mossambicus* dominate the ichthyofauna (Clark *et al.* 2009, Table 6.5). Of these, *O. mossambicus* is an extralimital freshwater species (although it is tolerant of hyper saline conditions) and *G. aestuaria* is usually confined to estuaries (although some penetrate far upstream into freshwater and freshwater populations have evolved in some coastal lakes); the others are marine species that occur in estuaries. Contributions by the different species to total abundance varied from year to year and between seasons without any clearly discernible patterns (Clark *et al.* 2009). The 2020 survey revealed some changes in community composition from the general historical pattern of dominance by southern mullet, but this was not unprecedented in the historical data, with estuarine round herring being the most dominant species in both summer and winter surveys undertaken during 2005 (Figure 6.9).

C. richardsonii is the species of greatest fishery importance and is the target of a longstanding (>100 years) gill net fishery in the estuary itself, which due to its negative impacts on the nursery function provided by the Berg River Estuary, was officially closed in 2005, although illegal fishing

with an estimated catch of approximately 400 tonnes per annum continues (S.J. Lambetth, DEFF, pers. comm.). An ongoing legal, commercial gill net fishery also targets the species in the adjacent St Helena Bay (Hutchings & Lamberth 2002a,b). *C. richardsonii* and *O. mossambicus*, along with elf *Pomatomus saltatrix*, white steenbras *Lithognathus lithognathus* and the alien freshwater carp *Cyprinus carpio* support subsistence and recreational linefisheries within the Berg River Estuary (Hutchings *et al.* 2008).

Figure 6.8. The most abundant harvested fish species in the Berg River Estuary, southern mullet *Chelon richardsonii*

The floodplain surrounding the upper parts of the Berg River Estuary was sampled during four of the historical sampling excursions — Winter 1992, 2003, 2004, 2005, and Spring 2005 — being the only time when significant parts of the floodplain were covered with water. The number of species captured in these floodplain samples varied from 1 to 6 and included mostly alien freshwater species (bluegill sunfish *Lepomis macrochirus*, *O. mossambicus* , *C. carpio* and the indigenous Cape galaxia *Galaxias zebratus*) but also some estuarine residents (*P. knysnaensis* and *C. nudiceps*) and a marine migrant species (*C. richardsonii*) (Clark *et al.* 2009). Densities of fish at the floodplain sites were all very low (generally <0.3 fish.m^{-2}). *Galaxias zebratus* is the only species considered to show a strong association with the floodplain areas, as it was not recorded in the main part of the estuary at all. The African sharptooth catfish *Clarias gariepinus* has on occasion been caught in Berg estuary fish surveys (summer 2005, spring 2005 and summer 2020), and this extralimital freshwater species is thought to use the inundated flood plains for spawning (S.J. Lamberth, DEFF, pers. comm.). Due to the predominance of alien freshwater species, these ephemeral floodplain pools are considered of little overall importance to the indigenous estuarine fish community (Clark *et al.* 2009).

The relative contribution by numbers of fish in the various estuary dependence categories defined by Whitfield (1998) also exhibited no clearly discernible pattern from year to year, or between seasons (Clark *et al.* 2009, Figure 6.10). However, variations in the relative abundance of the various categories of estuarine associated fish species were clearly a function of variations in the abundance of the dominant fish species themselves namely *C. richardsonii* and *G. aestuaria* (Figure 6.10). The 2020 survey data reveals a very similar contribution to that recorded during the summer and winter 2005 surveys, with a dominance of estuarine resident species, mostly *G. aestuaria* Ia) *P. knysnaensis* and *C. nudiceps* (Ib) with marine migrants (IIc) the third most abundant group largely represented by *C. richardsonii*.

Table 6.5. Total catch, diversity and percentage composition of dominant species from historical (1992-2006) and recent (February 2020) surveys of the Berg River Estuary fish fauna. Source: Clark *et al.* 2009.

	Range (1992-2006)	Average (1992-2006)	2020
Number of species	14 - 23	17	22
Total catch	20 402 - 243 226	62 952	68 154
Abundance (ind.m^{-2})	2 - 26	7	6.7
Composition			
Chelon richardsonii (%)	25 - 85	59	29
Gilchristella aestuaria (%)	3 - 58	22	58
Atherina breviceps (%)	0 - 25	8	4
Psammogobius knysnaensis (%)	0 - 8	3	0
Caffrogobius nudiceps (%)	0 - 6	2	6
Oreochromis mossambicus (%)	0 - 14	2	2

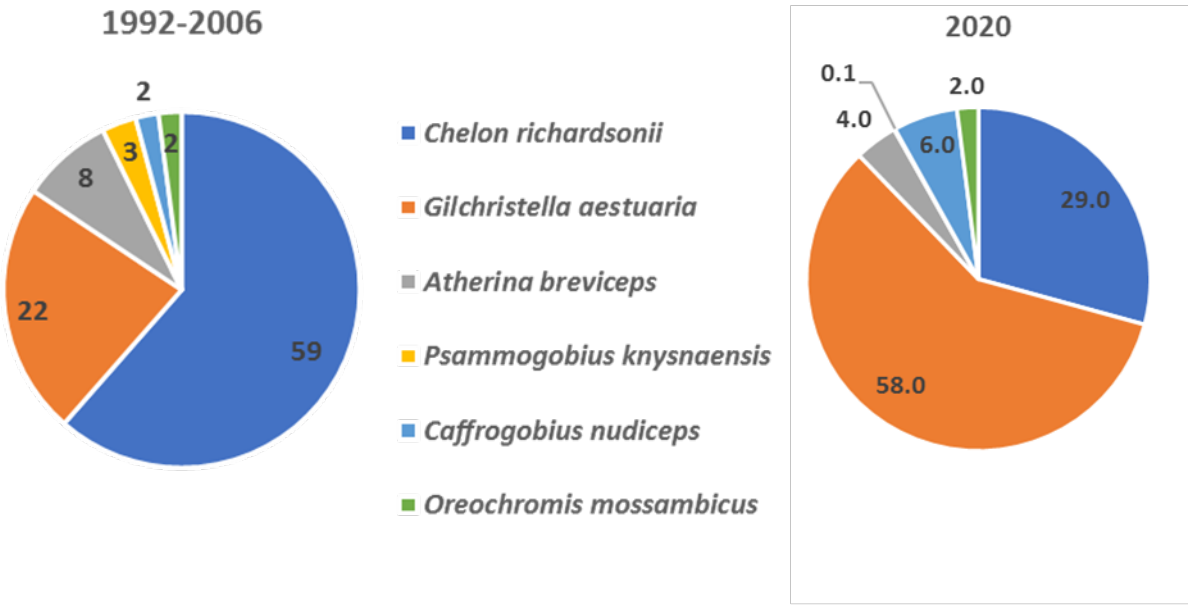

Figure 6.9. Relative proportion (%) of the six most abundant species in Berg River Estuary fish samples from historical (1992-2006) and recent (2020) seine net surveys. Source: Clark *et al.* (2009), this study.

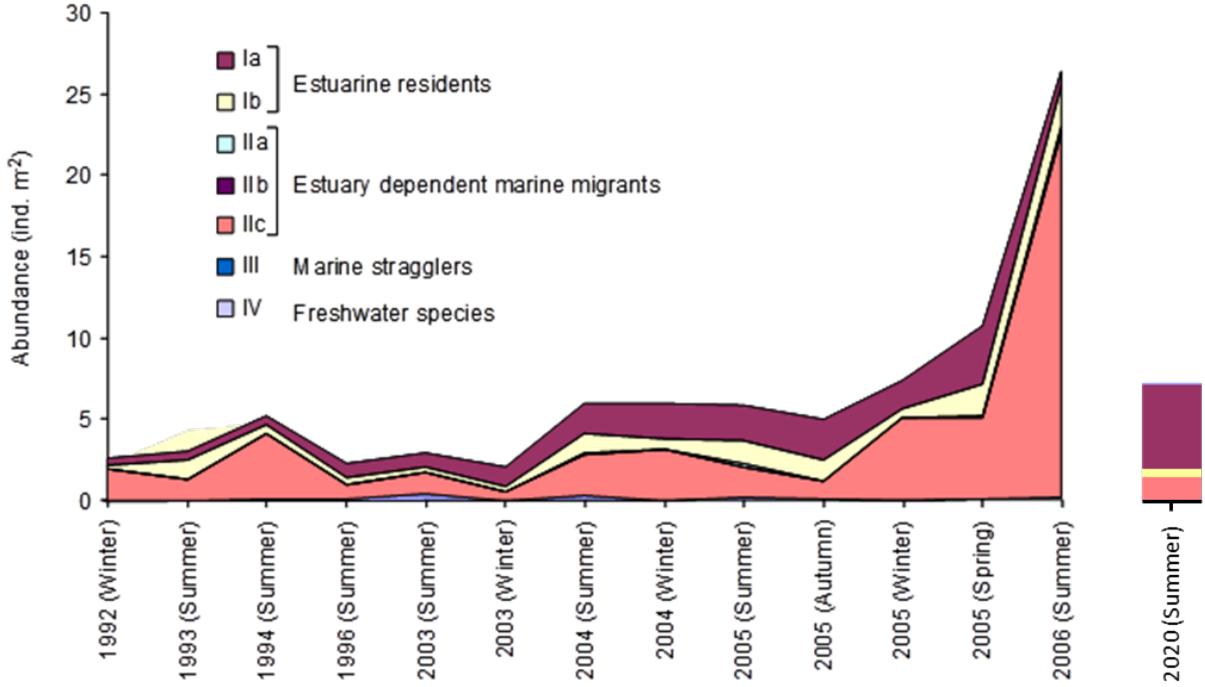

Figure 6.10. Contribution by various categories of estuary associated fish species following Whitfield's (1998) classification system (estuarine residents, marine migrants, marine stragglers and freshwater species) to the total abundance in the Berg River Estuary seine net surveys conductedover the period 1992-2006 and 2020.

Some species displayed clear and consistent distribution patterns up the length of the estuary (Clark *et al.* 2009). Some species (*G. aestuaria* and *Mugil cephalus*,) showed no clear preference for any particular portion of the estuary (Figure 6.11). Some (*P. knsynaensis, Caffrogobius gilchristi, P. saltatrix, Lichia amia, Lithoganthus lithoganthus and Solea bleekeri*) showed a clear preference for the middle reaches of the estuary (Figure 6.11). Some (*A. breviceps, C. richarsonii, Syngnathus temmincki, C. nudiceps, Clinus superciliosus, Rhabdosargus globiceps*, and *Rhinobatos blochii*) were more prevalent in the lower reaches (Figure 6.12), while alien freshwater species (*O. mossambicus, C. carpio, Gambusia affinis,* and *Micropterus dolomieu*) showed a clear preference for the upper reaches of the estuary (Clark *et al.* 2009). All of those favouring the upper reaches of the estuary were freshwater species, while those favouring the whole or other portions of the estuary comprised a mixture of estuarine resident and marine migrant species. The 2020 survey revealed similar distribution patterns for all marine migrant and freshwater species, but an upstream shift in the peak abundance of estuarine residents, namely *G. aestuaria*, *P. knsynaensis* and *Caffrogobius gilchristi (multifasciatus*) was apparent (Figure 6.11 and Figure 6.12). These three species still occupied much of their historical range throughout the estuary, but the peak abundance appears to have shifted upstream. This likely reflects the timing of the 2020 survey in late summer, (the historical data includes four summer surveys, three winter surveys, a spring and an autumn survey), with fish having undertaken a post-drought downstream shift to their normal salinity ranges, rather than a long-term shift in distribution due to the 2015-2018 drought.

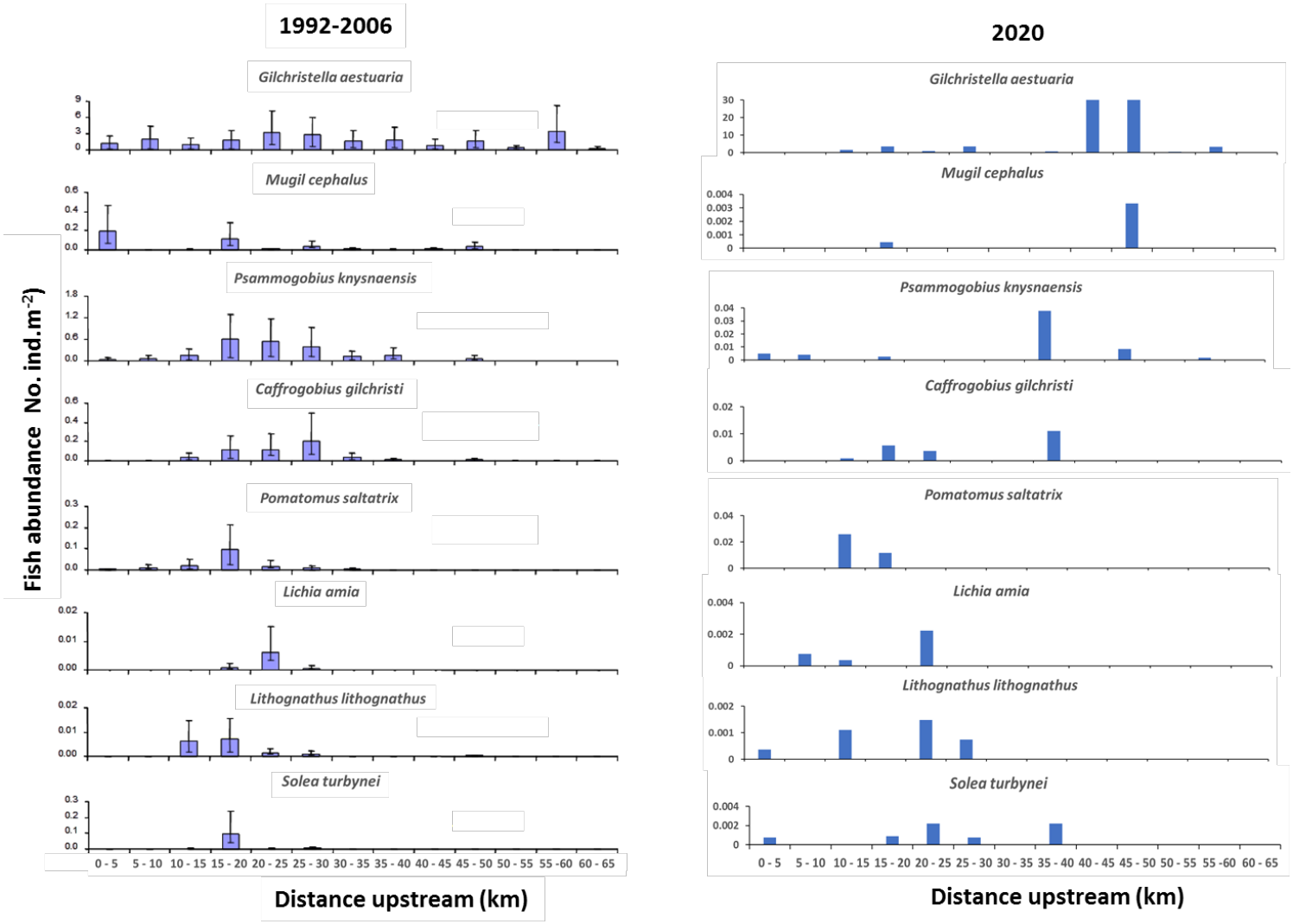

Figure 6.11. Variation in abundance of key fish species with distance upstream (I). Species distributed throughout the estuary and those favouring the middle reaches. Data from earlier survey (1992-2006) are shown on the left and 2020 survey data on the right. Source: Clark *et al.* (2009); this study.

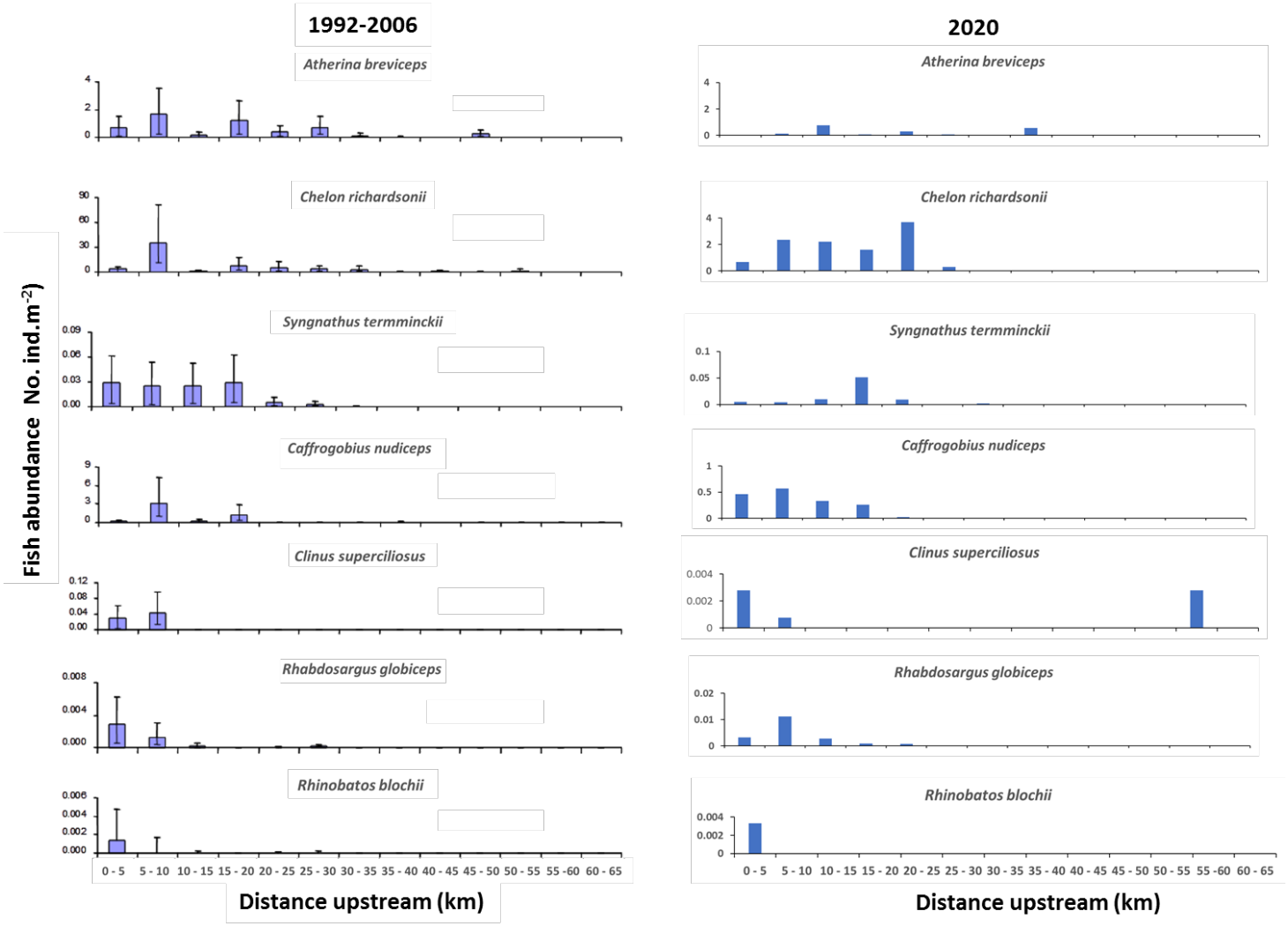

Figure 6.12. Variation in abundance of key fish species with distance upstream (II). Species favouring the lower reaches of the estuary. Data from earlier survey (1992-2006) are shown on the left and 2020 survey data on the right.. Source: Clark *et al.* (2009); this study.

6.5.4 Summary of fish community response to changes in freshwater flow

The fish fauna of the Berg River Estuary appear to show a limited response to fluctuations in freshwater flows with Clark *et al.* (2009) concluding that "The relative contribution by different groups of fish exhibited no clearly discernible pattern from year to year in relation to measured physico-chemical parameters including freshwater runoff to the estuary, or between seasons". The inference is not that there is no response in the estuarine fish fauna in response to flow, only that it was not discernible from data collected at the time, there may well be a delayed response. There does appear to be a lagged response between freshwater flow entering the Berg River Estuary and reported commercial gill net catches of *C. richardsonii* in St Helena Bay (S.J. Lamberth, DEFF pers. comm.). This is thought to be related to increased recruitment to the marine environment due to higher flow rates and to reduced gill net fishing effort in the estuary during high flow periods. It is pertinent to note that at least three of the fish surveys included in the data set analysed by Clark *et al.* (2009) reflect moderate drought conditions over the period 2003-2004. The 2020 fish survey revealed very similar fish diversity, abundance and community composition to that recorded in the summer of 2005 (i.e. post the 2003-2004 drought). The Berg River Estuary fish fauna seem to show a more dramatic response during the peak of the 2015-2018 drought, but these changes appear to have been reversible, with 2020 data (albeit a single survey) falling within the range of historical data from earlier surveys. To date, the west coast estuarine fish fauna therefore appears resilient to severe reductions in freshwater inflows and extended periods of a marine dominated estuarine state. Commercial catches of *C. richardsonii* in the sea are, however, correlated with estuary flow suggesting that there are negative impacts of flow reduction on the nursery function of the estuary (S.J. Lamberth, DEFF, Pers. comm.). This also does not imply that further reductions in freshwater flow could not have significant negative impacts on the Berg River Estuary fish fauna. Indeed, it is critical that there are sufficient freshwater flows to maintain a river-estuary interface zone required by most estuarine dependent species and that there is sufficient tidal intrusion of marine water to maintain suitable water quality (particularly in respect of salinity and oxygen).

6.5.5 Health assessment and scenario analysis

Under 2010 conditions, the fish fauna included significant populations of 12 fish species, down from an estimated 17 species under Reference conditions. Four species (white stumpnose, white steenbras, kob, leervis) were still represented in the system but such low numbers of individuals that they were not considered viable populations. At least one species had been lost from the system entirely (witvis *Barbus andrewi*). Reasons for the loss of these species are primarily non-flow related - overexploitation at national scale and introduction of alien invasive freshwater fish to the Berg system. Overall abundance in 2010 was estimated to be higher than under natural conditions and mostly related to increased abundance of the two dominant species– *Chelon richardsonii* and *Gilchristella aestuaria*. These two species, both filter feeders that can also switch to selective feeding, have benefitted from the increase in productivity of the system. The increase in *G. aestuaria* relative to *A. breviceps* also suggested that turbidity, (of which the former is more tolerant), had increased in the Berg estuary. Abundance of *C. richardsonii* was considered only marginally elevated above the Reference condition (~30%) but would have been very much higher in the absence of historical legal gill net fishing in the estuary and the sea, and the then illegal fishing activity in the estuary. Abundance of most of the marine migrant species (aside from *C. richardsonii*) was estimated to be severely depressed (mostly <10% of Reference) due to impacts of legal

and illegal fishing. Their numbers made up a very small proportion of total abundance and hence their loss contributed little to the change in abundance score. The reduction in low salinity habitat at the head of the estuary (i.e. between 35 and 45 km upstream of the mouth) would have negatively affected the abundance of many fish species in the estuary, particularly the estuarine resident species (e.g. *C. nudiceps*).

The health of the estuary under 2020 conditions was estimated on the basis of the hydrological changes as well as the fish survey in February 2020. The estuary remains an important nursery and feeding habitat for estuarine and marine fish on the west coast. The health of the fish community was at 67% of Reference, higher than estimated in 2010, due to the finding that several species thought to be in major decline in 2010 were still present in the system 14 years later. An average of 14 species (excluding alien freshwater taxa) from all surveys between 1992 and 2020 compared to the reference 17 species, resulting in an improved species richness score of 67%. The overall abundance of fish under present day conditions is considered to be higher than under reference, due to the rationale provided in the 2010 assessment that *Gilchristella aestuaria* have benefitted from enrichment, and increased marine dominance of the system and greater extent of open water with estuarine characteristics (salinity >0). This will also have benefited other estuarine species, given that freshwater flows would have limited the upstream influence of marine water under reference conditions. With the caveat that the 2020 data come from a single survey, *C. richardsonii* abundance and biomass is also expected to have increased in response to the increased nutrient and marine influence in the system, but had, however, declined by approximately 25% from reference, most likely due to overexploitation by ongoing legal marine and illegal estuarine gill net fishing (Horton 2018). A recent stock assessment of *C. richardsonii* in the nearby Saldanha Bay indicated that the stock was collapsed and a reduction in fishing mortality was required (Horton *et al.* 2019). The abundance of estuary-dependent marine species such as white steenbras, white stumpnose, elf and leervis are estimated to have declined by 90-99% from reference, primarily due to overfishing throughout their range.

Projected changes in species richness, biomass and community composition of estuarine fish under the future flow scenarios are summarised in Table 6.6. The health of the fish community is not expected to change much under future development or when summer low flow EWRs are respected but is projected to decline sharply under the influence of climate change (score drops to 50%) due to declines in estuarine residents and the loss of the remaining indigenous freshwater species (*G. zebratus)* and facultative catadromous species (*M. cephalus and Pseudomyxus capensis*) when freshwater inflows are severely limited.

Table 6.6. Changes in health scores for fish under the various flow scenarios (Hist. = Historical, P0 = Status Quo, EWR = Environmental Water Requirements , FD = Future Development, CC = Climate Change).

	Hist. 2010	PD 2020	P1	F0	F1	C0	C1
			PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
Species richness	56.0	66.8	66.8	66.8	66.8	50.1	50.1
Abundance/ Biomass	85.0	68.1	96.4	88.5	96.4	71.9	67.6
Community composition	86.7	88	86.3	85.8	86.3	85.5	85.5
Health score	56.0	66.8	66.8	66.8	66.8	50.1	50.1

6.6 Birds

6.6.1 Importance of the Berg River Estuary to birds

The Berg River Estuary is extremely important in terms of the diversity and abundance of bird populations that it supports, providing extensive and varied habitat for waterbirds. Since 1975, approximately 250 bird species have been recorded on the Berg River Estuary and adjacent floodplain, 127 of which are water-associated species (passerine and non-passerine). Cooper *et al.* (1976) and Ryan *et al.* (1988) identified the Berg River Estuary as being of international importance for waterbirds. Indeed, the estuary and wetlands support the highest recorded density of shorebirds on the East Atlantic Seaboard (Velasquez *et al.* 1991, Hockey *et al.* 1992), as well as significant populations of several threatened bird species, including African Marsh Harrier[5] and Cape Cormorant (regionally endangered), Caspian Tern and Great White Pelican (regionally vulnerable), Lesser Flamingo, Crowned Cormorant, Chestnut-banded Plover, Eurasian Curlew and Maccoa Duck (regionally and globally near threatened), Greater Flamingo and Greater Painted Snipe (regionally near threatened), and Bar Tailed Godwit and Red Knot (globally near threatened). The estuary is recognised as an Important Bird and Biodiversity Area by BirdLife International and is also under consideration for being assigned Ramsar status as a wetland of international importance.

