Quantifying land-based sources of plastic pollution in South Africa

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Declaration

This thesis reports original research that I conducted under the auspices of the FitzPatrick Institute of African Ornithology, University of Cape Town. All assistance received has been fully acknowledged. This work has not been submitted in any form for a degree at another university.

Signed by candidate

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Eleanor A. Weideman

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Abstract

It is widely claimed that at least 80% of the plastic litter entering marine environments comes from land-based sources, yet there is little empirical evidence to support this. Most studies to date predict the flux of litter from land to sea using global models based on a handful of field studies conducted mostly in developed, Northern Hemisphere countries; others use proxies such as per capita waste generation and proportion of mismanaged waste to predict litter loads entering the sea. It thus remains unclear how accurate these predictions are, particularly for African countries where few field studies have been conducted. Due to its rapidly growing human population, Africa is expected to become a much more significant source of litter into the sea. It is therefore important to identify major land-based sources of litter from the continent so as to implement effective mitigation strategies aimed at stopping this leakage of waste plastics into the marine environment.

In this thesis I investigate two possible land-based sources of plastic pollution in South Africa. In **Chapter 2** I show that stormwater run-off from Cape Town, a large coastal city, is a significant land-based source of litter into Table Bay. By placing nets over three stormwater outlets, each draining a different land-use type in the city, I estimate that some 70–630 tonnes of plastic litter are released as part of urban run-off from Cape Town each year which is a similar order of magnitude as estimates based on run-off litter collected in the 1990s and stranded beach litter. Overall, 40–78% of litter items by count and 52–64% by mass was plastic, of which most was single-use packaging. Compared to a similar survey of the same three stormwater outlets conducted in 1996, litter densities by count decreased by 50% in two of the three catchments but increased threefold in the commercial/residential area. It is clear that urban run-off is a significant source of litter into Table Bay, but the total amount is considerably less than model predictions that identified South Africa as the 11th worst source of marine plastics from land-based sources globally.

While several of South Africa's large cities are on the coast, the largest industrial centre and most densely populated part of the country is inland in the Gauteng Province. Much of this province lies within the catchment of the Orange-Vaal River system, which flows west across the country to eventually drain into the Atlantic Ocean. In **Chapter 3** I present the results of two extensive field surveys to

assess the amounts of plastic debris transported by the Orange-Vaal River system into the sea. By sampling for micro- and macro-plastics at 33 bridges spanning the lengths of both the Orange and Vaal rivers I show that densities of larger plastic items were highest at sites in the densely populated and highly urbanised upper reaches of the Vaal River, while microfibres were particularly abundant in the arid, sparsely populated lower reaches of the Orange River at the end of the wet season. It therefore appears that the Orange-Vaal River system may be a source of microfibres to the Atlantic Ocean, but most larger plastic items are retained near point sources in the upper reaches of the Vaal River. The Orange-Vaal River system thus does not appear to be a major source of plastics into the Atlantic Ocean and there is considerably less than the 0.095 tonnes·yr^{-1} predicted to be washed out of the Orange River by global models.

I also investigated whether dams retain microplastics and microfibres within their reservoirs by collecting bulk water and neuston net samples from above and below the walls of the five major dams in the Orange-Vaal River system. I present these results in **Chapter 4** and show that microplastic and microfibre concentrations were highest in dams on the densely populated Vaal River during dry conditions, whereas the opposite pattern occurred at dams farther downstream or on the sparsely populated Orange River during wet conditions. Overall, microplastic and microfibre densities were similar at sites collected above versus below dam walls and there was no significant correlation between microplastic and microfibre densities at a site and the distance from the site to the dam wall. Dams therefore do not appear to trap floating microplastics and microfibres, although the data were noisy (average CV = 184%) and so provide only a rough estimate of differences in plastic densities among sites.

Lastly, in **Chapter 5** I summarise the main results from the previous chapters and present recommendations for future research. Combining the findings from Chapters 2 and 3 I make a first, very crude estimate of the amount of plastic entering the sea from land-based sources in South Africa and compare this to the 0.09–0.25 million tonnes predicted by Jambeck et al. (2015) based on national estimates of waste production and management. I conclude that actual plastic emissions are some 1–2 orders of magnitude less than the global model estimate, which concurs with estimates based on beach litter data.