This section of the report summarises available information on the waterbird fauna of the Berg River Estuary based on a desktop review of the literature and analysis of bird counts made under the Co-ordinated Waterbird Counts (CWAC) monitoring programme. The CWAC counts have been conducted at the estuary twice per year (mid-summer and mid-winter) since 1994.

6.6.2 Description of waterbird habitats

The ecological functioning of the Berg River Estuary is determined by seasonal changes in river flow and consequent changes in turbidity and salinity. In winter, the estuary is flooded by muddy, fresh river water and marine species tend to disappear during this time. In spring, the waters start to recede, salinity increases, and the system shifts back to a predominantly marine environment. During this time, the shallow pools on the floodplain begin to dry out and there is a marked increase in the number of waterbirds.

In addition to the river channel, the floodplain of the estuary encompasses eight major wetland types: ephemeral pans, commercial salt pans, reed marsh, sedge marsh, salt marsh, halophytic (salt tolerant plants) floodplain, xeric (dry) floodplain and intertidal mudflats (BirdLife South Africa, 2015). Exotic trees along the riverbank provide additional habitat for waterbirds, although this has been reduced with alien clearing along the banks by the Working for Water programme.

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[5] For all scientific bird names refer to Table 6.9.

Because of its permanently open status, the estuary supports a relatively large area of intertidal habitat, which is important foraging area for waders and roosting area for a number of other waterbird species. The intertidal and shallow subtidal habitats have a variable cover of eelgrass and the filamentous alga *Enteromorpha sp*. which is particularly abundant in spring and dies back during summer. This cover affects the abundance and accessibility of invertebrates, and thus affects avian foraging.

Marshes are one of the most abundant habitat types on the estuary and floodplain and are dominated by tall stands of reeds (about 2-4 m). The marshes are typically adjacent to the river channel and are favoured by skulking rallids and herons.

There are extensive areas of commercial saltpans on the Berg River Estuary, most of which have replaced saltmarsh areas[6]. Water is pumped into the saltpans which then dry by evaporation. The saltpans are all in various stages of this process, and thus offer a variety of habitats to birds, ranging from deep water to exposed shorelines and hypersaline pans. Benthic invertebrates are abundant in pans of 30-60 ppt, particularly chironomid fly larvae, polychaetes, amphipods and mudprawns (Murison & Hockey 2002). Brine shrimps are abundant in extremely high salinity pans (>100 ppt). Those that fill with freshwater attract waterfowl and some terns, while those filled with pumped seawater tend to attract gulls and terns. As the saltpans dry out, waders are attracted to the newly exposed shorelines and shallow water habitats. When conditions favour chironomid larvae, the pans can support extraordinary densities of waders, particularly small species such as Curlew Sandpiper (Hockey *et al.* 1998). Greater Flamingos appear as salinities increase, feeding on brine shrimp. Lesser Flamingos are tolerant of a higher salinity range, and feed on phytoplankton. When dry, the pans are used by certain resident waders, mainly Chestnut-banded Plover, Kittlitz's Plover and Whitefronted Plover.

6.6.3 Species and their groupings

In this report we only consider waterbirds, which are species that specifically tend to use aquatic environments for at least part of their lifecycle, for activities such as feeding, breeding or roosting. Estuarine waterbirds include both passerine and non-passerine species[7]. However, most of the passerine birds associated with aquatic environments, such as bishops, cisticolas and reed warblers, are difficult to census and are not typically included in waterbird counts. Thus, in this assessment, we only consider non-passerine estuarine waterbirds excluding vagrant, extralimital exotic, domestic species and hybrids. Furthermore, we do not include marine cormorants in our analysis of abundance as this group comprises cormorants that feed in marine environments and use the estuary to roost – Cape, Bank and

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6 It should be noted here that there is a possibility that the commercial Cerebos salt pans may close. This could have a significant negative impact on overall bird numbers and bird community composition if these artificial wetland habitats are indeed lost.

7 Passerine birds, also known as perching birds or songbirds, are distinguished from other orders of birds by the arrangement of their toes. They have three toes pointing forward and one back, which allows the bird to easily cling to both horizontal and nearly vertical perches, including branches and tree trunks.

Crowned Cormorants. This group is neither directly nor indirectly sensitive to flow and is thus not given much attention in this study.

The waterbirds of the Berg River Estuary can be divided into ten different taxonomic orders. Excluding exotic and vagrant species, some 93 non-passerine waterbird species have been recorded in seasonal counts of the estuary (Table 6.7). Of these, 75 species are South African residents and 18 species are palearctic migrants.

Table 6.7. Taxonomic composition of non-passerine waterbirds on the Berg River Estuary.

Common groupings	Order	SA Resident species	Palearctic migrant species	Total
Waterfowl	Podicipediformes (Grebes)	3		3
	Anseriformes (Ducks, geese)	12		12
	Gruiformes (Rails, crakes, gallinules, coots)	6		6
Cormorants, darters, pelicans	Pelecaniformes (Cormorants, darters, pelicans)	7		7
Wading birds	Ciconiiformes (Herons, egrets, ibises, spoonbill)	17		17
	Phoenicopteriformes (Flamingos)	2		2
Waders, gulls, terns	Charadriiformes: Waders	11	16	27
	Charadriiformes: Gulls	3		3
	Charadriiformes: Terns	6	2	8
Kingfishers	Alcediniformes (Kingfishers)	3		3
Birds of prey	Falconiformes (Birds of prey)	4		4
	Strigiformes (Owls)	1		1
Total		75	18	93

Charadriiformes (waders, gulls and terns) account for 41% of the species recorded, with most of these being wader species (Table 6.7). More than half of the 27 wader species are regular migrants from the Palaearctic region of Eurasia, making up 83% of the summer wader population by numbers. Apart from these and two migratory tern species, the remaining species are species that breed in southern Africa, some making local or regional movements in response to rainfall. Among the resident species, the Ciconiiformes (herons, egrets, ibises, spoonbill) and Anseriformes (ducks) form the most diverse groups on the estuary, but most waterbird orders are well represented.

For the purpose of this analysis, we have also grouped the species into 13 functional groups on the basis of a combination of taxonomic and trophic characteristics. These groups are described in more detail in Table 6.8.

Table 6.8. Description each functional bird group found in the Berg River Estuary and their defining features.

Bird groups	Defining features, typical/dominant species
Herbivorous waterfowl	This group is dominated by species that tend to occur in low salinity or freshwater habitats and are associated with the presence of aquatic plants such as *Potamogeton* (pond weed) and *Phragmites* (common reed). The group includes some of the ducks (e.g. Southern Pochard), and all the rallids (e.g. Red-knobbed Coot, African Purple Swamphen). Some herbivorous waterfowl such as Egyptian Goose, Spur-winged Goose and South African Shelduck may do part or most of their feeding in terrestrial areas.
Omnivorous waterfowl	This group comprises ducks, which eat a mixture of plant material and invertebrate food such as small crustaceans - Yellow-billed Duck, African Black Duck, Cape Teal, Hottentot Teal, Red-billed Teal and Cape Shoveller. Most are tolerant of more saline conditions, but African Black Duck tends to be restricted to freshwater areas of higher flow.
Piscivorous waterfowl	This group comprises the grebes – Great Crested, Black-necked and Little Grebe. The first two tend to be restricted to lower salinities and deeper water, and Little Grebe tends to be found where there is abundant marginal vegetation.
Pelicans	Pelicans are piscivorous and feed from the surface while swimming.
Cormorants & darters	Cormorants and darters are piscivorous and feed while swimming and diving from the surface. This group includes marine cormorants that might not be feeding in the estuary, as well as Reed Cormorant and African Darter. The marine cormorants were not included in our analysis of abundance.
Omnivorous wading birds	This group comprises the herons, egrets, bitterns and spoonbill, which are carnivorous, and feed on a variety of live aquatic prey. Their diet includes a wide variety of aquatic animals, including fish, reptiles, amphibians, crustaceans, molluscs, and aquatic insects. They mainly feed by sitting and waiting or walking slowly. They tend to be tolerant of a wide range of salinities.
Benthivorous wading birds	This group comprises the ibises, openbilled storks and greater flamingos. Ibises have a fairly plastic diet, but this is dominated by invertebrates. Openbilled storks differ from other storks in that they are specialised invertebrate feeders (mainly bivalves and snails). Greater Flamingos feed on benthic invertebrates in a wide range of salinities.
Planktivorous wading birds	This group comprises one species, Lesser Flamingo, which is unique in its diet (phytoplankton) and salinity tolerance, tolerating high salinity to hypersaline conditions.
Waders	This group includes all the waders (e.g. Greenshank, Curlew Sandpiper). They are the smallest species and most numerous group on the estuary, and feed on benthic macroinvertebrates in exposed and shallow intertidal areas.
Gulls and terns	This group comprises the rest of the Charadriiformes. These species are primarily piscivorous, but also take invertebrates. Most are euryhaline, but certain tern species tend to be associated with low salinity environments.
Piscivorous birds of prey	This group comprises species such as African Fish Eagle and Osprey. Osprey are not confined to a diet of fish, also taking other vertebrates and invertebrates. These species are tolerant of a wide range of salinities but require marginal vegetation, particularly trees or shrubs.
Other birds of prey	This group includes African Marsh Harrier and Marsh Owl.
Kingfishers	This group includes several species of kingfishers that hunt small fish from perches on marginal vegetation or by hovering.

6.6.4 Species richness, abundance & community composition

An average of 62 (62.5 ± 4.71) and 59 (59.0 ± 4.55) non-passerine waterbird species have been recorded on the Berg River Estuary in summer and winter CWAC counts, respectively (1994-2019). The highest summer count was 70 species in 2004 and the highest winter count was 68 species in 2015. There has been a noticeable change in species richness over the CWAC count period in winter and summer (Figure 6.13). Between 1994 and 2006 species richness remained relatively stable. However, since 2007 the average number of non-passerine waterbird species recorded on the estuary has declined. Between 1994 and 2006 the average species count was 65 in summer and 61 in winter. This has decreased to an average of 60 and 57, respectively from 2007 to 2019.

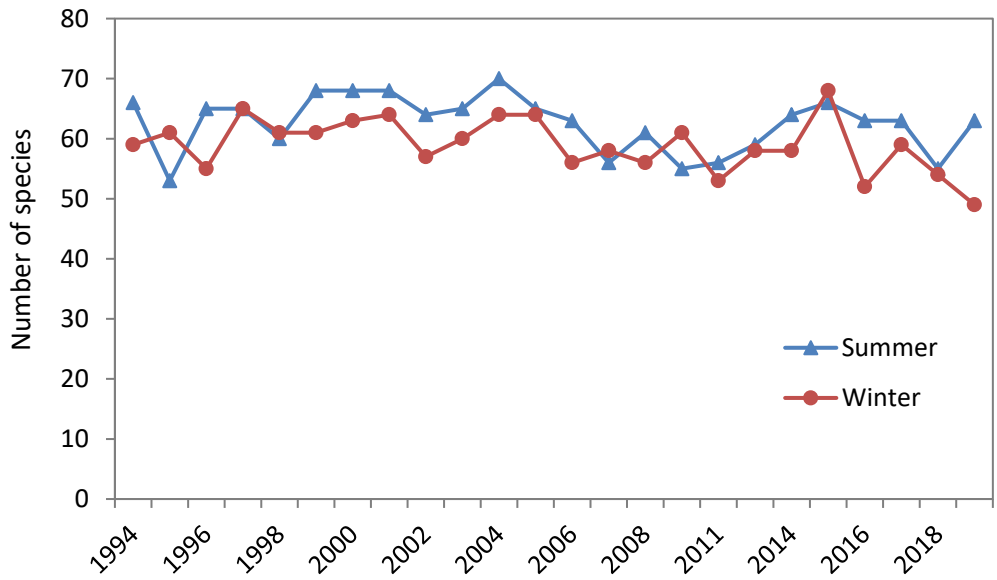

Figure 6.13. Number of non-passerine waterbird species recorded in CWAC counts of the estuary. Source: CWAC data.

An average of 13 178 and 10 454 non-passerine waterbirds were recorded in summer and winter CWAC counts, respectively. This excludes marine cormorants. Average numbers of each species are given in Table 6.9.

The relative contribution of each bird group to the bird numbers on the lagoon differs substantially over the summer and winter months, due to the prevalence of migratory birds arriving in summer (Figure 6.14). Waders account for almost half of the birds on the estuary during summer, with 83% of these being migratory. Other numerically important groups during summer are the benthivorous wading birds (ibises and Greater Flamingo) and gulls and terns. Avifaunal composition changes significantly in winter, with a far more even representation of taxonomic groups. Resident wader numbers increase slightly, and numbers of waterfowl and planktivorous wading birds (i.e. Lesser flamingos, Figure 6.14) increase substantially. Gulls and terns, cormorants and darters, kingfishers and birds of prey remain relatively stable throughout the year with little seasonal difference.

The influx of waders into the area during the summer months accounts for most of the seasonal change in community composition. Most of the Palaearctic migrants depart quite

synchronously in early April, but the immature birds of many of these species remain behind. The resident species take advantage of the reduced competition for resources and use this period to breed. The migrants return more gradually in spring, with birds beginning to filter in from August, and numbers rising rapidly during September to November.

Table 6.9. Non-passerine waterbird species recorded regularly on the Berg River Estuary Dec 1994 – Dec 2019, giving common and scientific names, and the average and maximum numbers recorded. Exotic, vagrant and extralimital species are excluded. Source: CWAC data

Common name	Scientific name	Mean		Max	
		Summer	Winter	Summer	Winter
Grebe, Great Crested	*Podiceps cristatus*	13.0	4.2	45	13
Grebe, Black-necked	*Podiceps nigricollis*	76.1	542.7	346	1 230
Grebe, Little	*Tachybaptus ruficollis*	27.9	43.0	106	100
Pelican, Great White	*Pelecanus onocrotalus*	81.0	58.8	227	298
Cormorant, White-breasted	*Phalacrocorax carbo*	189.9	101.6	643	270
Cormorant, Cape	*Phalacrocorax capensis*	1 995.8	7 394.6	23 284	56 866
Cormorant, Reed	*Phalacrocorax africanus*	129.2	97.5	273	266
Cormorant, Crowned	*Phalacrocorax coronatus*	1.7	8.7	21	74
Darter, African	*Anhinga rufa*	42.8	95.8	93	226
Heron, Grey	*Ardea cinerea*	67.5	46.1	116	104
Heron, Black-headed	*Ardea melanocephala*	5.6	10.2	20	134
Heron, Goliath	*Ardea goliath*	0.7	0.5	4	3
Heron, Purple	*Ardea purpurea*	8.2	4.2	21	12
Egret, Great	*Egretta alba*	0.5	1.3	6	7
Egret, Little	*Egretta garzetta*	75.7	56.2	141	87
Egret, Yellow-billed	*Egretta intermedia*	1.0	1.5	12	7
Egret, Cattle	*Bubulcus ibis*	5.5	14.3	52	69
Heron, Squacco	*Ardeola ralloides*	0.1	-	2	-
Night-Heron, Black-crowned	*Nycticorax nycticorax*	3.9	10.8	40	41
Night-Heron, White-backed	*Gorsachius leuconotus*	0.1	-	2	-
Bittern, Little	*Ixobrychus minutus*	0.2	0.2	1	1
Hamerkop, Hamerkop	*Scopus umbretta*	-	0.0	-	1
Stork, Black	*Ciconia nigra*	0.1	-	2	-
Ibis, African Sacred	*Threskiornis aethiopicus*	169.6	111.8	373	360
Ibis, Glossy	*Plegadis falcinellus*	4.2	97.4	40	420
Spoonbill, African	*Platalea alba*	80.2	60.8	220	126
Flamingo, Greater	*Phoenicopterus ruber*	1 357.1	1 628.8	3 254	3 071
Flamingo, Lesser	*Phoenicopterus minor*	339.3	1 333.9	1 604	3 346
Duck, White-faced	*Dendrocygna viduata*	0.1	0.3	2	7
Duck, Fulvous	*Dendrocygna bicolor*	-	-	-	-
Goose, Egyptian	*Alopochen aegyptiacus*	216.4	184.2	712	585
Shelduck, South African	*Tadorna cana*	132.8	123.1	601	405
Duck, Yellow-billed	*Anas undulata*	270.5	157.4	1 059	339

Common name	Scientific name	Mean		Max	
		Summer	Winter	Summer	Winter
Duck, African Black	*Anas sparsa*	0.1	1.3	2	11
Teal, Cape	*Anas capensis*	261.2	167.7	1 126	459
Teal, Hottentot	*Anas hottentota*	0.1	0.1	3	2
Teal, Red-billed	*Anas erythrorhyncha*	20.2	43.3	161	188
Shoveler, Cape	*Anas smithii*	43.2	121.0	245	303
Pochard, Southern	*Netta erythrophthalma*	0.5	13.0	8	92
Goose, Spur-winged	*Plectropterus gambensis*	170.3	65.8	1 279	178
Duck, Maccoa	*Oxyura maccoa*	1.8	12.8	36	119
Fish-eagle, African	*Haliaeetus vocifer*	3.0	3.8	8	16
Marsh-harrier, African	*Circus ranivorus*	3.1	6.7	10	15
Harrier, Black	*Circus maurus*	0.2	0.3	4	3
Osprey, Osprey	*Pandion haliaetus*	1.9	0.6	5	3
Rail, African	*Rallus caerulescens*	0.3	1.1	3	3
Crake, Black	*Amaurornis flavirostris*	0.2	0.3	2	3
Flufftail, Red-chested	*Sarothrura rufa*	-	-	-	-
Swamphen, African Purple	*Porphyrio madagascariensis*	3.6	2.5	16	18
Moorhen, Common	*Gallinula chloropus*	3.0	3.9	11	15
Coot, Red-knobbed	*Fulica cristata*	165.0	2 141.9	594	6 614
Painted-snipe, Greater	*Rostratula benghalensis*	0.0	-	1	-
Oystercatcher, African Black	*Haematopus moquini*	2.5	3.5	18	46
Plover, Common Ringed	*Charadrius hiaticula*	134.7	4.8	475	67
Plover, White-fronted	*Charadrius marginatus*	11.2	27.9	34	182
Plover, Chestnut-banded	*Charadrius pallidus*	86.5	115.3	295	254
Plover, Kittlitz's	*Charadrius pecuarius*	376.4	190.9	1 293	297
Plover, Three-banded	*Charadrius tricollaris*	17.0	27.9	93	84
Plover, Grey	*Pluvialis squatarola*	130.8	14.4	417	73
Lapwing, Blacksmith	*Vanellus armatus*	147.2	155.3	294	383
Turnstone, Ruddy	*Arenaria interpres*	1.3	7.1	7	70
Sandpiper, Terek	*Xenus cinereus*	0.1	0.0	2	1
Sandpiper, Common	*Actitis hypoleucos*	13.2	12.3	50	204
Sandpiper, Wood	*Tringa glareola*	4.0	0.5	26	6
Sandpiper, Marsh	*Tringa stagnatilis*	48.1	7.8	195	66
Greenshank, Common	*Tringa nebularia*	278.0	63.1	629	217
Knot, Red	*Calidris canutus*	4.6	0.1	54	1
Sandpiper, Curlew	*Calidris ferruginea*	2 853.0	302.7	9 341	769
Stint, Little	*Calidris minuta*	1 122.8	40.1	2 600	410
Sanderling	*Calidris alba*	33.2	0.5	437	4
Ruff	*Philomachus pugnax*	102.2	1.1	688	9
Snipe, African	*Gallinago nigripennis*	1.1	0.4	8	3
Godwit, Bar-tailed	*Limosa lapponica*	8.5	2.5	46	19
Curlew, Eurasian	*Numenius arquata*	17.3	4.8	48	18

Common name	Scientific name	Mean		Max	
		Summer	Winter	Summer	Winter
Whimbrel, Common	*Numenius phaeopus*	44.5	5.0	78	13
Avocet, Pied	*Recurvirostra avosetta*	101.3	229.7	292	467
Stilt, Black-winged	*Himantopus himantopus*	303.7	520.1	462	893
Thick-knee, Water	*Burhinus vermiculatus*	1.4	3.0	21	14
Gull, Kelp	*Larus dominicanus*	334.8	515.8	737	1 263
Gull, Grey-headed	*Larus cirrocephalus*	33.5	2.7	443	10
Gull, Hartlaub's	*Larus hartlaubii*	487.9	511.8	928	1 100
Tern, Caspian	*Sterna caspia*	46.3	30.3	138	216
Tern, Swift	*Sterna bergii*	75.3	111.3	312	1 604
Tern, Lesser Crested	*Sterna bengalensis*	0.3	-	6	-
Tern, Sandwich	*Sterna sandvicensis*	79.4	3.0	256	41
Tern, Common	*Sterna hirundo*	329.9	28.8	1 561	405
Tern, Little	*Sterna albifrons*	6.8	0.1	67	2
Tern, Whiskered	*Chlidonias hybrida*	2.3	0.2	33	2
Tern, White-winged	*Chlidonias leucopterus*	24.3	0.5	132	7
Owl, Marsh	*Asio capensis*	0.2	0.0	4	1
Kingfisher, Pied	*Ceryle rudis*	53.8	53.1	81	87
Kingfisher, Giant	*Megaceryle maximus*	0.3	0.3	2	3
Kingfisher, Malachite	*Alcedo cristata*	1.0	1.0	5	5

6.6.5 Patterns of distribution

Individual bird species tend to be highly consistent in their distribution along the Berg River Estuary, although some may shift their distribution seasonally. Certain birds are highly site specific. For example, the majority of Great-crested Grebes and Maccoa Ducks are found in one particular pan, Cape Cormorants and Crowned Cormorants are almost entirely confined to the mouth, and White-breasted Cormorants concentrate at the mouth but are found throughout the estuary (Table 6.10). Several species are confined or largely confined to the lower estuary (up to the railway bridge, some 11 km from the mouth). These are mostly migrant waders such as Grey Plover, but also include resident waders such as African Black Oystercatcher – Least Concern, population increasing, White-fronted Plover – Least Concern, although population decreasing, Kelp Gull, Swift Tern and Greater and Lesser Flamingos. Almost all other waders are generally more common in the lower than the upper estuary. This pattern also holds for African Spoonbill and Red-knobbed Coot. However, there can be strong seasonal distribution shifts in some species, such as the African Spoonbill, who breed in their hundreds in the reed marsh in the upper estuary when conditions are optimal.

Several species are fairly evenly distributed throughout the estuary (Table 6.10). These are mostly piscivorous species such as Great White Pelican, Reed Cormorant and Darter, but also include some waterfowl and African Marsh Harrier. Pied Kingfisher, Sacred Ibis, Little Egret and most herons are found throughout the estuary, tending to be slightly more common towards the mouth. Some species distributed throughout the estuary tend to be more common in the upper estuary. These are mostly waterfowl, but also include African Fish

Eagle, Moorhen and Wood Sandpiper (Table 6.10).

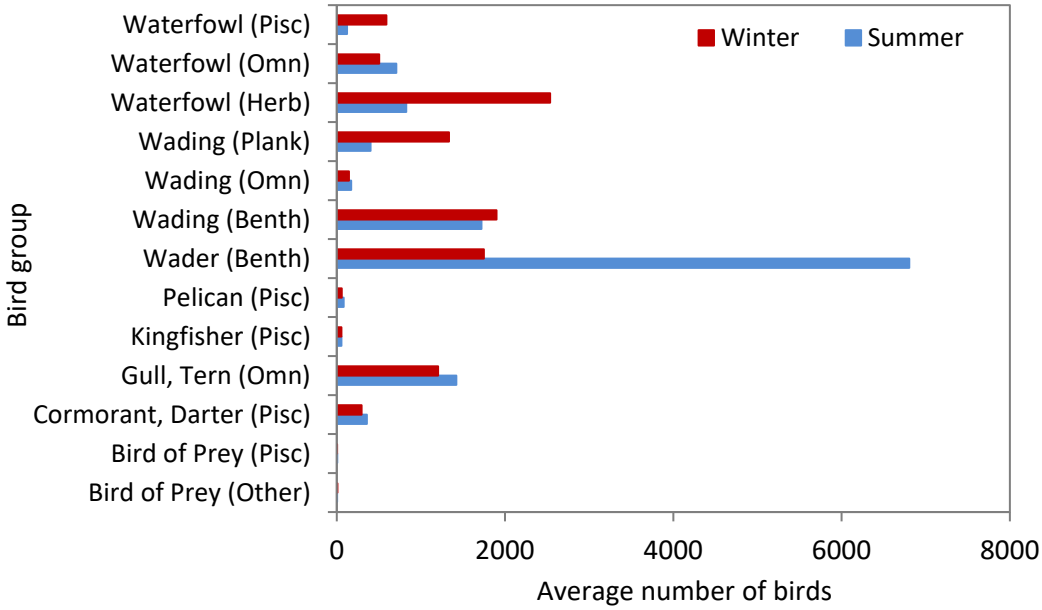

Figure 6.14. Average numerical composition of the birds on the Berg River Estuary during summer and winter (1994-2019). Pisc=piscivorous, Omn=omnivorous, Herb=herbivorous, Plank=Planktivorous, Benth=Benthic. See Table 6.8 for a description of each functional bird group. Source: CWAC data.

Table 6.10. Species typical of different parts of the Berg River Estuary and those that are common throughout

	Mouth	Lower estuary	Upper estuary
All of these or nearly all of these are found here	Cape Cormorants, Crowned Cormorants	Black-necked Grebe, African Black Oystercatcher, White-fronted Plover, Grey Plover, Ruddy Turnstone, Terek Sandpiper, Knot, Bar-tailed Godwit, Eurasian Curlew, Common Whimbrel	Ethiopian Snipe, African Black Duck, Goliath Heron, Marsh Owl
Most of these are commonly found here	White-breasted Cormorant	African Spoonbill, Red-knobbed Coot, Kelp Gull, Swift Tern, Greater Flamingo, Lesser Flamingo, Ringed Plover, Chestnut-banded Plover, Kittlitz's Plover, Three-banded Plover, Blacksmith Plover, Common Sandpiper, Marsh Sandpiper, Greenshank, Curlew Sandpiper, Little Stint, Sanderling, Ruff, Avocet, Black-winged Stilt.	Egyptian Goose, Red-billed Teal, Spur-winged Goose, African Fish Eagle, Moorhen, Wood Sandpiper
These species are common throughout	Great White Pelican, Reed Cormorant and Darter, Glossy Ibis, South African Shelduck, Yellow-billed Duck, African Marsh Harrier, Pied Kingfisher, Sacred Ibis, Little Egret, herons		

Very few species are confined to the upper estuary (from the railway bridge 11 km from the mouth to the top of the estuary at Steenbokfontein). These include Ethiopian Snipe, plus several species that are only occasionally recorded on the estuary, such as African Black Duck, Goliath heron, and Marsh Owl.

At a broad scale, there is marked spatial variation in bird community composition along the estuary. In summer, the mouth area is generally dominated by marine cormorants, terns and gulls, plus a few egrets, ibises and waders. Terns are usually replaced by flamingos and coots in winter. The numbers of marine cormorants peak during winter and mid-summer. The lower estuary is dominated by waders during summer. Flamingos are also a dominant group in the lower estuary during this time. There is a high diversity of other groups but in relatively small numbers. Less than a quarter of the waders remain in winter (resident species plus immature migrants), while flamingos increase in number and large numbers of coots appear in the lower estuary. While relatively few ducks occur in the lower estuary, they make up a large majority of the birds on the upper estuary during summer. The pelicaniformes (cormorants, pelicans) and ciconiformes (herons, egrets, ibises, spoonbills) tend to be more prevalent in the upper estuary than lower sites. Numbers of omnivorous ducks tend to drop in winter with herbivorous waterfowl increasing over these months, likely due to the increase in freshwater flows and the increasing abundance of freshwater plant food. Flamingos and coots increase in abundance during winter, linked to the greater degree of flooding of the floodplain over these months.

6.6.6 Long-term trends in abundance

The biannual CWAC count data provide the opportunity to examine the long-term trends in bird numbers at the Berg River Estuary up to the present day. This reveals an exponential decline in numbers of waterbirds on the estuary since 1994 (Figure 6.15). Bird numbers estimated in recent counts are only 34% of the numbers recorded in the baseline count of 1994.

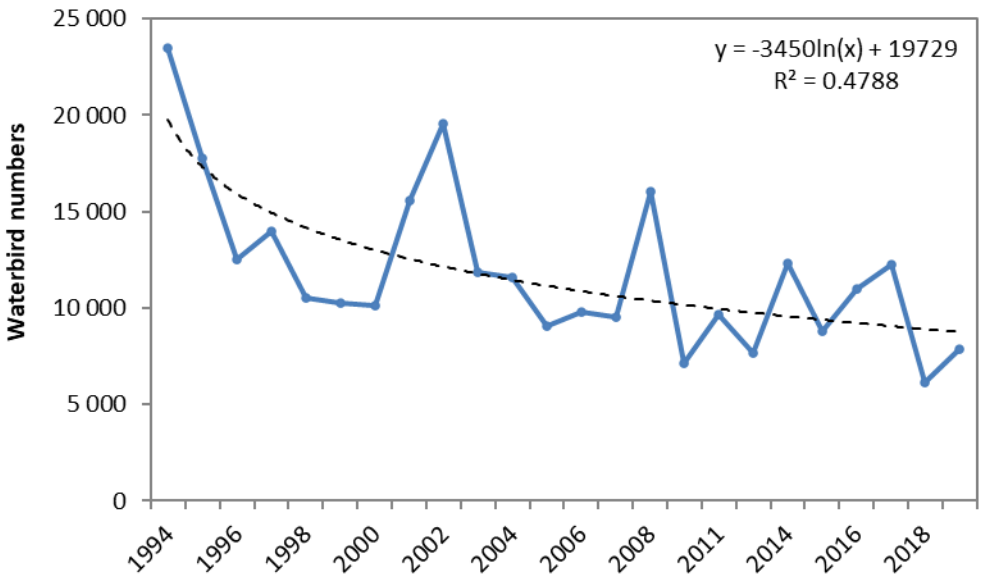

Figure 6.15. Changes in abundance at the Berg River Estuary 1994-2019 (excluding marine cormorants). Source: CWAC data.

To understand more about this decline we need to explore bird numbers by functional group. Wader numbers reveal a downward trend in the abundance of Palearctic waders at the estuary since 1994, especially in the last ten years (Figure 6.16). Very low numbers of Grey Plover, Curlew Sandpiper and Little Stint have been recorded on the estuary since 2009. The

total count of 2907 migratory waders in 2019 indicates an approximate 79% decline in numbers from the 1994 count of 13 983 birds.

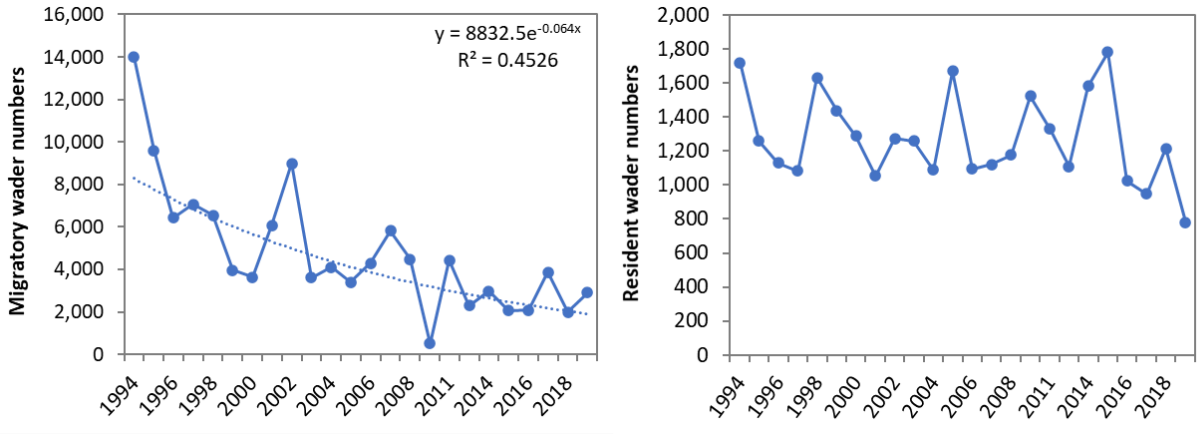

Figure 6.16. Long term trends in the numbers of (a) migratory waders and (b) resident waders on the Berg River Estuary. Source: CWAC data.

The reasons for these declines are varied and poorly understood but seem to be a result of the combination of loss and degradation of their breeding sites as well as of their over-wintering grounds during their non-breeding period (Dias *et al.* 2006). This downward trend in migrant wader numbers may echo global trends in certain wader populations. Indeed, Ryan (2012) reports on similar declines in migrant waders throughout the Western Cape over the last three decades, irrespective of the protection status of the areas where counts were undertaken. This suggested that factors outside of the Western Cape were at least partially responsible for the observed trends and probably reflect global population declines (Ryan 2012). However, what is also of concern is that there appears to be a declining trend in the number of resident waders (Figure 6.16). In recent years (2015-2019) winter resident wader numbers have declined substantially with counts lingering at ~60% of the pre-1995 average (Figure 6.16). A total of 781 resident waders were counted in 2019, the lowest since counting started in 1994.

This suggests that the conditions at the Berg River Estuary are at least partially to blame for the decline in wader numbers (migrant and resident species) during the study period. The most likely problems include loss or changes in feeding habitat with their associated invertebrate fauna and human disturbance, which has been shown to have a dramatic impact on bird numbers in other estuaries. Some important feeding areas lie within the zones that are intensively utilised for recreation. The low flows into the estuary during the drought of 2015-2018 may also have contributed to the noticeable decline in certain waterbirds during this time.

Herbivorous waterfowl, such as the South African Shelduck, Southern Pochard, Red-knobbed Coot and African Purple Swamphen occur in low salinity or freshwater habitats and are associated with the presence of aquatic plants. This group of waterbirds has also shown an exponential decrease in numbers over time, reaching as few as four birds counted in summer 2018 (Figure 6.17). Since the start of the drought in 2015, herbivorous waterfowl numbers have decreased consistently, only rising again slightly in the winter of 2018 and the summer of 2019 when freshwater flows into the estuary started to increase. This is expected, given their

low tolerance of saline waters and their association with freshwater aquatic plants that would have died off during the hypersaline conditions associated with the drought.

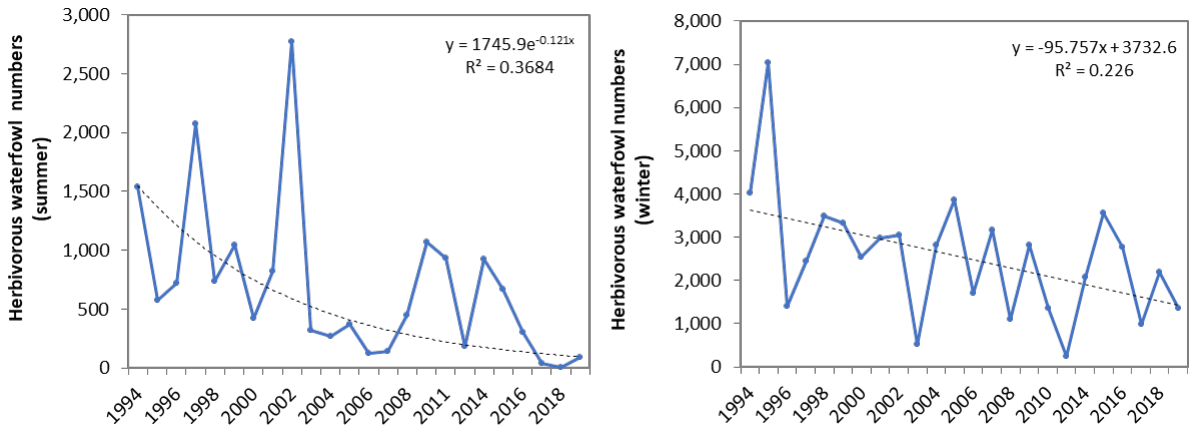

Figure 6.17. Long term trends in the numbers of herbivorous waterfowl on the Berg River Estuary in (a.) summer and (b.) winter. Source: CWAC data.

Great White Pelicans have also experienced a decline in numbers over the past decade (Figure 6.18). The abundance of the Great White Pelican has been in decline for some years but appears to have stabilised over the last few years at around 50-60 birds.

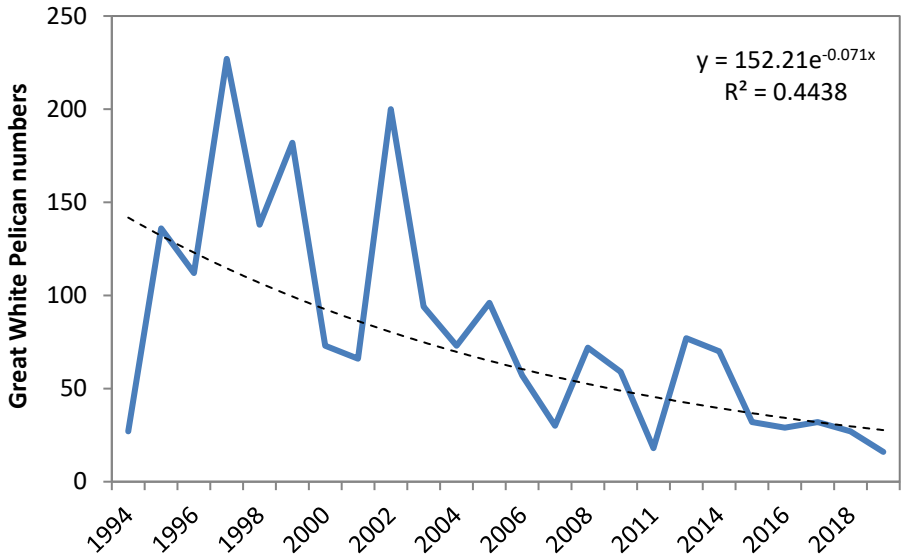

Figure 6.18. Long term trends in the numbers of Great White Pelicans on the Berg River Estuary. Source: CWAC data.

The long-term trends in the numbers of piscivorous waterfowl, birds of prey, kingfisher and cormorant and darters is shown in Figure 6.19. There has been a steady increase in the number of these birds on the estuary over time.

In all of these groups, including the cormorants and darters, their numbers have spiked in years with particularly low flows (2002-2003, 2015-2017). This suggests that a more marine-dominated system favours these birds. There could be a number of reasons for these increases. Firstly, the number of juvenile marine fish increase in the estuary when freshwater flows are reduced, increasing the amount of food that is available for this group of birds. Secondly, when the estuary is in a marine dominated state, the water is clear. Piscivorous birds are visual predators and it could be that when the water is clear the birds find it easier to catch their food.

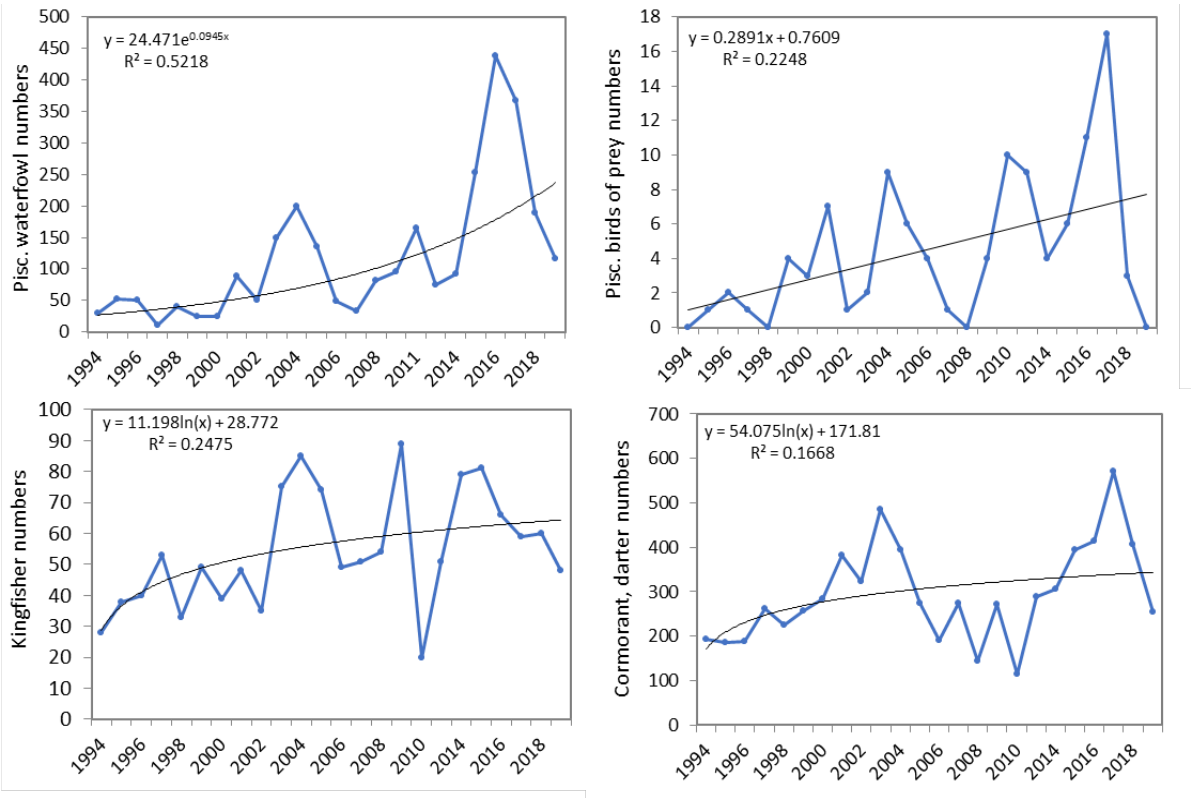

Figure 6.19. Long term trends in the numbers of (a.) piscivorous waterfowl, (b.) piscivorous birds of prey, (c.) kingfisher and (d.) cormorant and darters (excluding marine cormorants) on the Berg River Estuary. Source: CWAC data.

When the numbers of all piscivorous waterbirds (piscivorous waterfowl, piscivorous birds of prey, cormorants and darters, pelicans and kingfishers) from summer and winter counts are combined it becomes clear that the drier years (2002-2003, 2015-2017), especially in winter, yield higher numbers of these birds (Figure 6.20). These spikes in abundance in winter during dry years is likely due to the loss in other winter-feeding grounds that would usually be utilised by these birds. During dry years, a number of the freshwater wetlands and dams would dry out, forcing these birds to find alternative feeding grounds. Therefore, it is likely that the estuary is an important oasis for these birds during excessively dry years.

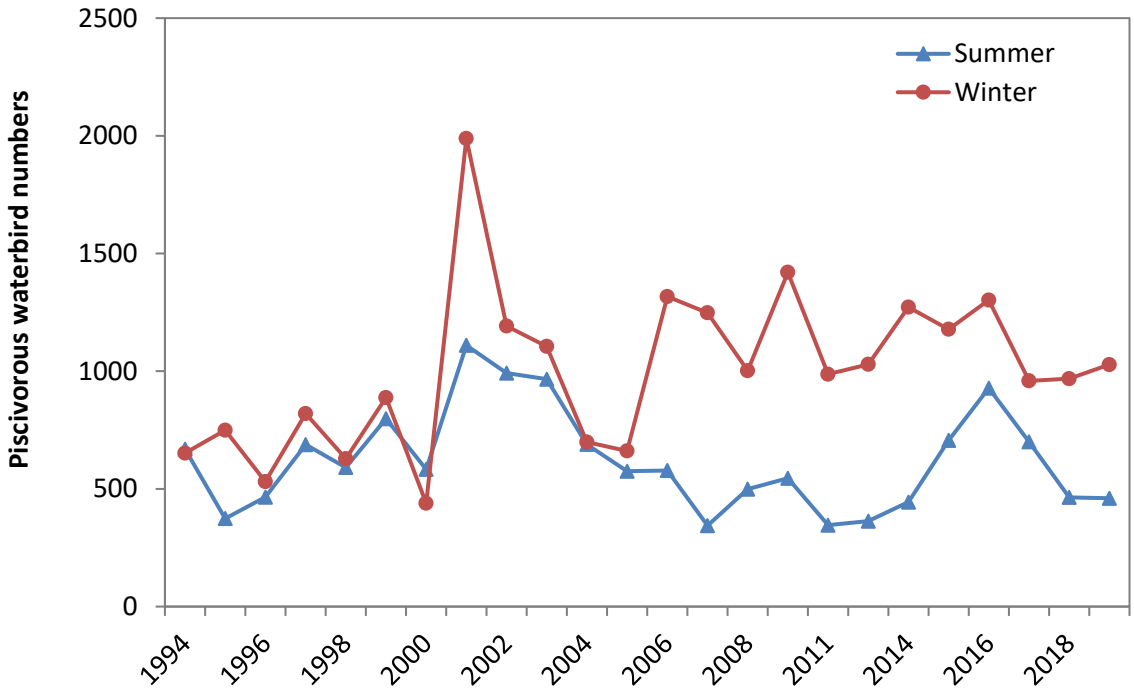

Figure 6.20. Long term trends in the numbers of piscivorous waterbirds on the estuary in summer (blue line) and winter (red line). Source: CWAC data.

Finally, long term trends in the numbers of planktivorous wading birds (Lesser Flamingos) on the estuary show a slight increase over time (Figure 6.21). However, this increasing trend is driven by the unusually high numbers of flamingos counted on the estuary during the drought of 2015-2018. In other dry years, the numbers of flamingos have also spiked, but not the extent that they did in the recent drought which saw on average 3200 birds on the estuary over this period. Numbers were down to 605 in 2019 (Figure 6.21). While hypersaline conditions are favoured by these birds, the higher flamingo numbers in drought years could also be due to the population aborting breeding in other areas, thus congregating on the Berg River Estuary in higher-than-normal numbers. Furthermore, Lesser and Greater Flamingos differ in their salinity response, with the Greater Flamingo moving into saltpans when salinity passes 40 and brine shrimp hatch and moving away from saltpans when salinity exceeds 140 and brine shrimp re-encyst (disappear). The Lesser Flamingo is hardier and is able to tolerate higher salinities, feeding on Cyanobacteria (blue-green algae). During the drought it is likely that the Greater Flamingos moved to the estuary from inland saltpans after the brine shrimp disappeared and the Lesser Flamingo moved to the estuary when the inland salt pans dried up completely.

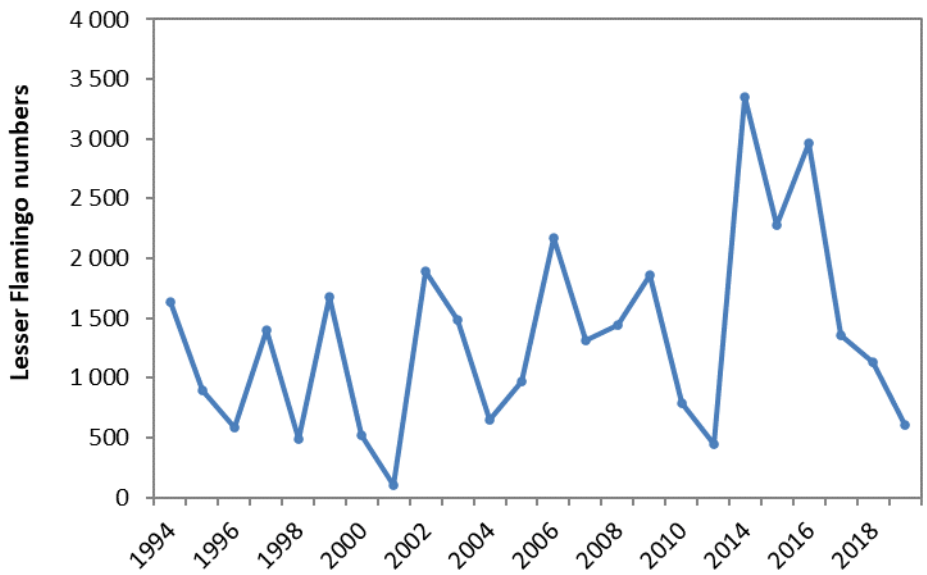

Figure 6.21. Long term trends in the numbers of planktivorous wading birds (Lesser Flamingos) on the estuary. Source: CWAC data.

6.6.7 The heronries

The Berg River Estuary is also home to some important heronries where certain waterbirds, such as herons, egrets, cormorants and darters, breed in large colonies. There are three known heronries on or in the vicinity of the estuary. The first is at the Cerebos salt works where Grey and Black-headed Herons breed off-river in alien gum trees, the second at Doornfontein in the middle reaches of the estuary, and the third at Kersefontein in the upper reaches of the estuary. The Kersefontein heronry, located in a reed marsh, has reportedly been in existence for about two hundred years. With very little human disturbance, this site has provided a safe environment for these birds to breed and their numbers can reach into the hundreds.

In the early 2000s, the Kersefontein heronry was mapped, and bird numbers and nests were counted. During this period hundreds of birds were counted and there was high species diversity. However, during the recent drought very little breeding activity occurred at the heronry. In 2017 it was reported that while there were some birds at the site (e.g. cormorants, darters, and Black-crowned Night Heron) there was no evidence of breeding, and in 2018 there were no birds reported at the site at all (G. Murison, pers. comm.). In 2019, while there was some bird activity, only a few Reed Cormorants, darters, Black-crowned Night Heron, Purple Heron and Goliath Heron were seen at the site (G. Murison, pers. comm.). With very limited data and only *ad hoc* surveys undertaken over the last 20 years, it is very difficult to make any definitive conclusions as to why the birds choose to settle at the site in particular years and what influences breeding success. However, it is very likely that the persistent reduced flows can have a significant impact on breeding productivity of approximately 13+ species at the Kersefontein heronry. It appears that when water levels are low birds do not settle at the heronry to breed. There could be a number of reasons for this, including die off of reed beds and changes in habitat structure, as well as higher salinities. However, the most plausible reason is that the lower water levels leave the heronries exposed to predation. Birds select sites for nest construction that will ultimately determine their breeding success - if water

levels are too low and their nests become exposed or easily accessed then they will not choose to settle.

There are very few long-term monitoring programs of heronries in Africa and very little is known about the importance of these sites in terms of (amongst others) abundance, species composition and breeding productivity (Harebottle 2019). A long-term monitoring program of the Kersefontein heronry would provide valuable information into the status of this breeding site and its productivity and could provide important insights into the link between bird numbers and breeding and freshwater flows into the Berg River Estuary. This is particularly relevant under future climate change impacts, as well as other conservation threats such as habitat loss and human disturbance (Harebottle 2019).

6.6.8 Summary of bird community response to freshwater flow reductions

Persistent low freshwater inflows and an increase in marine dominance of the system and the resulting changes in estuary habitats and productivity will lead to a change in bird community structure. Habitat changes include changes in water depths and salinity, changes in the extent of reed marsh and the level of inundation of the estuary floodplain.

Further reductions in freshwater flows into the estuary may result in further decreases in the number of herbivorous and omnivorous waterfowl, or the complete loss of certain saline intolerant species from the system. A drier estuary floodplain will reduce the attractiveness of the estuary for breeding waterfowl, as well as for wader species that frequent the drying pans after they have been inundated. Other species that are likely to be negatively affected include the species that are commonly associated with reedbeds, such as rallids and herons. The common reed *Phragmites* is intolerant to prolonged periods of high salinity, as evidenced by the die-off of reeds in parts of the estuary during the drought. Thus, lower freshwater inflows will likely see the loss of reed beds along the banks of the estuary which are favoured by these species. It is also quite likely that prolonged reductions in freshwater inflows would see the eventual permanent demise of the Kersefontein heronry.

Piscivorous species, on the other hand, may increase in numbers as a result of more marine fish in the estuary and the use of the estuary by these species as a haven during periods of drought. The increase in general productivity of the system as it becomes more marine dominated will likely result in an increase in the density of benthic invertebrates which could counteract some of the impact of loss of intertidal area as a result of lower water levels and decreased inundation of the floodplain.

Species that frequent the lower estuary, such as the migratory waders and flamingos are unlikely to be strongly affected by changes in freshwater inflows. Changes in community structure as a result of reduced freshwater inflows are more likely to be observed in the middle to upper reaches.

6.6.9 Health assessment and scenario analysis

By 2010, the overall health of the bird community was estimated to be 82% of Reference condition (based on the earliest bird count data from estuary in 1980). The estuary had experienced a loss in the degree of flooding relative to reference, which affected the attractiveness of the floodplain for breeding birds, and the length of time that birds were able to occupy the floodplain areas. Intertidal area was significantly reduced to less than two-thirds of original extent, affecting the most numerous group on the estuary, the waders.

Artificial saltpans increased suitable habitat for flamingos and other species. Areas of reed beds had decreased substantially, possibly having a negative effect on some skulking rallid and heron species (e.g. at the Kersefontein heronry). There had been no change in the general productivity of the system, but densities of benthic invertebrates were reported to have increased by 10%, counteracting the impact of loss of intertidal area, and numbers of fish had decreased slightly, to the detriment of piscivorous groups.

The overall health of the bird community in 2020 has declined markedly in the last 10 years, to an estimated 56% of reference condition. This is attributed primarily to a marked reduction in abundance of key avifaunal groups such as waders, wading birds and waterfowl as a result of habitat loss, disturbance and reductions in global breeding populations of migratory wader species. Indeed, birds that are experiencing stressed or less than optimal conditions tend to be more vulnerable to disturbance impacts.