Chapter 1

General introduction:

Sources of litter into the marine environment

"Marine debris poses a continuing threat to marine ecosystems … Most of the concern has focused on debris discharged from vessels. However, there is now ample evidence that land-based discharges are a major source of marine debris."

Nollkaemper 1994

Plastic pollution was first detected in the stomachs of seabirds in 1960 (Harper & Fowler 1987; Ryan 2015) and floating in the oceans in the 1970s (Carpenter et al. 1972). It has since been found in even the most remote marine habitats such as in Arctic sea ice (Obbard et al. 2014; Bergmann et al. 2019), deep-sea sediments (Van Cauwenberghe et al. 2013) and submarine canyons (Mordecai et al. 2011; Van den Beld et al. 2017). Over 550 species of marine animals have been found to ingest or become entangled in waste plastics (Kühn et al. 2015) and marine litter has been associated with increased risk of disease in coral reefs (Lamb et al. 2018). Ingested plastics can block the digestive tracts of animals (e.g. Laist 1987; Pierce et al. 2004; Stamper et al. 2009) and can cause a false sense of satiation resulting in reduced body condition (e.g. Lavers et al. 2014). Waterborne persistent organic pollutants can adsorb to the surface of plastics and can leach into the digestive tracts of organisms in potentially toxic concentrations (Teuten et al. 2009) which can in turn be bio-magnified through higher trophic levels (Engler 2012). Light-weight, buoyant plastics such as bottles can aid the spread of invasive species (Barnes 2002) and plastic bags can smother benthic habitats, giving rise to anoxic conditions (Green et al. 2015). Due to the many negative effects they can have on the natural environment, plastics have been identified as a threat to marine biodiversity (Derraik

2002; Moore 2008; Deudero & Alomar 2015). It is therefore imperative that we identify the main sources of plastics into the marine environment so as to implement effective mitigation strategies aimed at addressing the problem.

Sources of plastic debris into the marine environment

In 1975 it was estimated that 5.8 million tonnes of litter entered the marine environment from ocean-going vessels, military operations and accidental spills (National Research Council USA 1975). At-sea dumping of plastics was subsequently banned in 1988 in terms of Annex V of the International Convention for the Prevention of Pollution from Ships (MARPOL) (Bean 1987; Maheim 1988), yet waste plastics continue to be found littering the oceans (Bergmann & Klages 2012; Van Cauwenberghe et al. 2013; Van den Beld et al. 2017; Lebreton et al. 2018; Ostle et al. 2019). While accidental spills and illegal dumping from ocean-going vessels do still occur (Lebreton et al. 2018; Ryan et al. 2019a; Schumann et al. 2019), it is widely cited that at least 80% of plastics entering the marine environment come from land-based sources (Jambeck et al. 2015). However, a lack of empirical data quantifying the flux of litter from land to sea – particularly from developing countries – means that there is still debate over the accuracy of this figure (Blettler et al. 2018; Van Emmerik et al. 2018; Blettler & Wantzen 2019). This uncertainty contributes to our inability to balance the global waste plastic budget whereby estimates of the amount of plastic entering the sea are orders of magnitude greater than is estimated to occur in marine environments (Koelmans et al. 2017; Lebreton et al. 2019).

Plastics and other anthropogenic litter can enter coastal seas from various land-based sources, one of the most significant of which is thought to be from large coastal cities (Jambeck et al. 2015). Street litter and litter in mismanaged landfills can be blown directly into the sea from these large coastal cities or be transported as part of urban run-off via waterways (e.g. Armitage 2007; Moore et al. 2011; Cable et al. 2017). It can then either wash back up on local beaches or be transported out to the open ocean (Collins & Hermes 2019). Due to its high per capita waste production (~2 kg per person per day) coupled with high estimated levels of waste mismanagement (56%), Jambeck et al. (2015) ranked South Africa as the 11th worst

offender globally – and third worst in Africa – in terms of litter entering the sea. Jambeck et al. (2015) estimated that 90 000–250 000 tonnes of plastic is released from South Africa into the sea each year, most of which comes from large coastal cities. In support of this, beach surveys around the coast of South Africa show that densities of macroplastics (Ryan 2020), microplastics (Ryan et al. 2018) and microfibres (De Villiers 2018) are highest close to large urban centres. In Cape Town, research conducted in the 1990s and early 2000s showed that large quantities of macrolitter accumulated in stormwater systems and were washed out to sea as part of urban run-off (Arnold & Ryan 1996; Marais et al. 2004). However, there is little recent information on the flux of litter from South African coastal cities into the sea and it thus remains unclear how accurately Jambeck et al. (2015) predict this.