Projected changes in species richness, biomass and community composition of estuarine birds under the various flow scenarios are summarised in Table 6.11. Conditions are expected to improve very slightly if summer low flow RQOs are respected (score improves to 57%) but will remain largely unchanged (56%) under the future development scenarios if summer low flow RQOs are not respected. Again, conditions are expected to improve slightly (57%) if the summer low flow RQOs are respected under future development. Further declines are anticipated when impacts of climate change are included (score drops to 50%) and respecting summer low flow RQOs is not expected to assist with this at all. Note that these scores do not take forward projections of migrant species populations into account.

Table 6.11. Summary of projected changes in species richness, biomass and community composition of birds under the various future flow scenarios (Hist. = Historical, P0 = Status Quo, EWR = Environmental Water Requirements , FD = Future Development, CC = Climate Change).

	Hist. 2010	PD 2020	P1	F0	F1	C0	C1
			PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
Species richness	82	78.9	78.9	75.7	78.9	63.3	63.3
Abundance/ Biomass	82	56.3	56.7	56.2	56.6	49.6	50.5
Community composition	84	76.5	76.6	75.9	76.4	65.6	66.1
Health score	**82**	**56.3**	**56.7**	**56.2**	**56.6**	**49.6**	**50.5**

7 VALUE OF THE BERG RIVER ESTUARY

7.1 Introduction

Estuaries are rich and productive systems that produce a wide range of benefits to society. They derive their richness and productivity from nutrient and sediment inputs received from river and sea water, combined with the relatively sheltered aquatic habitat that they provide. Their characteristic biodiversity assemblages have arisen from the need for biota to cope with their salinity gradients and fluctuations. These unique characteristics make estuaries among the most valuable types of ecosystems on earth (Costanza *et al.* 1997, 2014).

The Berg River Estuary is among the top three most important estuaries in the country (Turpie *et al.* 2012), and one of the most valuable biodiversity assets along the West Coast of South Africa. A range of estuarine ecosystem services support the local economy. The estuary has been specifically recognised as an Important Bird Area, has unique estuarine vegetation, is a valuable nursery area for fish and is an important tourist attraction. This assessment of the value of the Berg River Estuary is undertaken to guide development in such a way as to promote water and biodiversity management and conservation. The value estimates generated in this study build on earlier surveys conducted on the Berg River Estuary and more recent regional and national assessments. Any updates to value estimates are made from available information, expert opinion and stakeholder input.

This section provides a description of the main ecosystem services and benefits provided by the estuary (Table 7.1) and an estimate of their current values. This assessment builds on earlier surveys conducted at the estuary (Turpie *et al.*, DWA, 2010), as well as more recent estimates generated for regional and national studies, including the Berg River Resource Quality Objectives study (DWS, 2017). This study benefited from the input of a number of stakeholders. Informal meetings and discussions were held with tourism representatives, estate agents, community members, fishermen, farmers, property owners and various municipal employees.

Table 7.1. Summary of the main ecosystem services and their benefits

Broad category	Ecosystem service	Benefits
Provisioning	Fish production	Fisheries
	Floodplain vegetative production	Livestock grazing
Regulating	Nursery habitat	Support to inshore marine fisheries
	Carbon sequestration	Avoided climate change damages
Cultural	Suitable location for harbour development	Productivity as a result of harbour facilities
	Opportunity for deriving satisfaction through active or passive use of the system for spiritual, recreational, or educational purposes, etc.	Experiential value, manifest as tourism value, local recreational value, and property value premiums
	Opportunity for deriving satisfaction from knowledge of continued existence of estuary features	Non-use value (existence, bequest value)

7.2 Socio-economic context

The coastal town of Velddrif, situated on the banks of the lower estuary, is located within the Bergrivier Municipality. The Bergrivier municipal area is characterised farmlands and a 40 km coastline that is used by local fishers and is a popular tourist destination. As of 2019, the Bergrivier Muncipality had a population of 71 518 people in 17 878 households, and an expected population growth rate of 1.7% between 2020 and 2024 (Western Cape Government, MERO, 2019).

The predominant language in Velddrif is Afrikaans which is spoken by 83.5% of the population, followed by isiXhosa (8.7%) and English (5.8%). The coloured community makes up more than half of the population (57.1%) in Velddrif followed by White (29.6%) and Black African (12.2%) communities. As of 2019, the Bergrivier Municipality had the highest proportion of population aged 65 and older (5.4%) of all the municipalities in the West Coast District with many of the small towns, such as Velddrif, seen as a safe and tranquil place to retire to (Western Cape Government, MERO, 2019; Bergrivier Municipality, SDF, 2019). In 2017, the average household income in the Bergrivier Muncipality was R13 819 (Western Cape Government, MERO, 2019). While this was lower than other municipalities in the West Coast District, the Bergrivier municipal area had the lowest income disparity in 2018 (Western Cape Government, MERO, 2019).

In 2017, the size of the West Coast District's economy was R29.8 billion, contributing 5.2% to the economy of the Western Cape Province (Western Cape Government, MERO, 2019). Between 2008 and 2017, the West Coast District realised growth rates higher than that of the Provincial economy, growing at an average annual rate of 2.4% compared to the provincial average of 2% (Western Cape Government, MERO, 2019). The Bergrivier Municipality is the third largest local economy (14.9%) within the West Coast District, with regional gross domestic product amounting to R4.4 billion in 2017 (Western Cape Government, MERO, 2019). Between 2008 and 2017, the Bergrivier municipal area realised annual average growth rates of 2.8%, higher than that of both the District and Provincial economy (Western Cape Government, MERO, 2019). In 2017, 29 448 people were employed in the Bergrivier Municipality. Estimates for 2018 indicated a small positive Gross Domestic Product per Region (GDPR) growth of 0.1% during the year, but a net decline of 12 jobs (Western Cape Government, MERO, 2019).

The economy of the Bergrivier Municipality is driven by the agriculture, forestry and fishing sector and the manufacturing sector (Western Cape Government, MERO, 2019). Not only is the agriculture, forestry and fishing sector a leading contributor of GDPR, it is also the main contributor of employment in the region with 51% of all employment falling within this sector. The tertiary sector was the Bergrivier Municipality's largest contributor to GDPR in 2017, valued at R1.9 billion or 43% of the total (Table 7.2, Western Cape Government, MERO, 2019).

The main drivers of economic activity in the tertiary sector were the wholesale and retail trade, catering and accommodation (13%), finance, insurance, real estate and business services (10%), and general government (9%) sectors. While the tertiary sector was collectively the largest contributor to GDPR in the Bergrivier Municipality, the two sectors which contributed the largest proportion to the District's GDPR individually were the agriculture, forestry and fishing sector (29%) and the manufacturing sector (23%). Between 2008 and 2017, the agriculture, forestry and fishing sector realised average growth rates of 4.4% per annum (Table 7.2). Other sectors which experienced strong growth rates over the ten-year period were the finance, insurance, real estate and business services sector (3.7%) and the construction sector (3.6%). However, estimates for 2018 indicate that the agriculture,

forestry and fishing sector contracted by 2.9%, causing the local economy to stagnate with an estimated growth of only 0.1% registered for 2018 (Western Cape Government, MERO, 2019).

Table 7.2. Bergrivier Municipality GDPR and employment performance per sector, 2017. Source: Western Cape Government, MERO, 2019.

	GDPR			Employment		
Sector	R million value 2017	Trend 2008-2017	Real GDPR Growth 2018e	Number of jobs	Trend 2008-2017	Employ-ment (net change) 2018e
Primary sector	1 306.5	4.2	-3	15 040	-3 637	-212
Agriculture, forestry and fishing	1 277.7	4.4	-2.9	15 006	-3 620	-209
Mining and quarrying	28.7	-2.3	-6.2	34	-17	-3
Secondary sector	1 236.4	2.5	2.5	3 105	285	17
Manufacturing	1 007.7	2.6	3.3	2 263	131	8
Electricity, gas and water	68.8	-1.6	1.7	58	17	-3
Construction	159.9	3.6	-1.9	784	137	12
Tertiary sector	1891	2.2	1.1	11 303	2 410	183
Wholesale & retail trade, catering & accommodation	572.8	1.8	0.3	3 875	817	70
Transport, storage and communication	227.6	0.9	-0.2	452	141	5
Finance, insurance real estate and business services	445.2	3.7	3.1	1 846	588	68
General government	418.1	1.3	-0.7	2 479	77	-6
Community, social and personal services	227.4	3.2	2.1	2 651	787	46
Total	4 433.9	2.8	0.1	29 448	-942	-12

In 2018 the unemployment rate in the Bergrivier Municipality was estimated to be 5%. Between 2008 and 2018, the Bergrivier Municipality registered the lowest average unemployment rates over this period within the West Coast District, at 4.4% (Western Cape Government, MERO, 2019).

Within the small coastal town of Velddrif, the fishing and tourism industries are very important to the local economy. The Berg River Estuary, which contributes significantly to the local and regional economy, is considered a critically important ecosystem as local communities are dependent on the estuary for income and livelihood support. As part of the classification of and resource quality objectives for the water resources of the Berg Water Management Area the Berg WMA was divided into distinct socioeconomic zones, and within these zones, smaller integrated units of analysis (IUAs). The gross economic output of water related economic activities in the Berg River Estuary IUA, which included Velddrif and the areas immediately surrounding the estuary, was estimated to be R436 million in 2015, with inshore marine fisheries representing almost 80% of this value, followed by tourism and recreation. Irrigated crops,

which cover approximately 105 hectares, represented only a small portion of this overall value. It was estimated that marine inshore fisheries directly employ some 1450 people and indirectly some 1870 people, the largest employer in the Berg River Estuary IUA. The tourism industry directly employs close to 150 people and indirectly about 280 people (DWS, 2017).

7.3 Value of estuary ecosystem services

7.3.1 Subsistence fisheries

The fish resources of the Berg River Estuary support both recreational and subsistence fishing. While there is no longer any legal commercial fishing in the system, there is an illegal commercial gillnet fishery. This section focuses on the subsistence fishery (Box 7.1). Subsistence fishers are considered to be those who fish or collect bait personally, use low technology gear, live near to the resource and either use the catches to meet basic food requirements or sell the catches locally to gain income to allow them to meet basic food requirements. The subsistence fishery on the Berg River Estuary includes both line fishing and bait collection. Subsistence fish catches tend to be dominated by a few species, namely harder, carp and elf (Hutchings *et al.* 2008).

Estimating the total number of "true" subsistence fishers on the estuary and the value of their catch is difficult. Information on subsistence fishing effort on the Berg River Estuary is available from two sources. Turpie & Clark, Cape Nature (2007) estimated the annual catches and values for subsistence fisheries in South African estuaries using data collected as part of the Subsistence Fisheries Task Group assessment (Clark *et al.* 2002). Estimates of the total number of subsistence fishers in the area were taken from Clark *et al.* (2002), while an estimate of the value of the annual subsistence catch from the estuary was derived by multiplying the average catch per resource (invertebrates and fish) caught per fisher per year by an estimate of the value for each as given by the fishermen themselves. The second source of information was a study by Hutchings *et al.* (2008) involving a survey of subsistence line fishing effort on the estuary over the period 2002-2005 (Table 7.3). The catch value was estimated by multiplying the average mass of each species by a protein replacement value of R34/kg.

Recent data on subsistence catches in the Berg River Estuary are not available. However, if we assume that catches have remained similar, the subsistence value of the Berg River Estuary would be in the order of **R385 500 – R1.2 million** per year (in 2019 Rands). The lower bound estimate represents the value derived from the data collected by Hutchings *et al.* (2008) who did not consider net fish catch or invertebrate catches from the estuary and the upper bound is based on estimations made in the report to Cape Nature - Turpie & Clark (2007).

Note that the estimates of values for subsistence fishing do not necessarily represent the **sustainable yield** that can be harvested without causing detriment to the system. There are, however, not many good estimates on what the sustainable yield is for the species targeted by subsistence fishers.

Box 7.1. The Berg River Estuary provides a value resource to subsistence fishers. Photo: R. Robinson

There are a significant number of local fishermen that catch fish from the Berg River Estuary for subsistence purposes. During informal discussions with these fishermen on the banks of the estuary they described the importance of the resource to them in feeding their families. They also noted the amount of pleasure that they derive from the activity of fishing.

"I catch fish every day!"

– Subsistence fisherman, Velddrifc

The subsistence fishers of Velddrif described how the availability and types of fish in the estuary have changed over the last few years. They noted a general improvement in catch, indicating that they are satisfied with what is currently available. They explained that the types of fish species they are catching have, in the last three years, changed from more freshwater fish to catches being dominated by saltwater fish. An important issue raised by all fishing groups during discussions was restricted access to fishing sites. There are limited areas where people can access the estuary to fish. There are areas upstream that used to be accessible, but some farmers are now charging for access to these fishing spots.

"We can't easily catch fish anymore. That man erected a wire fence and now you must pay for access…it was much easier in the past, you caught lekker fish over there"

– Subsistence fisherman, Velddrif

Table 7.3. Catch-per-unit-effort (fish angler^{-1}hour^{-1}) and total annual catch by species for subsistence fishers ocn the Berg River Estuary, December 2002-November 2005 (from Hutchings *et al.* 2008) and the estimated value per fish (in 2019 Rands– based on the Consumer Price Index (CPI), Stats SA).

Species	Catch-per-unit-effort	Annual catch (numbers)	Value per fish (Rands)
Elf	1.158	7 846	10.00
Harder	1.711	11 237	10.00
Carp	0.296	2 688	68.00
Barbel	0.017	158	51.00
White stump	0.017	109	14.00
Gurnard	0.002	13	17.00
Other species	0.006	48	10.00
All species	3.207	22 100	

7.3.2 Livestock grazing

The farms that border the estuary on both the northern and southern bank graze livestock in the estuary floodplain. The floodplain area provides valuable grazing for both cattle and sheep. The livestock are moved into the floodplain around September/October and are taken out in January/February. Not all of the stock graze in the floodplain at one time, with numbers fluctuating consistently. Each year the grazing capacity of the floodplain is assessed to determine the total number of animals that can be supported.

Over the last two decades the carrying capacity of the floodplain has decreased significantly as shown by the analysis of enhanced vegetation index (EVI) data on vegetation productivity for the period 2000-2020 (see section 6.3 on plant communities). This is due to decreases in the extent and duration of floods in the catchment, which is the result of both changing climatic conditions and extractions upstream that have reduced the flow rate into the estuary. Without consistent flooding of the floodplain each year, the area of vegetation suitable for grazing is reduced. It was reported during an interview with a farmer that they have been reducing stock numbers for over two decades due to the loss in grazing capacity. Because there is less flooding of the floodplain, there is a shorter grazing period. Without sufficient rains and decent flooding, the floodplain becomes saltier and grazing fodder decreases. Over the last two decades livestock farmers along the estuary have reduced their stock numbers by between 30-60% as a result of decline in the floodplain footprint.

During the recent drought the floodplain was not used for feeding livestock. Farmers were forced to offload animals by a third and bring in feed for the remaining stock. This was an incredibly expensive exercise. Given the unpredictable climatic conditions and the poor seasonal flooding of the estuary, floodplain farming has declined significantly in the area.

The value of livestock was estimated based on information provided by local farmers and statistics taken from the census of commercial agriculture (Stats SA 2007). There are thirteen farms along the estuary that farm cattle and sheep in the floodplain. Using land cover data for the Berg River Estuary and its floodplain the area of suitable grazing vegetation within the Estuary Functional Zone (NBA 2019, primarily defined by the 5 m contour) was estimated and used to determine the number of cattle and sheep supported by floodplain grazing. This was based on the assumption of 5 ha per head of cattle and 1 ha per head of sheep. Average annual offtake rates were taken from the census of commercial agriculture (33% for sheep and 45% for cattle) and prices were supplied by local farmers (~R1500/sheep, R10 000/head of cattle). Financial data from the census of commercial agriculture were used to get an estimate of farm expenditure as a proportion of gross farm income (72%), in order to estimate direct value added. Direct value added is the gross income from livestock production and livestock products less any costs of production, excluding labour costs.

Based on the above calculation, the total direct value added of livestock production associated with the estuary floodplain is estimated to be in the region of **R11.5 million** per year.

Figure 7.1. A view of the middle reaches of the estuary showing a farm house on the southern bank. Photo: C.F. Clark.

7.3.3 Salt production

Salt production is not strictly an estuary ecosystem service, but a benefit obtained from converting estuary salt marshes to productive salt pans. Furthermore, this activity is one that would potentially benefit from reduced freshwater flows to the estuary, since the estuary water on which it depends would be saltier. Since salt production is an integral part of the sense of place of the lower Berg River Estuary and surrounds, it is included here.

Salt production is practiced in three places on the south banks of the Berg River Estuary – immediately adjacent to the Carinus road bridge at Swaartjiesbaai, Flaminkvlei on the eastern side of the road bridge, and on the farm Kliphoek approximately 6 km upstream from the road bridge. The salt production in the estuary does not represent a service of the estuary per se, but rather the saltworks occupy former saltmarsh areas and rely mainly on pumped saline water from the estuary on the incoming tide. The salt pans attract numerous important waterbirds such as migratory and resident waders, and Greater and Lesser Flamingos. The birdlife on these pans attracts bird watchers from across the country.

The large National Salt (Ltd) (Cerebos) works were established in 1969 at Swaartjiesbaai with a long-term lease of the land. The salt ponds were further expanded to the area on the eastern side of the Carinus Bridge (called Flaminkvlei) in 1984. However, Cerebos was not the first to produce salt at this site. For decades before this, salt was produced informally at this site for the salting of bokkoms. The saltworks on Kliphoek farm started to operate in 1986. This is a much smaller operation than that of Cerebos and the focus is only on the production of raw salt (no processing or refining).

Salt production is weather dependent with the wind and sun being the two critical elements.

Therefore, production is limited to the summer months, starting in September/October and running until May of the following year. Harvesting generally takes place only once a year. At Cerebos, water is either pumped from the estuary or directly from the sea. At Kliphoek, water is pumped directly from the estuary on the incoming tide when the salt saturation is high enough using a gravitational system which feeds a large storage dam.

Figure 7.2. Salt pans adjacent to the estuary as you drive into Velddrif. Photos: J.Turpie.

Together the three production sites on the estuary produce approximately 55 000 tonnes of salt per year. Kliphoek produces roughly 15 000 tonnes and Cerebos the remainder (40 000 tonnes). Kliphoek Saltworks employs four permanent workers. Until recently, Cerebos employed 60 permanent workers but only two staff now remain on site. Salt production at the two Cerebos sites has stopped and the future of these saltworks remains uncertain. However, Kliphoek Saltworks has been bought by Royal Salt Works (owned by Donald Brown Group of Companies - one of the largest salt producers in Southern Africa) who are looking to expand operations and production in the area. The reason for Kliphoek Saltworks selling their operation is due to the tight regulations that restricted operations and made it very difficult for a small producer to operate. Raw salt is not sold for very high prices. The money in salt production falls within the value chain of processing and refining the salt to then be sold at market. Raw salt from Kliphoek Saltworks was sold at R150 per tonne. Therefore, the production value of raw salt from the saltworks along the estuary is approximately **R8.3 million** per year. This does not include any value added that would result from processing and refining the salt and does not include the costs of production. Although these were reported to be relatively small. The raw salt is not just used to produce table salt but is also used in the

agricultural sector for wildlife and livestock licks and for feed lots, as well as in abattoirs.

The recent drought of 2015-2017 had a positive impact on salt production due to less rainfall and less freshwater flow into the estuary. Kliphoek Saltworks experienced a 25% rise in salt production over this period. This equates to an additional 3750 tonnes of raw salt per year, worth just over R500 000.

7.3.4 Fish nursery habitat

Estuaries play an important role as nursery areas for many fish and invertebrate species that spend the rest of their life cycle in marine habitats, including many species that are harvested for recreational or commercial purposes (Whitfield 1994; Beck *et al.* 2001). The quantity and quality of freshwater inflows to estuaries as well as management of habitats within them affect their capacity to function as nursery areas. The Berg River Estuary is a very important nursery area for inshore marine fish along the West Coast. The affected fisheries in this region are primarily the recreational line fisheries, inshore commercial line and net fisheries, and inshore and estuarine subsistence fisheries (Box 7.2).

Fish species that use estuaries have been classified according to their relationships with estuaries, as described in the previous chapter. About 160 species of fish occur in South African estuaries, of which about 80 species are utilised in fisheries. Some 16 of these species occur in the Berg River Estuary. Of particular importance are the category II species, for which management of estuaries plays a crucial role in fisheries (Lamberth & Turpie 2003). Most estuary-dependent fish species enter estuaries as larvae or post larvae (Whitfield & Marais 1999) and once the estuarine dependent phase is complete, they leave the estuaries for the marine environment where they become available to marine fisheries, and upon maturity contribute to the spawning stock (Wallace 1975a,b).

Fish diversity and abundance differs between estuaries of different sizes and types, with higher species richness associated with larger and permanently open systems (Lamberth & Turpie 2003), such as the Berg River Estuary. However, estuary health, in particular, the quality and quantity of water entering an estuary can have a significant impact on fish abundance and diversity.

Estimates of nursery value have been made for all South African estuaries, based on inshore fishery catches and the level of dependence of each species on estuaries (Lamberth & Turpie 2003). Based on this 2003 study, it was estimated that the 50 or so estuaries in the Western Cape contribute about R250 million to the value of inshore fisheries (in 2013 Rands). However, many of the estuaries in the Western Cape have become degraded as nursery habitats because of altered flow and mouth manipulation. Furthermore, the fish stocks themselves have also been depleted through overfishing, both legal and illegal. Based on the scores given in a recent evaluation of the current health of fish stocks in some of South Africa's temperate estuaries (WRC and CSIR, unpublished data), in conjunction with information on estuary volumes, it has been estimated that the nursery outputs from estuaries in the Western Cape are now only about 27% of their original capacity, suggesting that services to the value of R675 million have been lost (Turpie *et al.* 2014). Indeed, escalating illegal gill net fishing (poaching) is responsible for more than half the catch in South Africa's important estuary fish nursery area, resulting in recruitment and growth overfishing and collapse of economically and socially important species such as harders and dusky kob (Van Niekerk *et al.* 2018). The most severely impacted are small scale coastal fisheries (Van

Niekerk *et al.* 2018).

The small-scale commercial inshore gillnet fishery that operates in St Helena Bay and targets harder (southern mullet, *Chelon richardsonii*) is the most important fishery in the study area. The freshly caught harder are brought from the sea to the traditional processing huts of Bokkomlaan adjacent to the Berg River Estuary where they are salted and hung out to dry in the warm Berg wind (Figure 7.3). Bokkoms (the salted and dried harder) are the most iconic food of the area, are an invaluable part of the cultural heritage in Velddrif and still make up the predominant source of protein for most families. Harders were once sold as bait fish to the Japanese bluefin tuna longline, but this was short-lived as the industry was not able to process and freeze the catch fast enough (S.J. Lamberth, pers. comm.).

Box 7.2. Commercial fisheries, livelihoods and cultural connections. Photo: R. Robinson

The sustainability of the commercial fisheries associated with the Berg River Estuary are of critical importance to the people of the region and their way of life. There is a long tradition and culture of commercial fishing, which strongly emerged in discussions with commercial fisherman based at Laaiplek. They made clear links to a shared lifestyle with other small fishing towns like St Helena Bay and Elands Bay and describe a deep love of the sea, attachment to this place and to the lifestyle enabled through the harder fishery.

"When you are catching from a young age you are learning from your parents, your father, grandfather, uncle. All of them were fisherman, they all grew up in this town and we have learnt from them. I have been fishing on these boats since I was 17 & I am now 34. It's a long time, but I will never leave the sea, I cannot stay away from the sea"

– Commercial fisherman, Laaiplek

"Fishing is in our blood. We all grew up in this place. Catching is our bread and our butter, no fish – no pay. This is my life, death and retirement work"

– Commercial fisherman, Laaiplek

However, this legal fishery is threatened by illegal gillnet fishing (poaching) on the Berg River Estuary (Horton, NBA, 2018). The Berg River Estuary gillnet fishery was closed in 2001 and following its closure the legally operated small-scale commercial gillnet fishers reported an increase in the numbers and size of harder being caught in St Helena Bay. However, this recovery was short lived. Monitoring has shown that illegal gillnet fishing in the estuary, most of which takes place at night, has intensified with approximately 400 tons of fish being caught per annum by 2018 (Horton, NBA, 2018; Horton *et al.* 2019). With the rise in illegal catches on the estuary, there has been a subsequent decline in the total reported catches by the legal commercial inshore fishery from 1500 tons of harders in 2006 to 400-500 tons in 2018. This decline in legal catches is directly attributable to the 400 tons of harder poached from the estuary. This gives a multiplication factor of 2.5 less harders in the ocean if illegal gill

net fishing in the estuary continues at present levels. Illegal fishing on the estuary is threatening the viability of the legal inshore fishery with only 28 of the 84 gillnet and beach-seine right-holders in St Helena Bay remaining active in the fishery in 2019 (Horton, NBA, 2018).

Figure 7.3. Freshly caught harder are brought from the sea to the traditional processing huts of Bokkomlaan where they are salted and hung out to dry in the warm Berg wind. Photos: R. Robinson

Figure 7.4. A bokkom producer in a processing hut in Bokkomlaan, Velddrif. Photo: R. Robinson

The nursery value of the Berg River Estuary was estimated based on fishery values from Lamberth & Turpie (2003) but adjusted for changes in effort in the inshore seine and gill net fishery. Without recent data on recreational fishers along the West Coast, we assume that effort in the recreational line fisheries has not changed. Effort in the inshore seine and gillnet fishery was adjusted based on the most recent data of catches from St Helena Bay. Based on this update, it is estimated that the nursery value of the Berg River Estuary is **R8.7 million** in 2019, suggesting that nursery services to the value of R2 million have been lost since Lamberth & Turpie (2003) first estimated the value in 2003.

7.3.5 Harbour facilities

In addition to the above, the estuary provides a sheltered harbour that supports offshore commercial fisheries and also enhances the recreational value of the area. There are relatively few sheltered harbours on the West Coast and the presence of this harbour is thus of particular significance on a regional scale. At the harbour there is a large fish factory that processes pelagic (deep sea) fish for canning and fish meal where the pelagic fishing boats offload their catches directly. The value of the harbour can thus be captured through the value associated with the catching and processing of pelagic fish. Without the estuary, the pelagic boats would not be able to land their fish directly at the factory and the factory would incur costs in transporting fish catches.

Figure 7.5. The sheltered Laaiplek harbour supports commercial fisheries and also enhances the recreational value of the area. Photo. R. Robinson.

Based on data from Hutchings *et al.* (2015) who assessed the small pelagics purse-seine (weighted nets) fishery off South Africa, we estimated the revenue associated with the Laaiplek fish factory. Taking into account the reduction in sardine Total Allowable Catch from 90 000 tonnes to 15 000 tonnes, due to the reduction in sardine stocks, and adjusting for inflation, it is estimated that the Laaiplek harbour supports the wholesale production of sardine, anchovies and other industrial fish to the value of **R107 million**.

During discussions with local fishermen, they noted the importance of fishing to livelihoods and they specifically identified the role of the fish factory in Laaiplek in providing jobs and being a cornerstone of the economy.

7.3.6 Carbon sequestration and storage

Estuarine habitats are highly productive systems that have the capacity to sequester carbon

at a rapid rate (Barbier *et al.* 2011, Beaumont *et al.* 2014). The relative carbon storage potential of estuarine habitats is now widely acknowledged and considered significant in the regulation of both local and global climate (Beaumont *et al.* 2014, Sidder 2018, Adams *et al.* 2019). Recent research by Krauss *et al.* (2018) has shown that even upper estuary habitats, such as tidal freshwater forested wetlands and low-salinity marshes, store significant amounts of carbon, in some instances even exceeding those of seagrass and salt marsh ecosystems. Until recently, in South Africa, the role of estuarine habitats as a source and sink of greenhouse gases has been comparatively unknown. The recent report by Adams *et al.* (2019) assesses blue carbon stored by South Africa's estuarine habitats and quantifies the extent and loss of these habitats and their ecosystem services.

The term "blue carbon" is used to describe the carbon found in three major coastal and marine ecosystems: mangroves, seagrasses, and salt marshes (Siikamäki *et al.* 2012). These habitats sequester carbon from the atmosphere and lock it into the soil where it can stay for millennia. These habitats are unique in that the carbon that they sequester during photosynthesis is often moved from the short-term carbon cycle (10-100 years) to the long-term carbon cycle (1000 years) and is continuously buried as slowly decaying biomass (Barbier *et al.* 2011). However, many of these coastal habitats are threatened, thereby reducing their capacity to provide ecosystem benefits. Drivers of this degradation include urban and industrial development, aquaculture, agriculture, tourism, forestry, coastal erosion and sea level rise (Beaumont *et al.* 2014). When these habitats are degraded, they emit large amounts of CO_2 into the atmosphere contributing to global climate change with impacts on biodiversity, water supply, drought and floods, agriculture and human health. Therefore, restoration or protection of these ecosystems is important for reducing ongoing losses and rebuilding carbon stores, i.e. mitigating or avoiding these damages, respectively. On the Berg River Estuary, erosion is a problem along some shoreline areas contributing to damage and loss of salt marsh areas. Furthermore, the reduction in freshwater flows into the estuary is having a negative impact on these habitats.

The Berg River Estuary contains 4408 ha of salt marsh and 206 ha of submerged macrophytes (Adams 2018). This represents 30% of the total area of salt marsh habitat in South African estuaries and 8% of submerged macrophyte area. Based on estimates of carbon stocks for blue carbon ecosystems from Adams *et al.* (2019), the total estuarine ecosystem carbon in the Berg River Estuary was estimated to be 1.2 Tg C (teragrams of carbon where one Tg = 1 million metric tonnes), which equates to approximately 4.2 Tg CO_2 (using molecular weight of CO_2/molecular weight of carbon; US Environmental Protection Agency, 2016).

Estimates of the global social cost of carbon vary greatly; the most recent estimate placed the social cost of carbon at US$417 per ton of CO2 (in 2018 US$; Ricke *et al.* 2018). The cost that would occur to South Africa would be based on the country-level social cost of carbon, which "captures the amount of marginal damage expected to occur in an individual country as a consequence of additional CO2 emission" (Ricke *et al.* 2018). South Africa's social cost of carbon is estimated to be US$3.31 per ton of CO2. Based on this, the avoided degradation and loss of 1.2 Tg C of the Berg River Estuary habitats represents avoided damages of **R1.6 billion per annum** at a global scale, and the damage costs to South Africa resulting from a loss of these carbon stocks nationally would be **R12.4 million per annum**.

Figure 7.6. The Berg River Estuary contains a significant area of salt marsh and submerged macrophyte habitat which play an important role in sequestering carbon. Pictured are saltmarshes dominated by *Sarcocornia* in the foreground. Photos: Jane Turpie.

7.3.7 Experiential values

Estuaries are a dominant feature of many coastal resort areas along the South African coastline. The attractions of estuaries as recreational areas are many, and include their aesthetic beauty, opportunities for water sports, birding and fishing. These contribute to the attractiveness of the area as a place to visit or to invest in property. This study focuses on the contribution of the estuary to tourism and property values. We recognise the importance of the Berg River Estuary to local recreational value but only report on this in qualitative terms.

Tourism value

The Berg River Estuary is a popular West Coast tourist destination and tourism is recognised as one of the major contributors to the regional economy. The estuary provides opportunities for fishing, bird watching, boating and appreciation of scenery and nature with walking trails and leisurely lunch spots along the banks of the estuary. The site is a national and international birdwatching destination. Birding is important not only for tourism but also providing opportunities for education (Fincham *et al.* 2018). Flocks of flamingos can usually be seen in the salt pans from the Carinus road bridge as one arrives in Velddrif, pelicans and herons are frequent visitors to Bokkomlaan where harders are processed and hung out to dry, and a wide variety of waders can be seen feeding on the expansive mud flats of the estuary floodplain. Recreational fishing is also an important activity on the estuary and along the coast. Sailing and canoe events such as the West Coast Canoe Challenge and the Berg River Canoe Marathon are important events in the tourism calendar for the area, and triathlons and swimming races (the Berg Mile) are developing sports.

Tourism has changed considerably over the last decade with online platforms such as Airbnb having changed the way people travel and experience travel destinations and having encouraged tourism outside of traditional tourist areas. As such, not only have the numbers of tourists visiting smaller towns increased, but the type of tourist has also changed with more international visitors being attracted to areas outside of the major tourism hubs. Velddrif is a

popular destination for domestic tourists to enjoy estuary-based activities and to relax away from the hustle and bustle of Cape Town, but has more recently experienced a growing regional and international tourist market who are attracted by cultural and heritage and nature-based experiences that are unique to this coastal fishing town. For example, the age-old tradition of catching, processing and drying of bokkoms on Bokkomlaan adjacent to the estuary is a drawcard. The tourism industry in Velddrif/Laaiplek has increased significantly over the last few years with tourism becoming less seasonal, whereas it used to be mainly in in December and over long weekends (pers. comm. C. Ellis Velddrif Tourism). Overseas visitors are mostly from the United Kingdom and Germany and their numbers remain relatively constant throughout the year (Western Cape Government, MERO, 2019). Domestic visitors are mostly from the Western Cape and Gauteng. The estuary is considered the most important attraction for visitors in the area. Visitors love being on or close to the estuary, or on the canals at Port Owen, whether it be for fishing, bird watching or just relaxing.

Two estimates have been made of the tourism value of the estuary, one based on local survey data (Turpie *et al.*, DWA, 2010), and the other on provincial statistics and spatial data (DWA 2017a).

Turpie *et al.*, DWA, (2010) estimated the tourism value associated with the Berg River Estuary using a questionnaire survey that was targeted at permanent residents, holiday home owners and visitors. They found that the coastal area (coast, beach and ocean) contributed about 30% of people's enjoyment of the Velddrif area, while the estuary alone contributes more than one third (35%), or as much as the other five attractions and amenities combined (Figure 7.7). At the time of the survey, relaxing, walking and swimming were the most important activities carried out on the estuary (35%) followed by fishing (19%), bird watching (15%) and boating (14%) which are all potentially affected by changes in flow of freshwater into the estuary (Figure 7.7). A tourism survey conducted by Wesgro in 2014 indicate similar findings in terms of what tourists are doing and enjoying when visiting Velddrif (Rumbelow 2016). Although there were fewer than 100 survey responses for Velddrif, these data complement those of Turpie *et al.*, DWA, (2010). When asked to select their three main tourist activities for Velddrif the following was recorded: Scenic Drives (65%); Birding (13%), Culture/Heritage/Museums/Township Tours (11%), Gourmet Restaurants/Cuisine (10%), Meetings/Incentives/Conventions (10%), Beaches (10%), Golf (10%), Crafts/food markets/slow markets (6%), Gambling (6%), Whale watching (4%), Fishing (3%), Wine Tasting (1%), Sports event/spectator (3%), Sports event participant (3%), Shopping (3%), National Parks (1%), Heath Wellness/Spa's (1%), Adventure (1%), Flowers (1%).

The survey by Turpie *et al.*, DWA, (2010) found that a high proportion of households have boats, with more than 50% of residents and holiday home owners using boats (Table 7.4), reflecting the high level of use of the estuary. Most of the boats are used either on the estuary or launched from the estuary.

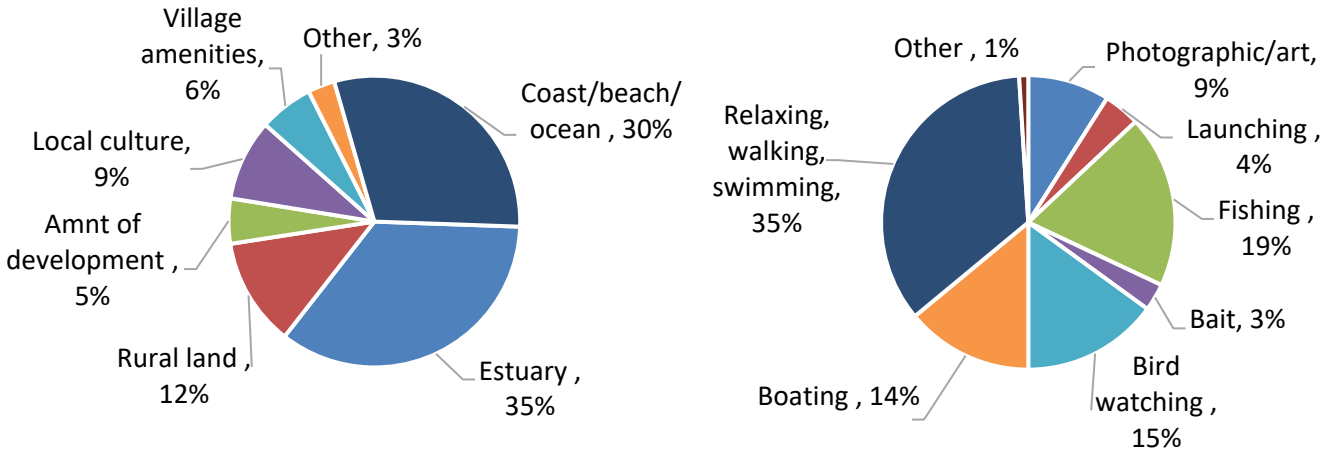

Figure 7.7. (a) Average percentage contribution of different amenities to enjoyment of the area and (b) average percentage contribution of different activities to enjoyment of the Berg River Estuary. Source: Turpie *et al.*, DWA, (2010).

Table 7.4. Average number of boat days per year and the percentage of households/groups with boats. Source: Turpie *et al.*, DWA, (2010).

User type	Average boat days per year		Percentage households/groups with boats	
	Non-powered	Powered	Non-powered	Powered
Residents	61.4	37.9	16.2	42.7
Holiday homeowners	43.4	86.2	24.5	65.3
Visitors	2.2	16.4	8.5	17.0

Indeed, the use of boats on the estuary has increased significantly over the last few years with reports that over the festive season (December/January 2019/2020) more than 200 boats were counted on the estuary at any one time. Not only is the excessive motorised boating on the estuary a public safety concern at times, it can also have a negative impact on estuarine habitats and biota. For example, shoreline erosion at various points appears to be related to wave action, possibly caused by motorised boats. Other impacts include disturbance of fish and birds and the degradation of vegetated areas. In September 2019 the Bergrivier Municipality introduced boat licensing on the estuary where all privately owned and commercial vessels, as well as canoes, paddle-skis and self-propelled craft, are required to purchase a boat license before going on the estuary. There are 30-day permits and annual permits, with the price of the permit increasing for larger boats or boats with higher engine capacities. The majority of the boat permits sold are under the category of 'annual pleasure' with a total of 19 commercial permits and 330 pleasure permits being sold over the four-month period (Table 7.5). More than 100 pleasure permits were sold in December alone.

Based on the data collected during the survey on visitor expenditure and estimates of visitor days, Turpie *et al.*, DWA, (2010) estimated that the total expenditure by holiday homeowners

and visitors to Velddrif was R88 million per annum with **R31 million** of this visitor expenditure being attributable to the estuary (converted to 2019 Rands using CPI table, Stats SA).

Table 7.5. Numbers of commercial and pleasure boat permits sold from September to December 2019. Source: Bergrivier Municipality.

Month	Commercial Annual	Commercial 30 Days	Pleasure Annual	Pleasure 30 Days
Sep-19	0	0	80	0
Oct-19	1	1	55	3
Nov-19	1	0	56	3
Dec-19	15	1	116	17

Over the past five years, social media tools have provided an alternative means of estimating tourism value, but analysing the densities of georeferenced photographs taken by the public and uploaded onto the internet, in conjunction with reliable regional statistics. In South Africa, this approach has been used at a national level (Turpie *et al.,* Ecosystem Services*,* 2017), provincial level (Turpie *et al.,* UN Environment Programme, 2020) and municipal level (Turpie *et al.*, World Bank Report, 2017), and was also used to value estuaries as part of the Berg Water Management Area Classification and Resource Quality Objectives Study (DWS, 2017). The latter study estimated that the Berg River Estuary accounted for **R36 million** in annual tourism expenditure (in 2019 Rands). This is comparable to the estimate of R31 million described above.

As with the rest of the Western Cape, at the height of the drought (~2017) visitor numbers to Velddrif declined, in response to extensive and intensive water restrictions. However, tourism establishments reported having an excellent season over December/January 2019/2020 with increases in occupancy rates of 30%, relative to 2015-2018. New tourism related businesses are popping up across the study area and existing tourism establishments are diversifying and expanding.

Given that tourism is strongly linked to the estuary, it would be reasonable to expect that tourists would be affected by estuary health to some degree. However, given the continuous rise in tourist numbers to the estuary over the last few years (a period over which the health of the estuary has deteriorated), it is clear that the overall health of the system may not be a limiting factor. This could be largely an artefact of history, in that resort towns that continue to attract people would have developed when the estuaries were in a good condition. The towns themselves are now a major part of the attraction and continue to draw visitors despite some changes in estuary condition. It is also possible that people only become sensitive to deterioration in estuary health beyond some threshold when the changes become significant and noticeable. It should be acknowledged here that with the presence of social media and the potential for ethical environmental publishing in the future, some consumers may choose to not visit the area, or take part in sporting events, if it becomes apparent that their presence is adding to environmental damage to the estuary.

Property value

The value that residents place on natural ecosystems is reflected, to an extent, in private property and real estate markets. When buying a home, certain preferences for different

characteristics are revealed through the amount that each homebuyer is willing to pay for the property, with homes that have a higher number of desirable characteristics usually selling for a higher price. Property attributes include the physical characteristics of the home such as size of the property, number of bathrooms and condition, neighbourhood characteristics such as access to amenities, and environmental characteristics such as scenic views and the amount of natural area surrounding a property. Therefore, if residents do value natural systems then it would be expected that these values should be revealed in property prices. This is particularly true for estuaries. People do not only visit pleasant environments, they may also pay a premium to live near them. This is as additional amenity value that is not reflected in tourism expenditure.

Development around the Berg River Estuary comprises four main suburbs – Velddrif, Laaiplek, Port Owen and Noordhoek (Figure 7.8). Laaiplek is situated near the mouth of the estuary, closest to the beach. Port Owen is situated on the estuary with a large proportion of properties located on the marina system. Velddrif is east of Laaiplek and Port Owen, further from the mouth but adjacent to the estuary. North of Velddrif is Noordhoek, the suburb furthest from the estuary. The south bank of the estuary, to the west of the Carinus road bridge, remains relatively undeveloped with all residential and commercial facilities located on the northern side of the estuary.

Figure 7.8. The three main suburbs in the town of Velddrif. Source: Google Earth, February 2020

The Velddrif/Laaiplek property market is strongly linked to the estuary. Property owners have a sentimental attachment to the estuary and properties that are located on the estuary demand higher premiums. A local estate agent reported that while there has been a regression in property sales country wide, Velddrif is experiencing growth in the property industry. Over the last two years, interest has grown significantly. Unsurprisingly, the property

market is also linked to the tourism sector. When tourism is booming then there tends to be spill over into the property market as more people become attracted to the area and the opportunities it provides.

There is large variation in property price across the Velddrif/Laaiplek town area. Much of this variation is linked to distance from the estuary. Being such a small town, the distance to neighbourhood amenities such as schools, shops and health care does not have a large bearing on house prices and people choose to purchase property based on its position in relation to the estuary. The local estate agent reported that there is always a continuous stream of interest for properties situated on the estuary. The town has become popular with retirees who are searching for a lifestyle that is focused on peacefulness and tranquillity.

Property value associated with environmental assets is generally estimated using the Hedonic Pricing Method, a form of multiple regression analysis, where property price is estimated as a function of structural variables, neighbourhood variables and environmental variable to isolate the influence of the latter (Rosen 1974). Structural variables include the size of the property, number of bedrooms, number of bathrooms and condition of the property. Neighbourhood variables include access to amenities such as schools, shops and hospitals. Environmental variables include the amount of or distance to green open space areas, access to parks, proportion of trees in the neighbourhood.

Turpie *et al.*, DWA, (2010) generated a hedonic pricing model using information collected from a survey of homeowners in Velddrif in 2009 as well as information obtained from Google Earth. They found that houses with canal and estuary frontage were worth about R1.7 million and R1.6 million on average respectively, more than double the average price of properties with no water frontage or view (Turpie *et al.* 2010). Furthermore, there was a significant positive correlation between house price and distance to the estuary. The total property premium associated with proximity to the Berg River Estuary was estimated to be just under R900 million, which translated into an estimated annual turnover of about R49 million in the financial and property sectors (2009 Rands).

In this study we conducted an empirical hedonic analysis using data on property size, location and value for 3518 properties from the Bergriver Municipal valuation roll. We estimated the Euclidean or straight line distance of each property to the Berg River Estuary. Property prices vary considerably across suburbs and within suburbs (Table 7.6). Properties in the suburb of Port Owen and Admiralty Island have the highest average property price of just over R3 million, while the lowest prices are associated with Laaiplek and Noordhoek. As expected, there was a significant positive correlation between house price and distance to the estuary (Figure 7.9). In the hedonic model, a logarithmic function was used to account for the expected decline in effect of distance from the estuary on price. The property model was highly significant with a good model fit ($n=4218$, $R^2=0.77$, $P<0.001$; Table 7.7). The estimated coefficients for the variables were all highly significant with the expected signs. The logged distance to estuary variable indicates, with all else constant, price decreases with increasing distance from the estuary, but the rate of decrease lessens with increasing distance from the estuary. The significance of this variable highlights the importance that residents place on accessibility and proximity to the estuarine environment. The structural variable of square meterage was positively related and highly significant, as expected.

Table 7.6. The number of properties, average property prices (2019 Rands), average property size and average distance to the estuary per suburb. Note that the suburb of Port Owen includes Admiralty Island. Source: Valuation roll data, Bergrivier Municipality.

Suburb	Average property price	Max property price	Min property price	No. of houses	Average property size (m^2)	Average distance to estuary (m)
Laaiplek	534 546	2 519 400	103 740	862	626	685
Noordhoek	148 877	1 393 080	37 050	1 579	342	1801
Port Owen	1 308 141	5 631 600	222 300	914	619	87
Velddrif	956 573	2 964 000	128 934	863	887	261
Total	644 149	5 631 600	37 050	4 218	572	887

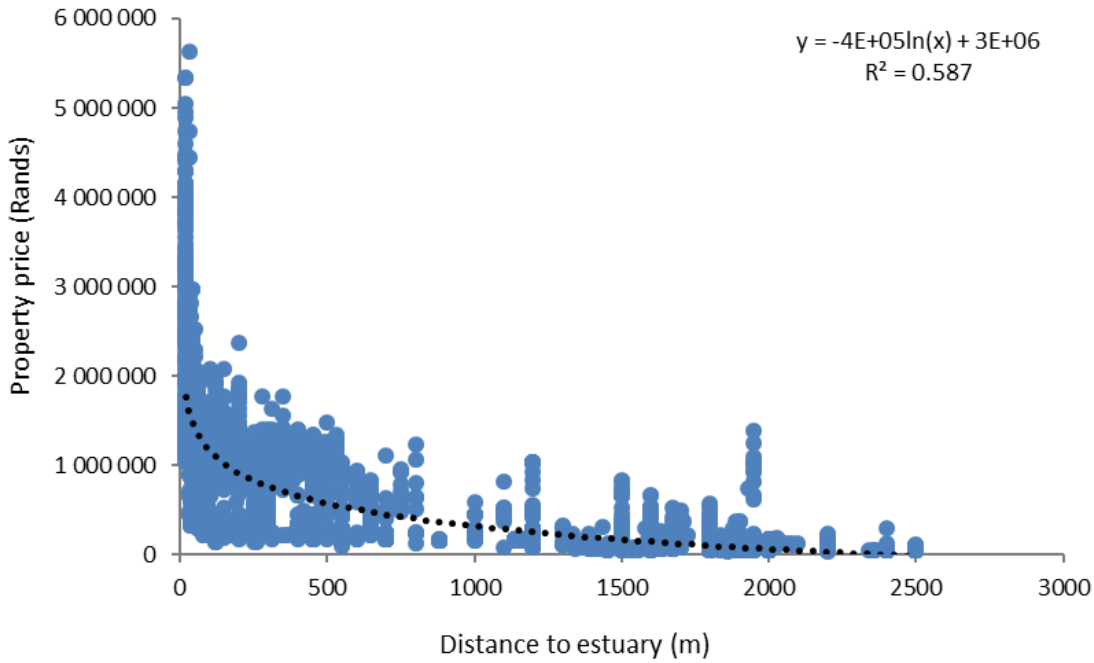

Figure 7.9. Relationship between property price and distance to the Berg River Estuary (n=4218, 2019)

Table 7.7. Model estimation results (n=4218, 2019, *** p-value <0.01)

Variable	Co-efficient	Standard error	t-value	Pr(>|t|)	p
(Intercept)	12.103	0.1316	91.92	< 2.2e-16	***
Ln (Distance to estuary)	-0.545	0.0069	-78.65	< 2.2e-16	***
Ln (Property size)	0.635	0.0171	37.1	< 2.2e-16	***

Based on this model, the total premium associated with proximity to the Berg River Estuary was estimated to be just over **R1.9 billion** (Table 7.8). While the total capital value compares well with the inflated estimate from Turpie *et al.* (2010) of R3 billion (2019 Rands), the premium

associated with the estuary is estimated to be slightly higher in this study when compared to the R1.5 billion (in 2019 Rands) estimated by Turpie *et al.* (2010). This is not unexpected, given the current property market and the demand for properties with estuary or canal frontage. The values generated in this study align well with the estimates of coastal premiums calculated by DWA (2017a). More than half of the total premium is associated with the properties of Port Owen and Admiralty Island which is expected given the location of this suburb and the number of properties with canal or estuary frontage.

Table 7.8. Estimated contribution of the estuary to property value by suburb (2019 Rands)

Suburb	Premium (R million)	Annualised value (R million)	% contribution
Laaiplek	259.1	22.6	13%
Noordhoek	59.4	5.2	3%
Port Owen	1 044.7	91.1	54%
Velddrif	570.0	49.7	29%
Total	1 933.2	168.5	

The property market does not appear to be sensitive to changes in estuary condition. Because the estuary is permanently open to the sea there is always water in the system and therefore changes in freshwater flow do not seem to have an impact on how homeowners perceive the estuary. A local estate agent seemed to think that while homeowners definitely appreciate that the estuary should be kept in a healthy state, they are not aware of what that state should be and cannot notice these changes when they do occur. As with tourism, it is likely that homeowners only become sensitive to deterioration in estuary health beyond some threshold when the changes become significant and noticeable. Such changes would include lengthened periods of poor water quality, algal blooms, bad smelling water, fish kills, excessive erosion and loss of shoreline habitats.

Figure 7.10. Aerial view of the Berg River Estuary and the town of Velddrif. Photo: J. Turpie.

Local recreation

In addition to estuaries generating revenues through domestic and international tourism, and through property values, they also generate value through recreational use. This relates to

the recreational and cultural use of the estuary by residents who live near to the estuary. Residents may spend time on the estuary fishing, bird watching or swimming. They may enjoy photographing or painting the various landscapes and fauna and flora of the estuary. This recreational use is an important component of the amenity value of the Berg River Estuary but is inherently difficult to value in monetary terms. However, after visiting Velddrif and speaking with individuals from the historically disadvantaged community it became apparent that access to the estuary is limited to certain user groups which has a bearing on how and when people are able to utilise the estuary for recreational purposes. These issues are discussed in Box 7.3.

Box 7.3. Recreational activities on the estuary are limited due to access and economic barriers. Photo: R. Robinson

A large proportion of the local population in the area is made up of historically disadvantaged people. Recreational opportunities linked to the estuary that are available to them (e.g. recreational line fishing and swimming) are currently severely limited by economic barriers and restricted physical access. However, it should be acknowledged that the Bergrivier Municipality is attempting to ameliorate the situation through by-laws and specific use zones for the estuary.

"The most famous spot for recreation is under the road bridge, there is no other place. We all use this area because we do not have swimming pools"

– Community leader, Noordhoek

Due to limited access to the estuary, the community does spend time at the beach, swimming there instead. The Laaiplek harbour is also an important recreational area. Residents highlighted the continued erosion of access points to the river, estuary and the beachfront as a concern. This clearly impacts on the sense of place, disrupting past recreation patterns and restricting potential future opportunities. The community also described estuarine recreational activities such as boating, skiing, kite surfing, and bird watching as being undertaken by other locals and tourists. They acknowledge this may be good for the local economy, but it is also a source of frustration. They explained that during holiday times there are too many speed boats on the estuary, which creates confrontation. This user group conflict can be managed in the future through the conservation initiatives introduced by the municipality and Cape Nature in the form of Cape Nature estuary rangers or through management of the municipal boating permit system.

For recreation, people mostly go to the beach to swim. People who catch fish with a line stand in the estuary at Velddrif or go to the harbour. People bring their children too, and from a young age they catch from the harbour walls." *– Recreational fisherman, Laaiplek*

7.3.8 Intrinsic and non-use values

When we talk about the intrinsic value of biological biodiversity, we are referring to its true,

inherent and essential value – that all life forms should be accorded equal value and should be conserved because they exist and have a right to continued existence. Intrinsic value is therefore linked to the biodiversity and condition of the Berg River Estuary and how people feel about it and its protection. The Berg River Estuary is one of the most valuable biodiversity assets along the South African coastline and its conservation is important to those who live near to it and visit it, as well as to those who do not, but appreciate its existence and would like to see it enjoyed by future generations. Therefore, this intangible measure is directly related to the health of the system and the need to facilitate the continued protection of all life forms in their natural habitats.