Plastics and other anthropogenic litter can also be transported to the sea from farther inland via rivers. Macroplastics (>5 mm) can enter rivers either by being illegally dumped into the catchment or blown or washed off landfills, streets or via stormwater drains (Moore et al. 2011; Cable et al. 2017). Large quantities of microfibres and microplastics (0.025–5 mm) can be released into rivers from wastewater treatment plants (WWTPs) (McCormick et al. 2016; Leslie et al. 2017) or be aerially deposited (Free et al. 2014; Barrows et al. 2018a; Allen et al. 2019; Bergmann et al. 2019). Once litter enters a river, it can be retained within the catchment by being buried in bottom sediments (Hurley et al. 2018; Nel et al. 2018), becoming trapped in riparian vegetation (Ivar do Sul et al. 2013) or washing up on river banks (Lattin et al. 2004; Rech et al. 2014), or eventually be transported out to sea, particularly during periods of high flow (Veerasingam et al. 2016; Hurley et al. 2018). Lebreton et al. (2017) and Schmidt et al. (2017) estimated that 0.4–2.4 million tonnes of waste is released into coastal seas via the world's rivers every year. However, their global models were based on a handful of field studies conducted mostly in developed countries in the Northern Hemisphere and did not include any studies from Africa because there is virtually no information on litter loads in African rivers (Blettler et al. 2018). In South Africa, Nel et al. (2018) reported microplastics in sediments from the Bloukrans River in the Eastern Cape Province and a report by the Water Research Commission showed that microplastics were pervasive pollutants in surface waters of the Vaal River in the Gauteng Province (Bouwman et

al. 2019). De Villiers (2019) found high concentrations of microfibres in beach sediments collected from river mouths around the coast of South Africa, suggesting that these rivers act as sources of microplastics into the sea. However, no other studies have been conducted in South Africa and no studies to date have quantified macrolitter loads in any South African rivers. It therefore is unclear how accurately the global models used by Lebreton et al. (2017) and Schmidt et al. (2017) predict the flux of litter from land to sea via rivers, particularly in Africa.

Overview and outline of this thesis

Plastics are very diverse in terms of composition, colour and their physical properties, but one of the key factors that determines their impact on the environment is particle size. Size varies from megaplastics (>1 m) to nanoplastics (<1 µm), and different approaches are needed to sample across this massive size spectrum. Because size is so important, researchers tend to report data by size categories, but there is debate as to the boundaries between categories. The most common classification used in the scientific literature – and the one I have used throughout this thesis – is to classify plastics into microplastics (0.025–5 mm) and macroplastics (>5 mm) (e.g. Hoellein et al. 2017; Lebreton et al. 2017; Schmidt et al. 2017; Hurley et al. 2018). However, it should be noted that the Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection Working Group 40 recently recommended that macroplastics be split into mesoplastics (5 mm–2.5 cm) and macroplastics (2.5 cm–1 m) (GESAMP 2016). Microplastics are further classified according to their origin. Primary microplastics, such as industrial production pellets and beads in facial cleansers, are manufactured to be a specific size while secondary microplastics derive from larger items which have broken down, most often as a result of UV radiation or mechanical abrasion (GESAMP 2016). The most abundant type of secondary microplastics are microfibres derived from textiles such as polyester or nylon, although recent research has shown that many microfibres found in the environment are of natural origin such as wool or cotton (GESAMP 2016; Stanton et al. 2019, Suaria et al. 2020).

In this thesis I use a variety of different methods to investigate two possible land-based sources of plastic pollution in South Africa. In **Chapter 2,** I repeat a 1996

survey of three stormwater outlets in Cape Town to assess whether run-off from this large coastal city is a significant land-based source of macrolitter into Table Bay. I report how much and what types of litter are found in urban run-off and use these data to estimate how much litter washes out of Cape Town annually. By comparing my data to the 1996 survey, I can assess the extent to which the amount and composition of street litter has changed over the last two decades, during which time the population of Cape Town has increased by 80%.