Most valuation studies use stated preference methods to determine non-use value of ecosystems. These methods involve questionnaire surveys of the affected population and elicit peoples' willingness to pay for the benefit of retaining or acquiring an improvement in an environmental asset or willingness to accept compensation for its degradation or loss. Although there has been an estimate of the non-use value of South African biodiversity in general (Turpie *et al.* 2003), and some attempt to estimate the existence value of estuaries (Turpie & Clark 2007), these estimates are outdated and did not fully capture the magnitude of the existence value of the Berg River Estuary. More recently, Turpie & Letley (in review) conducted a survey of Capetonians in order to estimate willingness to pay for procuring water from more alternative sources in order to avoid having to push the rivers and estuaries of Cape Towns water source catchments into a poor state of health. Their aggregate willingness to pay, which was over and above willingness to pay to have a secure water supply, was in the order of R0.5 billion per year. Even if only 10% of this was allocated to the Berg River, one of the two largest river systems, then this suggests that maintaining the health of the Berg River system in a C-category rather than an E-category could be worth at least R50 million per year. This estimate is very imprecise, but suffices as a ball-park estimate. Conservatively we estimate that the existence value of the system could range from R12.5 million in an E category to R50 million in a B or A category.

Figure 7.11. The Berg River Estuary is one of the most valuable biodiversity assets along the West Coast of South Africa and its existence value is thought to be substantial. Photo – White-fronted plover: J. Turpie.

7.3.9 Summary of values

The value of the Berg River Estuary is estimated to be in the order of **R378 million** per year (Table 7.9). The amenity or use value associated with the estuary (nature-based tourism and property values) accounts for more than half of the total value of the system. The contribution that the estuary makes to fisheries, through important nursery areas and providing a sheltered harbour, accounts for about one third of the total value.

Table 7.9. Summary of values associated with the Berg River Estuary in its present condition.

Benefit	Value (2019 Rands)
Subsistence fishing	1 200 000
Livestock grazing	11 500 000
Salt production	8 300 000
Tourism value	36 000 000
Property value	168 000 000
Nursery value	8 660 000
Carbon sequestration & storage	12 386 000
Harbour facilities	107 000 000
Bequest and existence value	25 000 000
Total	**378 046 000**

The recent drought (2015-2018) provided the opportunity to assess how changes in freshwater flows into the Berg River Estuary might affect the delivery of ecosystem services and the benefits that they provide. A marine dominated system favours the abundance of certain marine fish species which has positive effects on fisheries values. Tourists and property owners do not appear to be sensitive to changes in freshwater flow. In fact, these values have increased over the last few years in and around Velddrif and the estuary. The Berg River Estuary is permanently open with a high tidal influence which ensures the continued movement of water in and out of the estuary. Furthermore, when the system is marine dominated, the water is clear which is seen to be more attractive than when the system is freshwater dominated and 'muddy'. While residents and visitors are likely to appreciate that the estuary should be maintained in a healthy state, they are unaware of what this state is and/or should be and do not notice subtle changes in condition. It is likely that residents and visitors only become sensitive to deterioration in estuary health beyond some tipping point when the changes become significant and noticeable. Therefore, it is expected that up to a certain point, amenity values are likely to remain insensitive to changes in condition. However, beyond some threshold, this could change, and these values could be lost. This is explored in more detail in the following section.

As the system moves further and further away from "natural" condition, its ecosystems deteriorate, and biodiversity is lost. This loss in biodiversity can lead to a breakdown in the functioning, resilience and productivity of the ecosystem where the decline has occurred which in turn leads to a deterioration of the benefits that humans obtain from these ecosystems. Biodiversity loss also leads to an increase in the likelihood of ecological 'shocks', such as impacts of climate change, floods, fisheries collapse, and eutrophication.

Therefore, the conservation of the Berg River Estuary and its biodiversity is critical to maintaining economic outputs and human welfare for present and future generations.

Policies that focus on the conservation of biodiversity are more likely to promote higher overall human well-being through the preservation of various ecosystems services that are generated by these important estuarine habitats.

7.4 Potential changes in value under different flow scenarios

7.4.1 Relationship between estuary condition and estuary values

The condition (determined ecological class) of the Berg River Estuary varies under the different flow scenarios described in the preceding sections, varying from a near-natural B category to a highly modified E category. In order to evaluate the impact on estuary value, it is necessary to understand the underlying links between ecosystem structure and function and the supply of ecosystem services as well as their demand. In this study, the impacts of changes in ecological condition are estimated on the basis of assumptions on the relationships between ecosystem health and capacity to supply ecosystem services, and the value of these services. The main types of ecosystem services considered in this scenario analysis are summarised in Table 7.10, along with the characteristics that are likely to be the main drivers of these values.

Table 7.10. Types of values provided by the Berg River Estuary and the independent variables related to estuary condition

Types of values	Description	Independent variables related to estuary condition
Subsistence fishing	Invertebrates and fish collected on a subsistence basis for consumption or bait	• Invertebrate abundance • Freshwater fish abundance • Estuary line- and net fish abundance
Nursery value	Contribution to marine fish catches due to the nursery habitat provided by the estuary	• Abundance of estuary dependent marine fish
Tourism value & property value	The estuary's contribution to recreation/tourism appeal of a location	• Overall health • Line fish abundance • Water quality
Carbon storage & sequestration	Contribution to the amelioration of climate change damages through sequestration of carbon from the atmosphere by estuarine habitats	• Overall health • Water quantity and quality
Floodplain farming	Contribution to livestock production due to floodplain vegetation provided by the estuary	• Water quantity and quality

In order to inform this analysis, data from the recent Berg River Resource Quality Objectives study were used. The RQO study explored the relationships between abiotic and biotic scores and the overall health score for estuaries. In general, it was found that the component scores were strongly correlated with the overall health scores, with all having a slope close to unity. Variation was highest for birds, which are influenced by non-flow disturbance factors, fish, which are influenced by fishing, and macrophytes, which are influenced by habitat loss through development. These findings suggested that the overall relationships are generally consistent with the Ecological Health Index (EHI) score. For this

analysis we have used the conversion factors generated by DWA (2017a) to estimate the change in ecosystem service values based on changes in estuary health condition.

Based on the information collected during this study and observations of change during the recent 2015-2018 drought, it became clear that changes to freshwater flows into the Berg River Estuary and subsequent changes in estuary condition appear to have not had a noticeable impact on some estuary values. In fact, it appears that the marine dominated state associated with lower freshwater flows has had a positive impact on the biomass of targeted fishery species and has had no discernible negative impact on tourism and property values. In fact, the tourism and property markets have been increasing in Velddrif over the last few years and have not shown any indication of slowing down. This suggests that for the Berg River Estuary, certain values are less sensitive to changes in estuary condition. This is unsurprising given that for a large open estuary, a change away from natural does not necessarily manifest as a reduction of productivity or functioning, but rather a shift to a different kind of system. Such shifts are not always negative from a user perspective. However, even in a relatively resilient system such as the Berg River Estuary, there will be a point beyond which decreasing condition is likely to lead to noticeable degradation for the casual observer. Such changes include lengthened periods of poor water quality, algal blooms, bad smelling water, fish kills, excessive erosion, and loss of shoreline habitats. Based on the results presented above on the likely impacts of future scenarios on estuary health, we believe that these changes would occur when the health of the estuary deteriorates to an E class.

There are some estuary values that have already been negatively affected by the reduction in freshwater flows into the estuary. Farmers have had to reduce livestock numbers due to the loss of floodplain grazing areas which is linked to the loss of floods that feed these floodplains. Extended periods of low fluvial input are also likely to have had an impact on carbon storage and sequestration values as estuarine habitats have been degraded or lost. Intrinsic and non-use value is also expected to decline with the persistent reduction in freshwater inflows to the estuary.

7.4.2 Impacts of flow scenarios on estuary value

Changes in value as a result of changes in estuary condition were broadly estimated for subsistence fishing value, nursery value and tourism and property values, carbon storage, salt production and floodplain farming, as summarised in Table 7.11. Harbour facilities are unaffected by changes in flow and estuary condition and therefore the benefit remains constant across all scenarios.

The ecological condition of the estuary does not change from the status quo (P0) for Scenarios P1, F0 or F1, remaining in a D-class. Therefore, the estimated value of the benefits do not change for these scenarios. Under the climate change Scenarios C0 and C1, the overall value of the benefits considered could decline by 30% as a result of losses in tourism and property values as well as losses in fishery values, carbon storage and floodplain farming values. Salt production is expected to increase slightly under these two scenarios. Under scenarios P2, F2 and C2 where the estuary is returned to health category C-class, the value of the benefits could increase by more than 20%, with significant improvements in fishery values as well as small increases in tourism and property values. In order to be comparable, these estimates are an indication of how different the values would be today, were the estuary in that condition. In future, the value of the estuary will grow, given that there will be more demand from an increased population with higher average income, coupled with a

decreased supply of natural capital and ecosystem services. The implications of this are explored further in the synthesis chapter.

Table 7.11. Estimated values of the Berg River Estuary under different scenarios. Note that the status quo is P0.

Health category	B	C	D	E
Corresponding flow scenarios	**Not feasible**	**P2, F2, C2**	**P0, P1, F0, F1**	**C0, C1**
Subsistence fishing	2	1.7	1.2	0.7
Livestock grazing	14	12.7	11.7	5.9
Salt production	8.3	8.3	8.3	10.4
Tourism and property value	293.7	280.9	204	114.1
Non-use value	50	45	25	12.5
Nursery value	14.7	12.1	8.7	4.7
Carbon sequestration & storage	14.9	13.6	12.4	6.2
Harbour facilities	107	107	107	107
Total	**504.6**	**481.3**	**378.3**	**261.5**

8 SYNTHESIS AND RECOMMENDATIONS

8.1 Introduction

Understanding the value of investing in natural capital is essential to achieving policy decisions that result in maximising long-term benefits to society. Because of the strong reliance of estuaries on freshwater inputs and their sensitivity to local development and use, decisions on the extent to which estuary health should be prioritised are particularly complex. They rely on understanding not only of the multitude of catchment and estuary-based activities that impact on estuary health, but also on the factors determining the health and value of the estuary itself. Given the regional and national importance of the Berg River Estuary in terms of biodiversity and ecosystem services, there are strong calls to increase its level of protection. Understanding what values are at stake and the factors affecting those values is helpful in weighing up the costs and benefits of management decisions.

The allocation of the ecological Reserve is central to the environmental, economic and social outcomes of a region (Figure 8.1). Water is not only directly critical to social and economic development, but also indirectly, by supporting key ecological systems which provide essential ecosystem goods and services that underpin development and human wellbeing. Economic activities that depend on the licenced use of water in the Berg River include urban supply, irrigation agriculture and industry. Activities linked to the quality of the estuary itself include tourism, recreational angling, estuarine subsistence fisheries and marine commercial fisheries. Similarly, social wellbeing within the study area is determined by both water supply and instream flows, namely the abstraction and supply of water for domestic purposes, the supply of abstracted or instream water to economic activities which provide employment opportunities, and the supply of instream flows which lead to the provision of instream water, natural resources and opportunities for recreation.

To assess water allocation trade-offs, the effects of changes in estuary health need to be valued and compared with the economic costs of reducing the system's yield for water users. Ideally, the marginal benefits should just equal the marginal cost (or shadow price) of restoring one additional unit of fresh water to the estuary. Since the Berg River Estuary is located in a water-stressed catchment, the costs are estimated in terms of water demand and supply interventions that compensate for the impacts on planned system yield without impacting on sectoral outputs. The question as to whether water should be reallocated from agriculture or urban/industry is therefore moot.

The situation becomes complicated under climate change, when a policy decision needs to be made as to how the expected reduction in rainfall should be suffered by aquatic ecosystems *versus* water users. Should we be attempting to maintain pre-climate change standards set for ecosystems, at the expense of surface water system yields, or should aquatic ecosystems be allowed to degrade in concert with reduced rainfall, by setting environmental flow requirements as a proportion of changing natural runoff? In this study, we contemplate both extremes as well as a middle-of-the-road "shared losses" scenario.

This chapter draws together the information presented in the preceding sections on the ecological functioning and status of the estuary, and how this will change under the different scenarios (other than the scenarios that are defined by condition). Here we also introduce the option of increasing flows to the system to achieve an increase in health category. Following this, we consider the costs and benefits of supplying more water to the estuary.

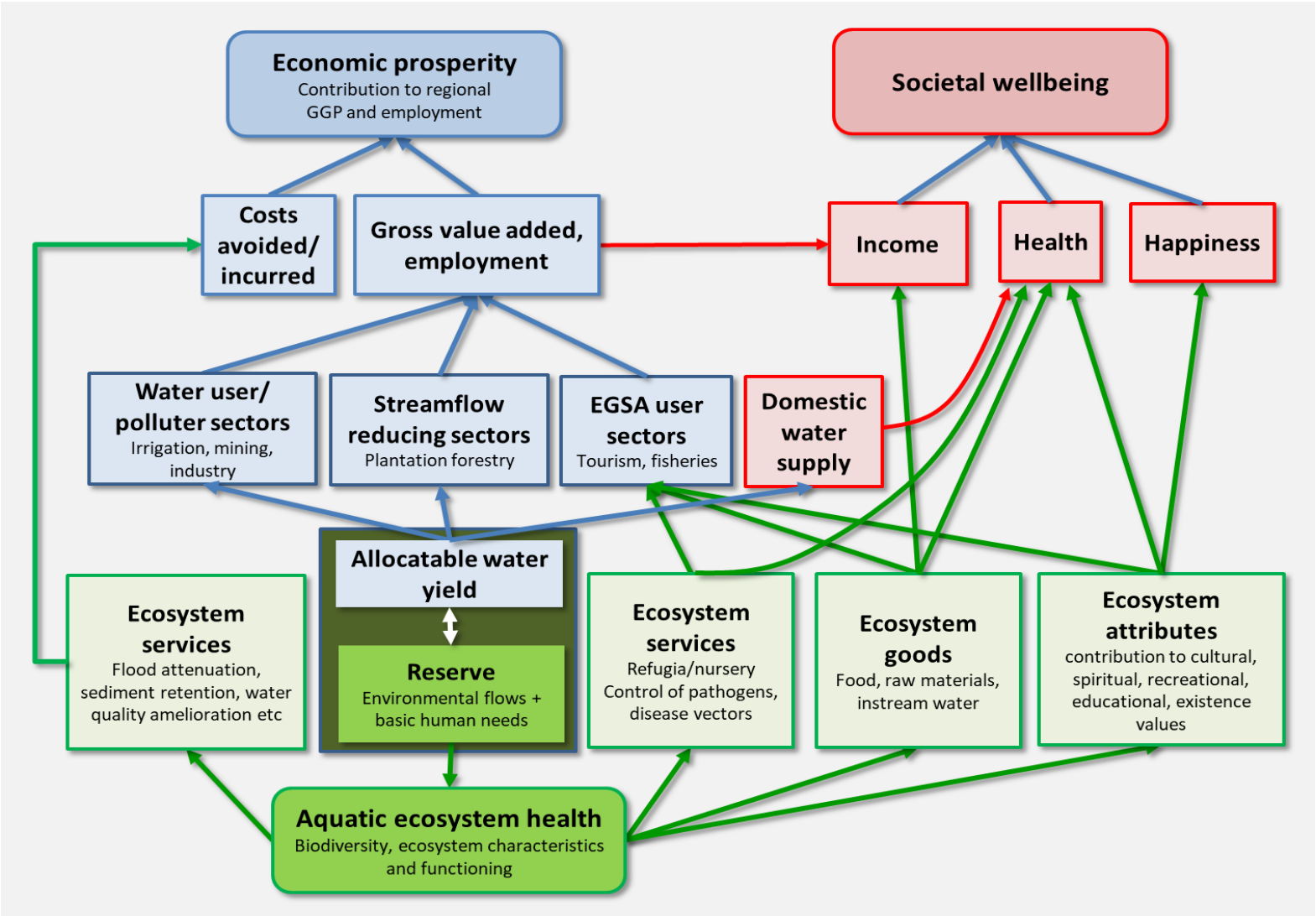

Figure 8.1. Linkages arising from the trade-off between water abstracted for use and water retained for the ecological Reserve (Source: DWS, 2017). EGSA stands for ecosystem goods, servicew and attributes.

8.2 Present Ecological Status of the Berg River Estuary

8.2.1 Summary of the influence of freshwater inputs on estuary characteristics and functioning

Freshwater flows entering the estuary affect salinity structure, water residence time, turbidity and sediment deposition in the system. They bring nutrients and also influence the balance between inputs from the catchments and from the sea. Floods affect the degree of inundation of the floodplain and sediment scouring, and tend to have the highest nutrient loads. Freshwater flows affect the nature and extent of physical habitats (sand and mud) while salinity and the degree of inundation affect the nature and extent of biotic habitats (e.g. mudflats, reedbeds, salt marshes).

Salinity affects the species composition of all of the biotic components, with different species having different salinity tolerance ranges. Abundance and productivity are largely influenced by availability of inorganic nutrients and food. Freshwater flows bring the bulk of the nutrients into the system, which directly or indirectly feed all of the biotic components, and together with tidal forcing, determine the water residence time in the estuary, which allows the nutrients to be used in micro- and macrophyte production. It is hypothesised that there is an optimal level of flow which maximises microalgal productivity under natural levels of nutrient input. Microalgal productivity is the most important determinant of overall biomass of estuarine biota, with most trophic pathways originating in microalgal rather than macrophyte (plant) productivity.

The temporal patterns of flow are also an important factor shaping the nature of the system. For example, unseasonal flooding may not benefit floodplain birds or facilitate fish recruitment into the estuary.

The biomass of all consumer groups is determined by a combination of food and habitat, either of which may be limiting, but both of which are influenced by some aspect of flow. There are artificially high nutrient inputs into the system, thus it is likely that habitats requiring higher levels of inundation or scouring may constitute the most significant limiting factor in the system. Any reduction in flow will have a greater impact on the fauna and flora of the system through loss of habitat rather than reduction in food supply.

Species diversity is primarily determined by habitat but is also a function of overall system productivity and stability. While variable habitats may support high instantaneous diversity at times, when conditions attract opportunistic species, specialist resident species will only occur when specific habitats or conditions are permanently available. Eutrophication resulting from excessive nutrient inputs can also lead to a situation where some species become overwhelmingly dominant and exclude other species.

8.2.2 Recap of the 2010 assessment

A comprehensive assessment of the Berg River Estuary was conducted in 2010, post completion of the Berg River Dam (DWA 2010). This assessment resulted in a biotic health score of 63% of Reference and an overall Estuarine Health Score of 64% for the Berg River Estuary (C). It was noted in 2010 that the estuary was on a negative trajectory of change, because of the extremely low dry season flows under the present state (< 1 m^3s^{-1}), particularly

during the summer months, and was expected to decline unless flows were improved.

The Berg River Estuary is considered an endangered estuarine ecosystem with no current formal protection status (unprotected); it is part of the National and Cape Biodiversity Priority Core Set of estuaries and is considered as a High Priority in terms of fish nursery habitat (Van Niekerk *et al.* 2018). These factors provide the motivation for setting the future management objectives of restoring the estuary to an A or B category of health (DWA 2012). However, given the existing demands on the catchment, the best attainable state was estimated to be a C-category. Thus, the Recommended Ecological Category (REC) of the Berg River Estuary was set at a C.

In addition to the low freshwater flows, other pressures and threats to the biodiversity identified in these assessments and the Berg River Estuary Management Plan (Anchor 2010) included:

- Alien invasive vegetation and macrophytes;
- Increasing nutrient and sediment enrichment due to catchment agriculture;
- Damage to estuarine vegetation by livestock;
- Potential risk of pollution entering from the harbour area and sea;
- Past, ongoing and potential future overexploitation of living resources;
- Potential future tourism-related development along the estuary affecting sense of place and increasing disturbance;

These pressures on the Berg River Estuary biota have not abated.

8.2.3 Updated present ecological status of the estuary

In this study, we reassessed the health of the estuary. This latest health score is significantly lower than that of 2010, partly because we have greatly improved our understanding of the system hydrology and functioning. Freshwater inflows have been lower than was previously estimated, suggesting that previous scores were overestimated. The lower score for 2020 is therefore due both to a reassessment of historical hydrology as well as continued reduction of the quantity and quality of inflows. In addition, direct pressures on the estuary have increased, and the global populations of migratory birds continue to decrease.

The 2010 assessment (DWA 2010) was largely based on data collected in the period 2003-2005. Since then, MAR has dropped from an estimated 500 Mm^3/a (54% of Natural) to an estimated 453.28 50% Mm^3/a (50% of Natural). Irrigation demands below Misverstand Dam have increased dramatically since 2010 (up from around 15 Mm^3/a to over 24 Mm^3/a). Summer lows flows meet the EWR requirements of 0.6 m^3/s less than 15% of the time and ceased completely during the recent (2017/8 drought). The Berg River Estuary is far more marine dominated than under the Reference condition and more so now than it was in 2010. The long-term reduction in freshwater flows was exacerbated by the unprecedented 2015-2018 drought with dramatic consequences for water quality. This included extended periods (18 months) of elevated salinity, which sometimes led to a reverse salinity profile where salinity increases upstream and hypersaline conditions developing in the middle reaches of the estuary. Turbidity was reduced in the middle and upper reaches because sea water was penetrating further upstream. The magnitude and/or frequency of large floods appears to have changed little since the construction of the Berg River Dam in 2006, but since small and medium-sized floods in winter are much reduced, flooding of the upper estuary is also dramatically reduced, an observation that is strongly supported by anecdotal evidence

supplied by riparian landowners. Nevertheless, the permanently open mouth and substantial tidal exchange ensured enough flushing to maintain water quality suitable for most marine and estuarine biota, even during the drought. The less salinity-tolerant biota such as *Phragmites* reeds have retreated up the system and did suffer further mortalities during the recent drought. Numbers of birds, the only biotic group that has continued to be monitored, have continued to decrease across several groups, although most dramatically for migratory waders. While some groups have responded positively to the increasingly saline conditions, the overall response is a reduction in numbers and diversity in the estuary.

Table 8.1. Health scores and corresponding ecological condition under historical (2010) and present day (2020) conditions.

Variable	Weight	Historical 2010	Present Day 2020
Hydrology	25	72	36.3
Hydrodynamics/mouth condition	25	90	60.8
Water quality	25	40	31.1
Physical habitat alteration	25	71	66.0
Habitat health score	50	**68**	**48.6**
Microalgae	20	75	68.3
Macrophytes	20	54	45.7
Invertebrates	20	54	49.1
Fish	20	56	66.8
Birds	20	82	56.3
Biotic health score	50	64	57.2
Estuarine Health Index Score		66.3	52.9
Ecological Reserve Category (ERC)		C	D

Since 2010, the overall health of the estuary has changed from 66.3% similarity to Reference condition to 52.9%, and has dropped a whole category – from a "C" (moderate modified) to a "D" category (largely modified)[8]. **The system is thus no longer compliant with the gazetted RQOs** which require maintaining the system in a C category. This change in health has largely been driven by reductions in physical health (down from 68% to 49%) with changes in biotic health lagging somewhat (down from 66% to 57%), which is not uncommon in these situations (Turpie *et al.* 2012).

8 Based on the Estuary Health Index, Turpie *et al.* (2012).

8.3 Impacts of alternative flow scenarios on estuary health

This study modelled the effects of (a) ignoring estuary water requirements, so that the quantity of flows to the estuary are effectively what is left over from water abstractions in the catchment, (b) meeting the recently-gazetted EWR requirements, and (c) allocating more water than the gazetted requirement, specifically to achieve a significant improvement in health of the estuary. These environmental flow options are considered under conditions of present-day water demand and supply infrastructure, as if implemented immediately. We also consider the impact taking into account the planned future water infrastructure developments as part of the WCWSS. These are also considered for a hypothetical future without climate change, so that the effects of infrastructure from the impacts of climate change could be understood by the reader.

The predicted further substantial declines in freshwater flows under climate change scenarios (Scenarios C0 and C1 where flows are predicted to decline by a further 33%, to around 35% of Reference flows) could result in two dramatically different outcomes for the Berg River Estuary biota. The best-case outcome is that the permanently open mouth continues to ensure sufficient marine water exchange to maintain tolerable salinity and oxygen concentrations throughout the estuary. With this outcome some estuarine resident species that require or prefer salinity less than 35 are expected to decrease in abundance as they shift their distributions upstream and the narrowing of the estuary channel results in less habitat available within the system. Marine migrants are expected to increase in abundance as they utilize an increased proportion of the estuary area. From an estuary health perspective, for this best-case outcome, the Berg River Estuary will still be further away from the reference state (i.e. the biotic health score would decrease), but will continue to be functional and productive. Notably, water column and benthic diatoms, saltmarsh vegetation, marine migrant fish (category II species) and piscivorous birds appear to thrive under drought conditions, whilst freshwater dependent vegetation and estuary resident invertebrate and fish taxa do not.

The worst-case outcome with the predicted freshwater flow reduction in Scenarios C0 and C1 sees hypersaline, low oxygen water developing in much of the upper and middle reaches of the estuary. Hypersalinity and low oxygen conditions have been observed during drought conditions, and with a permanent, large reduction in freshwater input, this would be expected to be common under C0 and C1. Conditions within this zone would be unsuitable for nearly all biota (essentially equivalent to habitat loss) and dramatic declines in abundance of estuarine resident and catadromous fish species (e.g. *Mugil cephalus*) are predicted. This is a likely outcome if the productive river estuary interface (REI) zone becomes hypersaline or hypoxic (oxygen depleted). Marine migrant species are also expected to show substantial declines in abundance as the available estuary habitat is reduced and especially if the productive REI zone is reduced in size or becomes unsuitable habitat. This worst-case outcome is considered less likely and biodiversity health scores for scenarios C0 and C1 are based on the more optimistic outcome described in the previous paragraph.

Anticipated changes in health under a range of scenarios are presented in Table 8.3. Under a future development context without climate change (F0), MAR decreases from 50 to 47% of Reference (natural) flows, and the overall health of the system would be expected to decrease from 52.9 to 51.1% of Reference conditions. Respecting the low flow EWRs yields is not expected to yield much benefit in overall estuary health in future (overall health score rises to 52.6%). If expected climate change is considered, the estuarine health score drops

even further (39.8% for both C0 and C1) and tips the estuary into an E-class. Note that the scenarios involving increasing overall water allocations to the estuary were not considered here, as their health was predetermined to be in a C-category.

Table 8.2. Hydrological scenarios assessed in the study. P0 is the assumed status quo. The naming of the scenarios is linked to development and climate context: present context (P), future hypothetical context without climate change (F) and future predicted context with climate change (C); and to the EWRs: ignoring EWRs (0), meeting gazetted EWRs (1); meeting the flow requirements for a C-class (this is higher than gazetted).

		No EWRs.	Gazetted low flow EWRs (0.6 m3/sec)	Meeting requirements* for a C-class
Present-day WCWSS Infrastructure (PDI)	Present-day	**P0**	**P1**	**P2**
Future Infrastructure (FI) as planned	Without climate change	**F0**	**F1**	**F2**
Future Infrastructure (FI) as planned	With climate change	**C0**	**C1**	**C2**

* As updated in this study following reassessment of hydrology.

Table 8.3. Health scores and corresponding ecological condition under present day conditions and for the different runoff scenarios.

Variable	Weight	Hist. 2010	P0 2020	P1	F0	F1	C0	C1
				PD+EWR	FD	FD+EWR	FD+CC	FD+CC+EWR
Hydrology	25	72.0	36.3	38.0	35.5	40.8	22.9	28.7
Hydrodynamics/ mouth condition	25	90.0	60.8	60.8	58.9	61.3	46.4	46.8
Water quality	25	40.2	31.1	32.5	28.7	30.8	21.5	22.1
Physical habitat alteration	25	71.0	66.0	66.0	66.0	66.0	59.0	59.0
Habitat health score	50	68.3	48.6	49.3	47.3	49.7	37.5	38.0
Microalgae	20	75.0	68.3	67.5	62.7	64.2	44.5	42.9
Macrophytes	20	54.0	45.7	45.2	42.0	43.0	34.9	35.0
Invertebrates	20	54.0	49.1	49.5	47.1	47.1	31.3	30.9
Fish	20	56.0	66.8	66.8	66.8	66.8	50.1	50.1
Birds	20	82.0	56.3	56.7	56.2	56.6	49.6	50.5
Biotic health score	50	64.2	57.2	57.1	55.0	55.5	42.1	41.6
Estuarine Health Index Score		**66.3**	**52.9**	**53.2**	**51.1**	**52.6**	**39.8**	**39.8**
Ecological Reserve Category (ERC)		**C**	**D**	**D**	**D**	**D**	**E**	**E**

8.4 Balancing Estuary EWRs with water availability in the WCWSS

In terms of balancing the gazetted requirements for the estuary with that of the other users of

the WCWWS, the results indicate that providing for the minimum flow requirements (i.e. 0.6 m^3/s) has very little impact on the available yield from the system under both the current and future infrastructure scenarios. This is because the actual volume of water required is relatively small and this could relatively easily be ensured through improved operation of the system. Providing additional storage at Misverstand Dam, combined with improved monitoring downstream, would further improve the ability to maintain the minimum EWR requirements without impacting on the demands of other users. Under the future climate change scenario, however, even providing the minimum flow requirements will impact on the available yield from the system, although this only results in an 9.7 Mm^3/a, or 2% reduction in the HFY compared to the no EWR scenario.

The biggest impact on estuary health is in terms of the winter flow requirements. Under the current climate scenario, there is only a 7% reduction in the average volume relative to present day, while under the future climate change the average volume of winter water is reduced by up to 37%, or around 145 Mm^3/a. In order to maintain the necessary winter water flows to the estuary under this scenario without reducing the current allocation to users of the WCWSS would require this volume of water to be replaced, on average from an **alternative source**.

Table 8.4. Estimated Firm Yield under the different scenarios, and their difference from yields that would be obtained without an EWR requirement.

Scenario	MAR (Mm^3)	Firm Yield (Mm^3)	Difference from no EWR (Mm^3)
Reference	912		
P0	**459**	**508**	
P1: PD + EWR	469	508	
P2: PD + C class	593	374	-134
F0: Future	**433**	**528**	
F1: Future + EWR	438	528	
F2: Future + C class	593	322	-206
C0: Future + CC	**303**	**480**	
C1: Future + CC + EWR	312	470	
C2: Future + CC + C class	593	205	-265

Note: the yield impacts of the C-class scenarios are preliminary estimates.

Given that there are no other major surface water options considered viable in the WCWSS, and in fact these themselves would also be similarly impacted by climate change, the likely alternative supply option would be either the Table Mountain Group (TMG) aquifer, or direct potable re-use or desalination. All three of these options are currently being considered by both the City of Cape Town and DWS. The latest estimates suggest that average production costs for these alternative water supply options (i.e. excluding the capital costs) would range between R5/kl and R9/kl. Even using the lower of these cost estimates suggests that to provide alternative water supply to off-set the requirements for maintaining the EWRs would be on average around R725 million per year (in 2020 Rands). Note that this is a very rough estimate and requires further analysis based on an actual time series of demands. It is also based on an average annual flow requirement.

Improving estuary health and value involves securing more and/or better quality water than is reaching the estuary under present day conditions. This will come at some cost to society. If flows are to be increased to the estuary, this means that either there will be a lower firm yield to allocate to existing agricultural, industrial and/or urban users, or that further investment will be needed to improve efficiency and/or maintain water supply by other means in order to maintain existing or future projected levels of economic output and welfare. Potential options to free up water for the estuary are summarised in Table 8.5.

Table 8.5. The most feasible options to free up more water for the Berg River Estuary

Type	Options	Comment	Potential	Cost (R/m^3)
Demand side	Urban and industrial demand management through pricing	Proven and feasible	50 Mm^3 p.a.	0
	Agricultural demand management or curtailment	Options largely exhausted, low feasibility	Negligible potential	n/a
Supply side	Removal of invasive alien plants	Cheapest supply option	55 Mm^3 p.a.	R6.76
	Conventional infrastructure and recycling	Options already planned in	Limited further potential	n/a
	Desalination	Unlimited	Balance of requirements	R8.82

On the demand side, reducing consumption could be achieved by the implementation of measures to improve efficiency, which is usually incentivised through increased water pricing. The recent drought experience in Cape Town has demonstrated what can be achieved in terms of urban and industrial demand with extreme measures involving a combination of restrictions and pricing. For example, it was estimated that water usage in the agri-processing sector was reduced by about 18.5% with only limited reduction in the volumes processed (Oosthuizen *et al.*, DEDAT, 2019). Urban demands were reduced by more than 40%, from about 1000 to about 500 ML per day (City of Cape Town 2019, Ziervogel 2019). This suggests that a reduction in demand could be achieved on a more sustainable basis through pricing strategies, perhaps in the order of 150 ML per day (about 50 Mm^3 p.a.) relative to the "normal" pre-drought demand[9]. Given the relative inelasticity of water demand, this would potentially be achieved at zero net cost.

There is probably less opportunity for demand management in agricultural areas, where efficiency gains have already been high in the past. Indeed, these areas were subject to significant curtailment during the drought. A more extreme option is curtailment, where the allocations to certain user groups such as farmers are reduced. This will have the impact of

[9] It is not advisable to push for very much more efficient water use as this reduces the ability to manage demand and it may also increase the cost of water as it has to be provided at a higher level of assurance of supply. An amount of 150l/c/d is a good target and is what is currently being targeted by the city. The current thinking is that demand will go back to around 80% of water it was pre-drought, but that it will then continue to grow in line with population and economic growth.

reducing economic outputs and employment, and as such, is likely to be politically infeasible.

On the supply side, more water could be made available through the removal of invasive alien plants (IAPs) in catchment areas, increasing conventional infrastructure such as water supply dams, and developing alternative sources such as groundwater, recycling of waste water and desalination of sea water. There is already a strong case for investing in alternative water supplies (PDG, DEDAT, 2019, TNC, 2018). The conventional and recycling options have already largely been exhausted in the current plans. Thus, the main opportunities lie in clearing IAPs, recycling and desalination. A recent study has established that an investment of R372 million in IAP clearing will generate annual water gains of over 55 Mm3 a year within five years and that it is the lowest cost of the alternative supply options (Turpie *et al.* 2019).

Desalination is currently the most expensive option for augmenting supply, and is considered a last resort once all other alternatives have been exhausted. The City of Cape Town has recognised that desalination will become an important part of their water future and have stated that in the medium and longer term, "desalination is very likely to become an increasingly significant share of the mix because it is scalable and not dependent on rainfall" (City of Cape Town 2019). Furthermore, desalination has become much more efficient and cost-effective in recent years thanks to advances in technology, reductions in costs and energy use, increase in plant size and more competitive project delivery (World Bank 2019). Indeed, costs (including capital) have been reduced to as low as US$0.50 per m^3 of desalinated water in some parts of the world, with the general cost range now falling between US$0.60-1.80 depending on the size of the plant and the type of technology used (as well as site specific conditions, The Water Wheel 2018, World Bank 2019). Further large cost reductions are expected with the World Bank (2019) reporting declines of up to two-thirds over the next two decades. Within 5 years it is expected that the cost of water through desalination will range from US$0.6-1.0 and that in 20 years it will be as low as US$0.3-0.5 (World Bank 2019). Based on this information, we assume that the cost of desalination in South Africa will follow these trends but have used a conservative unit cost of US$0.6 per m^3 (the lower end for the 5-year forecast and upper end of the 20-year forecast) to cost the additional water needed to cover the respective deficits under each scenario. This equates to R8.82 per m^3 (1 US$ = R14.70, January 2020). Locally, the costs would have to incorporate local electricity prices and the infrastructure requirements to link into the grid.

The latest estimate from the City of Cape Town put ths at around R15/m^3 URV, with unit capital costs of around R1.6 billion for a 50 ML/d plnat and production cost of around R9/m^3.

8.5 Costs and benefits of securing environmental flows

This study considered a range of scenarios in which gazetted estuary water requirements are ignored, partially or fully met, or exceeded in order to improve the health class of the estuary to one that might be more in line with its national ranking in terms of importance for biodiversity. These options are analysed under present condition as well as under future development and climate scenarios. In general, supplying a higher proportion of the Reference level flows to the estuary, in line with seasonal requirements, results in a more healthy and valuable estuary (Figure 8.2). Based on differentiation of the function in Figure 8.2, this suggests a marginal value in the order of R673 per ML at present. The marginal value of water inputs to the estuary will decrease with increasing inputs, however.

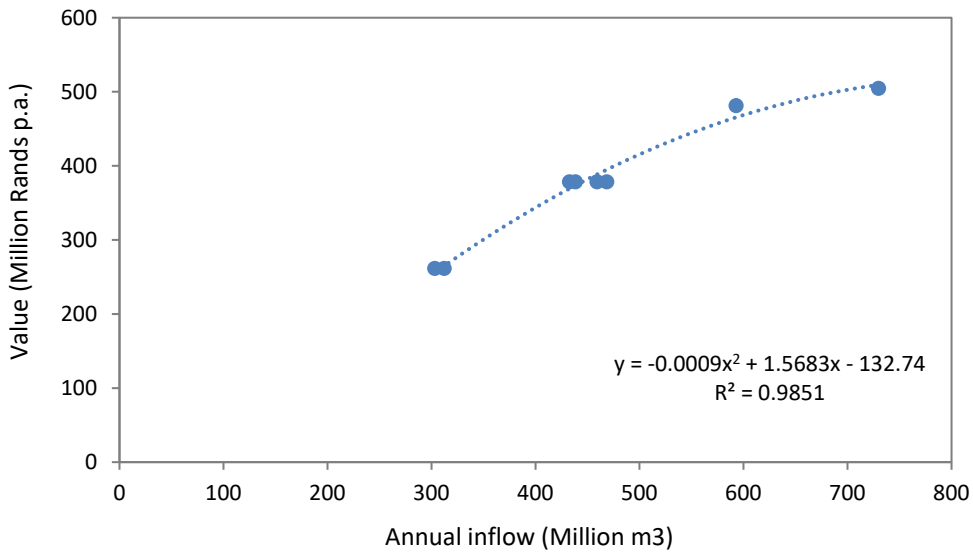

Figure 8.2. Estimated value of the Berg River Estuary in relation to annual freshwater inflow (change in rands/annual inflow = gradient)

Based on the above options, we provide a range of estimates of the costs of meeting water supply ranging from the full use of options, including water demand management, to just relying on desalination. This is summarised for the different scenarios in Table 8.6. Depending on how many interventions are used to tackle the problem, the cost of supplying the additional water needs to the estuary range from about R4.70 to R8.80 per m^3.

Finally, the estimated changes in value of the estuary are compared with the estimated additional costs of water supply (Table 8.6). From this it is evident that, while increasing flows to improve the condition of the estuary to a C-category will have a measurable impact on yields, the value gains could outweigh the costs, especially if efforts are made to restore catchment areas and curb urban demands through more appropriate pricing strategies.

Table 8.6. Estimation of the net benefit of meeting full EWR requirements for a C-class estuary, under present and future infrastructure development and climate change scenarios. Difference in yield and asset value is calculated relative to the corresponding situation with currently-gazetted EWRs. Costs of yield replacement are based on three alternative policy scenarios: L includes demand management, IAP clearing and desalination, M includes the last two and H is based on desalination alone.

Scenario	MAR (Mm3/a)	Health category	Firm Yield (Mm3)	Difference in yield from planned (Mm^3)	Asset value (Rm)	Difference in asset value (Rm)	Replacement cost for change in yield (NPV)			Net benefit of supplying full C-class EWRs (Range, in R m NPV)
							DM + IAP + Desal (Rm)	IAP clearing + Desal (Rm)	Desalination only (Rm)	
Reference	912	A	-		-					
P0	459	D	508		5 376					
P1: PD + EWR	**469**	**D**	**508**		**5 376**					
P2: PD + C class	593	C	374	-134	6 839	1464	-626	-1067	-1181	283 to 837
F0: Future, no EWR	433	D	528		6 386					
F1: Future + EWR	**438**	**D**	**528**		**6 386**					
F2: Future + C class	593	C	322	-206	8 125	1739	-1261	-1702	-1815	-76 to 478
C0: Future + CC	303	E	480		4 414					
C1: Future + CC + EWR	**312**	**E**	**470**		**4 414**					
C2: Future + CC + C class	593	C	205	-265	8 125	3710	-1786	-2227	-2340	1370 to 1925

8.6 Water trade-offs can be eased by improving water quality

It is very important to note that this study has only considered changes in freshwater inflows to the estuary. To some extent, there are trade-offs between water quantity and quality (DWA 2017b, Turpie & Clark in prep.). Specifically, the quantity of inflows required to achieve a certain class of health can be lowered if measures are taken to restore the quality of inflowing water. In the case of the Berg River Estuary, this is a very important consideration, since the water quality of the river inflows has deteriorated markedly. If water quality were fully restored to Reference levels (which would not be completely feasible), then the %MAR required to reach a C-class would decrease from 65% to 43% of Reference (Turpie & Clark in prep; Figure 8.3). This is equivalent to some 200 Mm^3 per annum. Thus, even partially restoring water quality, through measures to improve the quality of wastewater treatment works and agricultural return flows, could lead to a significant improvement in estuary health and value. Further work is required to determine how the costs of improving water quality compare the costs of augmenting system yield.

Both clearing of IAPs and addressing water quality issues in the catchment of the Berg River Estuary would align with global sustainable development goals to restore catchment areas and their aquatic biodiversity as well as address growing water insecurity (Abell *et al.* 2019, Flitcroft *et al.* 2019). There is also significant and increasing potential for the development of financing mechanisms to undertake such measures.

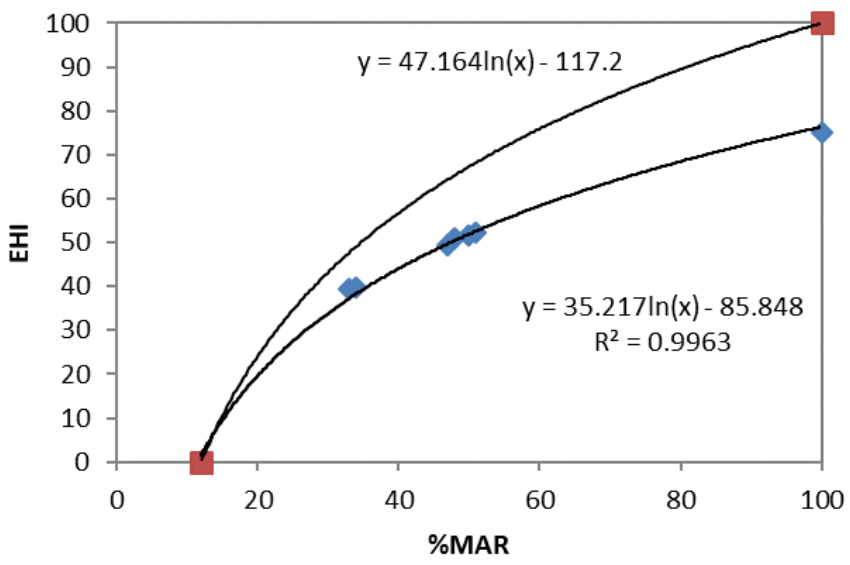

Figure 8.3. Relationship between the health of the Berg River Estuary as measured by the estuary health index (EHI) and flow in terms of percentage of natural mean annual runoff (MAR) under existing vs natural water quality conditions. Source: Turpie & Clark (in prep).

8.7 Conclusions and recommendations

The conclusions that can be drawn from the overall results are:

- **The Berg River Estuary is a highly valuable asset**. Not only is the estuary one of the most important systems in the country in terms of estuarine biodiversity, but also produces goods and services worth at least R378 million per year, not counting the value of carbon storage to the rest of the world. The annualised contribution of the estuary to property value accounts for almost half of this (R168 m per year). The asset value of the Berg River Estuary is currently just over R5 billion.

- **As at 2020, the estuary's health is classified as "largely modified" (D category)**. The last comprehensive assessment determined the estuary to be "moderately modified" (C-category). The difference is due to (a) a reassessment of the scores with additional information on hydrology, water quality, fish and birds, and (b) an actual decrease in health due to continued decreases in flows and water quality and other factors. Given the updated understanding of the estuary's hydrology and also the way it was scored in 2010, it is possible that the estuary was already in a D category in 2010.

- **The gazetted environmental water requirements (EWRs) are not being met at present**. The recent (2018) Classification study deemed that the estuary should be in a "moderately modified" (C) category and set flow requirements based on the existing understanding of the system which have been gazetted. While winter flows largely meet the requirements, summer flows fall considerably short. In addition, water quality falls well short of requirements.

- **Meeting the EWR requirements as gazetted will not be sufficient to achieve a "moderately modified" (C) category, even under historical ("present-day") climate conditions**. The relationship between freshwater inflows and estuary health from which EWRs are set has been updated as a result of new information and analyses in this study. The estuary will require more and/or better quality water than currently gazetted in order to be restored to the desired C-category of health that befits it's biodiversity conservation and socio-economic importance.

- **Without increasing the EWR allocation, climate change will reduce health to a "seriously modified" E-category**. By 2040, with the planned future developments and expected climate change, the health of the estuary will degrade to an E category (39% of Reference). This could lead to the estuary losing about 30% of its value. What is required is a higher EWR allocation, preferably for a C-class.

- **A "moderately modified" (C-category) estuary will require more water than previously estimated.** Based on this study, it is estimated that 65% of MAR would be needed to achieve a "moderately modified" C category, rather than 46% as previously estimated. In both cases this assumes no change in water quality. Restoring water quality could lower the flow quantity requirements by as much as 30%, but the relative costs of these options have not been explored here.

- **Securing more water to increase estuary health can be justified**. Under present-day conditions, supplying enough water to the estuary to restore it to a "moderately modified" C category would cost an estimated R0.6-R1.2 billion in water demand and supply measures, but this would increase the ecosystem asset value by an estimated

R1.5 billion. Under climate change, system yields would be reduced by 56%, and their replacement costs would amount to between R1.7 and R2.3 billion. However, this is less than the estimated difference in value of the estuary of around R3.7 billion. Given that the full value of the estuary goes beyond economic outputs and can never be fully quantified, these results suggest that as much as possible should be done to free up water for the estuary to allow it to recover to and remain in a C-category, as befitting its biodiversity importance and highly-regarded sense of place. Furthermore, the quantity of inflows required could be significantly lower if measures are taken to restore the quality of inflowing water. Further work is required to determine to what extent investments should be made into water quality improvement in order to achieve estuary health gains in the most cost-efficient way.

The following recommendations are made:

- Establish a rated section at the head of the estuary for **monitoring freshwater inflow** so that the actual amount of flow reaching the estuary can be recorded; given the low flows in summer, a weir is not advised as this will impact negatively on the system;
- **Update the gazetted flow requirements** to restore estuary from "largely modified" D to a "moderately modified" C-category by (a) increasing MAR to at least 55% of natural + (b) halving anthropogenic nutrient inputs from the catchment;
- Reinforce and support measures to **reduce anthropogenic nutrient inputs** from the catchment and estuary shores, through a range of measures including riparian buffers, control of agricultural return flows, higher treatment standards and polishing of treated wastewater;
- Implement measures to **increase water flows from the catchment** and thereby minimise the cost of water supply to the estuary, including clearing of invasive alien plants, better compliance monitoring of water users;
- Reinforce and **support implementation of the estuary management plan**, including
    - conservation planning to avoid excessive river bank development and protect sensitive areas;
    - securing formal protection;
    - visible patrols and better enforcement of estuary activities, particularly fishing;
    - raising public awareness of the value of conserving the system

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A1 Appendix :1 RQOs for the Berg River Estuary

Table 1.1. RQOs for the Berg River Estuary – water quantity, quality and habitat.

Component	Sub-component	Indicator	RQO Narrative	RQO Numeric													
Quantity	Surface flow	Flow	River inflow should never drop below 0.6 $m^3.s^{-1}$ and should not below 1 $m^3.s^{-1}$ for longer than 4 months; Flood frequency Should not increase/decrease by more than 10% from 2004 baseline conditions	Months	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	al
				MMR/MAR (% Natural)	31.21 (46%)	12.55 (36%)	3.92 (25%)	1.61 (19%)	1.50 (23%)	1.66 (20%)	9.13 (36%)	22.18 (26%)	64.25 (42%)	123.35 (61%)	137.15 (68%)	78.34 (63%)	486.86 (52%)
Quality	Nutrients	DIN	Inorganic nutrient concentrations not to exceed TPCs for macrophytes and microalgae	Estuary (low flows < 1 $m^3.s^{-1}$, summer): DIN <300 µg/l; DRP <100 µg/l in Zones A and B, DIN <80 µg/l ; DRP <30 µg/l in Zones C and D													
				Estuary (high flows > 5 $m^3.s^{-1}$, winter): DIN <800 µg/l; DRP <60 µg/l in Zones A-D													
		DIP		River inflow (< 1 m $^3.s^{-1}$, summer): DIN <80 µg/l; DRP <20 µg/l													
				River inflow (>5 $m^3.s^{-1}$, winter): DIN <800 µg/l; DRP <60 µg/l													
	Salinity	Salinity	Salinity distribution not to exceed TPCs for fish, invertebrates, macrophytes and microalgae	Salinity <20 for longer than 3 months at 20 km upstream from the mouth; Salinity <1 ppt above 40 km upstream of the mouth; Salinity of Salinity everywhere in estuary <35; Groundwater salinity on floodplain <45; TDS of river inflow <3500 mg/l													
	System variables	Temperature	System variables not to exceed TPCs for biota	"River inflow: 7 < pH < 8.5													
		pH		Estuary: 7 < pH < 8.5 "													
		Dissolved oxygen		"River inflow: DO >4 mg/l													
		Secchi depth		Sechii depth >1 m													
	Pathogens	Enterococci	Concentrations of	Zones A and B <1.0 m during low flow (< 1$m^3.s^{-1}$)													

		Escherichia coli	waterborne pathogens not to exceed limits considered suitable for recreational use	≤185 Enterococci/100 ml) (90th percentile, HHazen system)
				≤500 E. coli/100 ml (90th percentile, Hazen system)
Habitat	Hydrodynamics	Mouth state	Habitat health adequate for microalgae, macrophytes, invertebrates, fish, birds and recreational use	Permanently open
		Tidal variation		<10% change from present state
	Sediments	Sediment characteristics, Channel shape/size	Habitat health adequate for microalgae, macrophytes, invertebrates, fish, birds and recreational use	Bathymetry and sediment MdØ change <10% from baseline

Table 1.2. RQOs for the Berg River Estuary – biota.

Component	Sub-component	Indicator	RQO Narrative	RQO Numeric
Biota	Microalgae	Biomass and community composition of phytoplankton and benthic microalgae community	Phytoplankton biomass and composition suitable for invertebrates, fish, birds and recreational use	Blue-green algae <10% of phytoplankton cell counts, Benthic microphytobenthic < 40 mg/m^2 chlorophyll a, The frequency of dinoflagellates < 5% of the total phytoplankton counts
	Macrophytes	Extent, distribution and richness of macrophytes	Macrophyte cover and composition suitable for invertebrates, fish, birds and recreational use	Maintain the present distribution (2003-2005) and abundance of the different plant community types and estuarine habitats (intertidal mudflats with Zostera capensis 206 ha, intertidal salt marsh 499 ha, open pan 1159 ha, halophytic floodplain 1521 ha, xeric floodplain 919.1 ha, reeds and sedges 586.6 ha and sedge pan 292.5 ha), Prevent an increase in mats of macroalgae in the lower intertidal reaches, Reduce the area covered by water hyacinth (Eicchornia crassipes) in the upper reaches by 50% compared to the present state (2003-2005), Prevent an increase in size of the open pan dry areas (1159 ha in 2003-2005), Prevent a decrease in size of the sedge pan areas (293 ha in 2003-2005). Juncus maritimus, and waterblommetjies Aponogeton distachyos are present, Prevent the spread of invasive aliens in the riparian zone (e.g. Acacia mearnsii and Eucalyptus camaldulensis), Maintain intact reed and sedge stands along the banks of the estuary by ensuring that salinity is not

				greater than 20 ppt for 3 months at 20 km from the month during summer, Prevent an increase in bare ground in the halophytic and xeric floodplain habitats by maintaining the present-day flooding patterns
	Invertebrates	Macrofauna community composition, abundance and richness	Abundance and community composition of Invertebrates suitable for fish, birds	Retain present species richness, distribution of species and mix (low species abundance, high dominance) in Zones A to the middle reaches of Zone C. One or two species will always be present at high densities compared to others (e.g. Pseudodiaptomus hessei, Grandidierella sp.) in these Zones (A to C), Indicator species such as Capitella capitata, should not dominate benthic species at any site, Callianassa kraussi and Upogebia africana distribution patterns remain similar to present state.
	Fish	Fish community composition, abundance and richness	Abundance and community composition of fish community suitable for birds	Retain the full complement of estuarine resident (7 species) and estuary associated marine (5 species) present in the estuary with population sizes sufficient to ensure their persistence in perpetuity, Ensure that exotic freshwater species do not increase to levels where they can exclude any more indigenous species through predation or competitive interactions, Maintain recruitment of adult and juvenile fish at present levels. This requires maintaining sufficient flow for freshwater plume (temperature, salinity and olfactory gradient) entering the sea. This implies that there should be a significant number of 0 -1-year-old fish and no missing year classes.
	Birds	Avifauna community composition, abundance and richness	Health avifauna community contributing to conservation of avifauna species in SA	Retain at least 90% of the baseline species richness, abundance and diversity of the bird community determined using regression slope based on a 3-year running average

A2 Appendix 2: Land cover maps

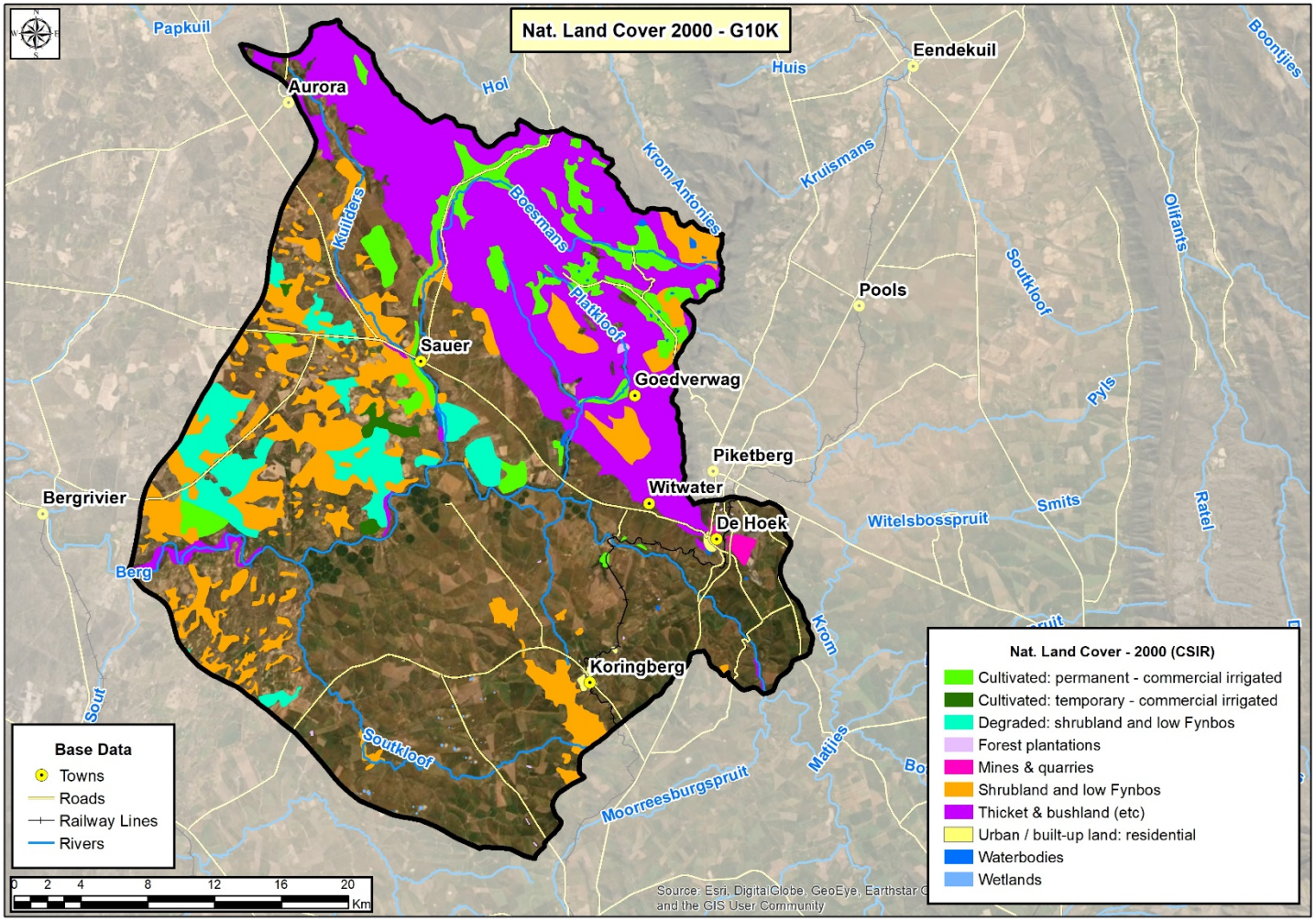


Figure 2.1. Landcover in the lower Berg River derived from the CSIR National Land Cover for 2000.

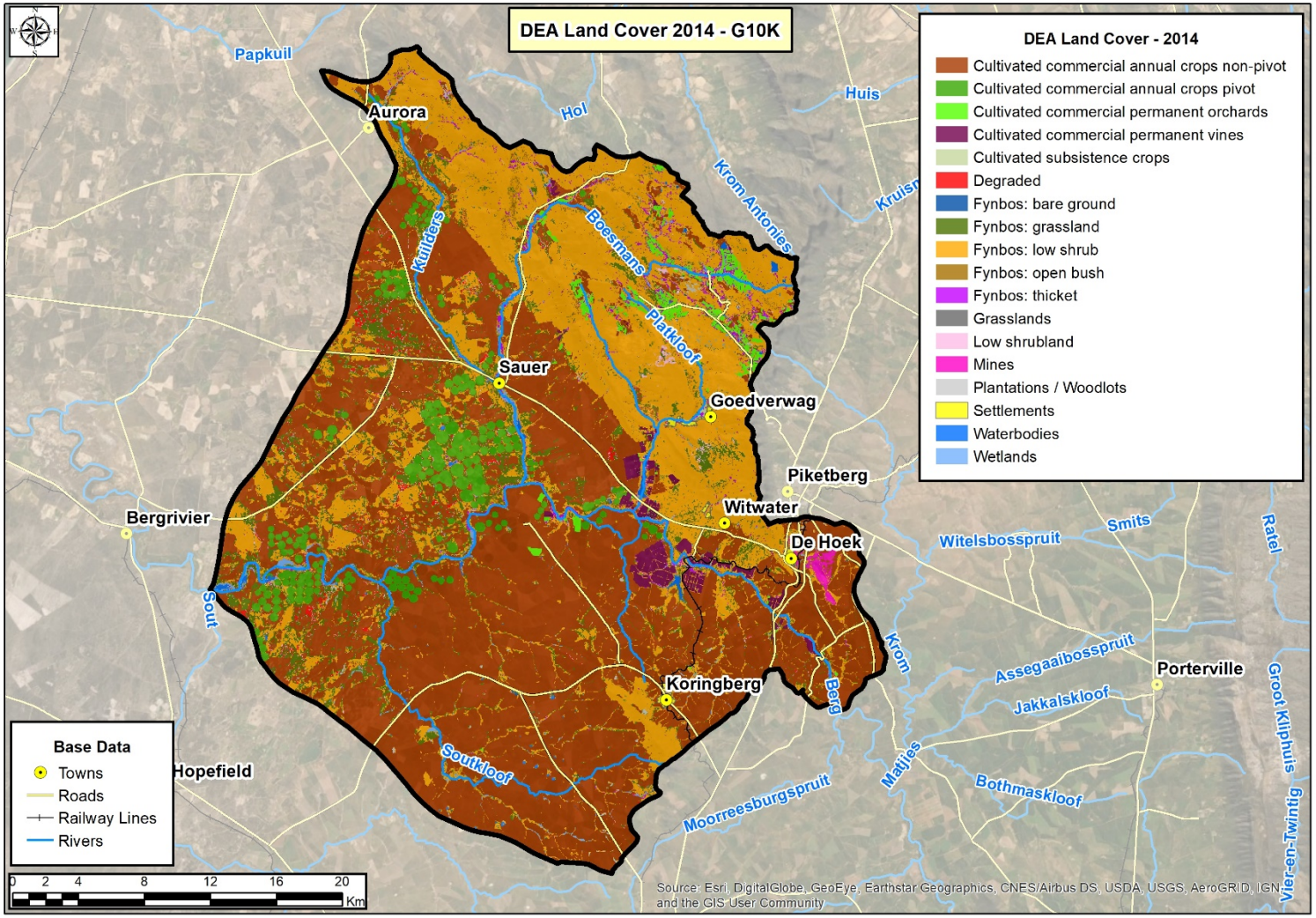


Figure 2.2. Landcover in the lower Berg River derived from the National Land Cover (NLC) for 2014.

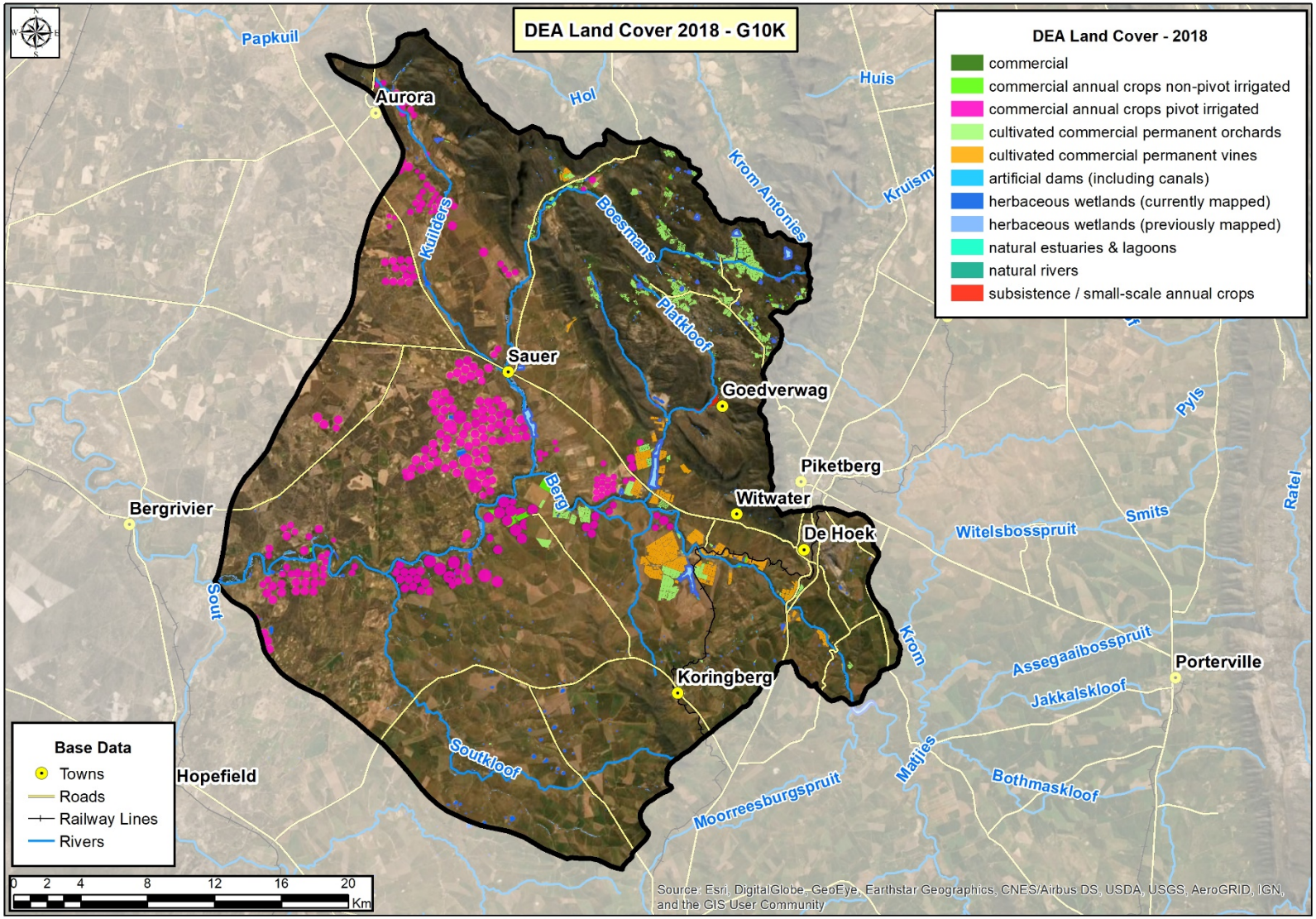
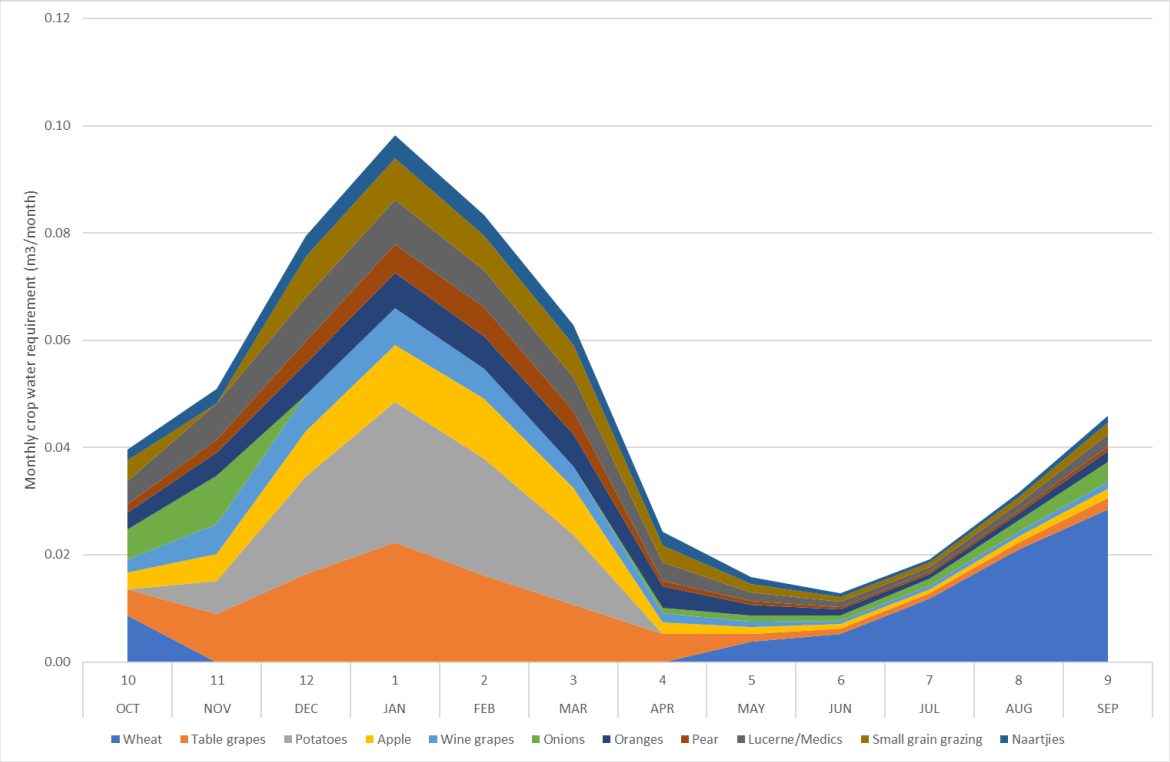

DEA Land Cover 2018 - G10K
DEA Land Cover - 2018
commercial
commercial annual crops non-pivot irrigated
commercial annual crops pivot irrigated
cultivated commercial permanent orchards
cultivated commercial permanent vines
artificial dams (including canals)
herbaceous wetlands (currently mapped)
herbaceous wetlands (previously mapped)
natural estuaries & lagoons
natural rivers
subsistence / small-scale annual crops
Base Data
Towns
Roads
Railway Lines
Rivers
0 2 4 8 12 16 20 Km
Aurora
Sauer
Goedverwag
Piketberg
Witwater
De Hoek
Bergrivier
Koringberg
Hopefield
Porterville
Papkuil
Hol
Huis
Kruism
Krom Antonies
Kuilders
Boesmans
Platkloof
Berg
Sout
Soutkloof
Moorreesburgspruit
Matjies
Krom
Witelsbosspruit
Smits
Pyls
Ratel
Assegaaibosspruit
Jakkalskloof
Bothmaskloof
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community

Figure 2.3. Landcover in the lower Berg River derived from the National Land Cover (NLC) for 2018.

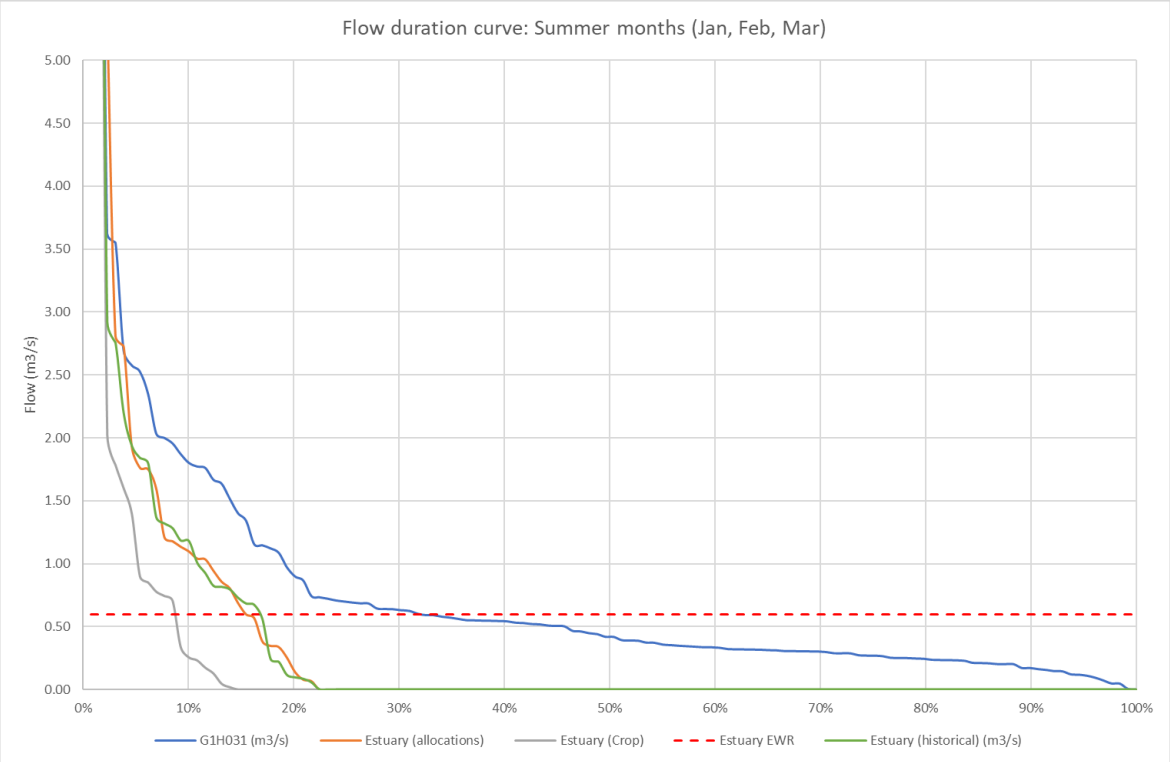
A3 Appendix 3: Historical Crop Water Requirements

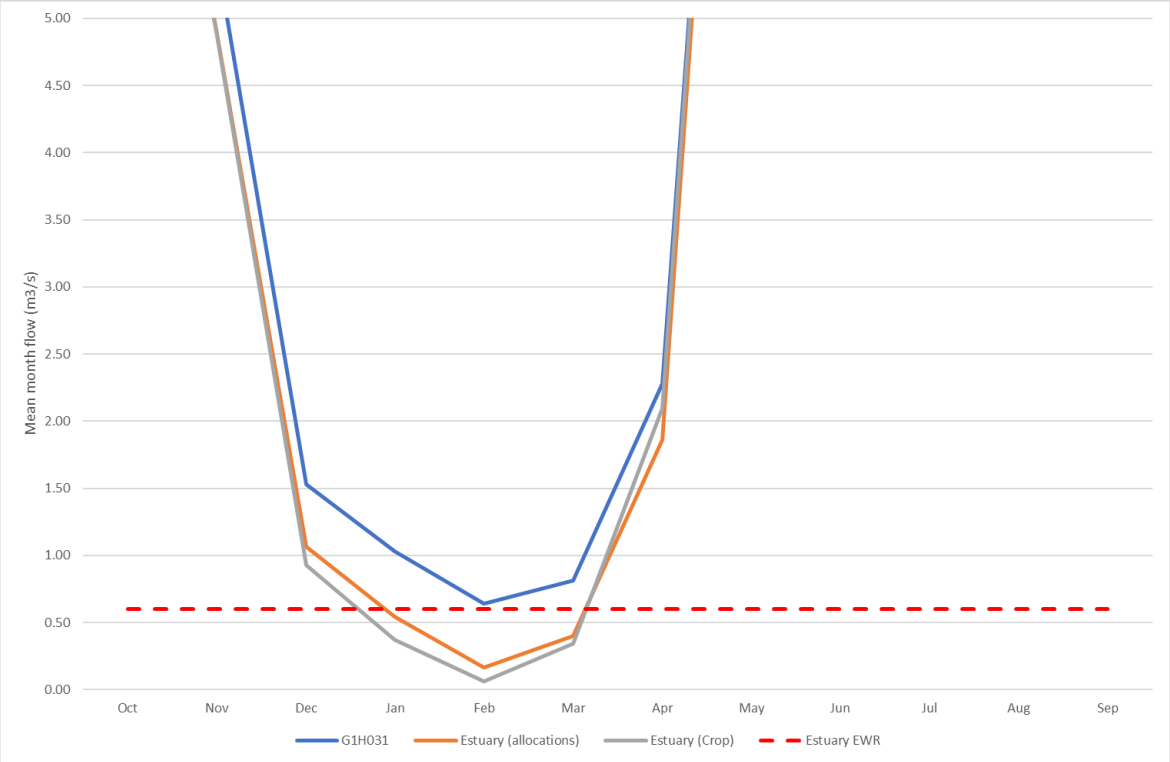
Crop type from census (2018)	Assumption (from WR90 crop factors)	Crop factors (applicable to A-pan data) Source WR90											
		OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP
Wheat	Wheat	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.34	0.79	1.00	0.87
Table grapes	Grapes (t)	0.15	0.20	0.30	0.40	0.35	0.25	0.20	0.10	0.10	0.10	0.10	0.10
Potatoes	Potatoes (m)	0.00	0.29	0.70	0.99	1.00	0.65	0.00	0.00	0.00	0.00	0.00	0.00
Apple	Deciduous	0.24	0.27	0.36	0.45	0.57	0.48	0.20	0.20	0.20	0.20	0.20	0.20
Wine grapes	intensive (trellised)	0.23	0.38	0.38	0.38	0.38	0.30	0.20	0.20	0.20	0.20	0.20	0.20
Onions	Onions	0.70	0.80	0.00	0.00	0.00	0.00	0.15	0.32	0.40	0.57	0.66	0.70
Oranges	Citrus	0.45	0.45	0.50	0.55	0.60	0.65	0.70	0.65	0.60	0.45	0.45	0.45
Pear	Deciduous	0.24	0.27	0.36	0.45	0.57	0.48	0.20	0.20	0.20	0.20	0.20	0.20
Lucerne/Medics	Lucerne for frost-free area	0.70	0.80	0.80	0.80	0.80	0.80	0.70	0.60	0.50	0.50	0.50	0.60
Small grain grazing	Pasture	0.70		0.80	0.80	0.80	0.80	0.70	0.60	0.50	0.50	0.50	0.60
Naartjies	Citrus	0.45	0.45	0.50	0.55	0.60	0.65	0.70	0.65	0.60	0.45	0.45	0.45
Peach	Deciduous	0.24	0.27	0.36	0.45	0.57	0.48	0.20	0.20	0.20	0.20	0.20	0.20
Pomegranate	Deciduous?	0.24	0.27	0.36	0.45	0.57	0.48	0.20	0.20	0.20	0.20	0.20	0.20
	G10K												
	Evap Zone 23B	8.67	12.08	14.83	15.09	12.45	11.51	7.06	4.00	2.62	2.55	3.57	5.57
	MAE (A-pan) mm	166.00	231.28	283.94	288.91	238.37	220.37	135.17	76.58	50.16	48.82	68.35	106.64
Gross water requirement (mm)		OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP
Crop type	Weighted area	10	11	12	1	2	3	4	5	6	7	8	9
Wheat	31%	28.22	0.00	0.00	0.00	0.00	0.00	0.00	12.25	17.06	38.57	68.35	92.78
Table grapes	19%	24.90	46.26	85.18	115.57	83.43	55.09	27.03	7.66	5.02	4.88	6.84	10.66
Potatoes	9%	0.00	67.07	198.75	286.02	238.37	143.24	0.00	0.00	0.00	0.00	0.00	0.00
Apple	8%	39.84	62.45	102.22	130.01	135.87	105.78	27.03	15.32	10.03	9.76	13.67	21.33
Wine grapes	6%	38.18	87.89	107.90	109.79	90.58	66.11	27.03	15.32	10.03	9.76	13.67	21.33
Onions	5%	116.20	185.03	0.00	0.00	0.00	0.00	20.28	24.51	20.07	27.83	45.11	74.65

Oranges	4%	74.70	104.08	141.97	158.90	143.02	143.24	94.62	49.78	30.10	21.97	30.76	47.99
Pear	4%	39.84	62.45	102.22	130.01	135.87	105.78	27.03	15.32	10.03	9.76	13.67	21.33
Lucerne/Medics	4%	116.20	185.03	227.15	231.13	190.69	176.30	94.62	45.95	25.08	24.41	34.18	63.99
Small grain grazing	3%	116.20	0.00	227.15	231.13	190.69	176.30	94.62	45.95	25.08	24.41	34.18	63.99
Naartjies	3%	74.70	104.08	141.97	158.90	143.02	143.24	94.62	49.78	30.10	21.97	30.76	47.99
Peach	2%	39.84	62.45	102.22	130.01	135.87	105.78	27.03	15.32	10.03	9.76	13.67	21.33
Pomegranate	2%	39.84	62.45	102.22	130.01	135.87	105.78	27.03	15.32	10.03	9.76	13.67	21.33
Gross demand (m^3)	100%	0.041	0.053	0.083	0.103	0.088	0.067	0.025	0.016	0.013	0.020	0.032	0.047


0.12
0.10
0.08
0.06
0.04
0.02
0.00
Monthly crop water requirement (m3/month)
10 OCT
11 NOV
12 DEC
1 JAN
2 FEB
3 MAR
4 APR
5 MAY
6 JUN
7 JUL
8 AUG
9 SEP
Wheat
Table grapes
Potatoes
Apple
Wine grapes
Onions
Oranges
Pear
Lucerne/Medics
Small grain grazing
Naartjies

A4 Appendix 4: Estimated estuary inflows for current crop areas





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