In **Chapter 3** I present the results of two extensive field surveys which I undertook to the Orange-Vaal River system, the largest and most turbid river system in South Africa. I collected surface water samples and conducted observations from 33 bridges along the lengths of both rivers and sampled each site twice, first at the end of the wet season and then at the end of the dry season. The Orange River flows west across the country from the highlands of Lesotho to the arid Northern Cape where it delivers large amounts of sediment into the Atlantic Ocean. Its catchment is relatively sparsely populated and little urbanisation or industrialisation has occurred on its banks. Conversely, the Vaal River, which is the largest tributary into the Orange River, drains the southern half of the Gauteng Province, which is the most densely populated and highly urbanised and industrialised area of South Africa. If plastics are most abundant in the upper and middle reaches of the Vaal River, it would suggest that plastics are retained close to urban source areas rather than being transported out to sea. Conversely, if plastics are evenly distributed downstream from this area, it would suggest that there is effective transport over long distances.

The Orange-Vaal River system also has five major dams, two of which lie between Gauteng and the sea. In **Chapter 4** I assess whether these dams trap and retain plastics within their catchments by investigating whether floating microplastics and microfibres are (1) more abundant at sites closer to dam walls than sites farther away and (2) at sites above versus below the dam walls. If so, it would suggest that dams do trap microplastics and microfibres and thus prevent them being transported out to sea.

Finally, in **Chapter 5**, I synthesise the results from the preceding chapters, and make a first, very crude estimate of the amount of plastic entering the sea from

land-based sources in South Africa. This is compared to the 90 000–250 000 tonnes predicted by Jambeck et al. (2015) based on national estimates of waste production and management.

I have written the main chapters (Chapters 2-4) as stand-alone documents to facilitate publication in peer-reviewed journals and there is therefore some repetition in the introduction and methods sections. Versions of Chapters 2, 3 and 4 have already been published; details of the publications and contributions made by the other co-authors on these papers are included at the start of each chapter. Except for some fieldwork and assistance with data analysis, the work presented here is my own. Vonica Perold assisted with the collection of litter from the three stormwater outlets during a time when litter loads were particularly high (Chapter 2) and Kyle Maclean helped to sort and weigh the litter from one of the rain events. Vonica Perold, Anna Blaylock and Thobile Dlamini accompanied me on field trips to the Orange and Vaal rivers (Chapter 3 and 4) and my supervisor Prof. Peter Ryan assisted with data analysis and editing of drafts.

Chapter 2

Quantifying litter loads in urban stormwater run-off in different areas of land-use in Cape Town, South Africa

Me at the mixed commercial/residential stormwater outlet in Cape Town before emptying the net. Photo: V. Perold

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This chapter has been published:

Weideman, E. A., Perold, V., Arnold, G. & Ryan, P. G. (2020) Quantifying changes in litter loads in urban stormwater run-off from Cape Town, South Africa, over the last two decades. *Science of the Total Environment*, **724**: 138310. https://doi.org/10.1016/j.scitotenv.2020.138310.

Author contributions: EAW, VP, GA and PGR planned the field work; EAW, VP and GA conducted the field work; EAW and GA processed all samples in the lab; EAW analysed the data with assistance from PGR; EAW wrote the draft; PGR assisted with manuscript edits/preparation; VP contributed to drafts.

Abstract

Most of the litter entering the oceans is thought to come from land-based sources, particularly in run-off from coastal cities. In order to implement effective mitigation strategies to stop this flow of litter from land to sea, it is important to identify key pollution sources. I sampled plastic and other anthropogenic debris in urban stormwater run-off by placing nets over three stormwater outlets in Cape Town, South Africa (7.5 x 7.5 mm mesh size). The outlets were selected to represent three different land-use types: residential, industrial and a mixed commercial/residential area. Sampling was conducted during seven rain events (11 days) and confirmed there was trivial run-off on days when it did not rain. Overall, organic matter constituted 79% of material by mass (industrial: 51 ± 22%, commercial/residential: 86 ± 4%, residential: 88 ± 13%). The nets caught 3.2–373 litter items·ha^{-1}·day^{-1} (1.5–244 g·ha^{-1}·day^{-1}), with significantly higher densities of litter found in the industrial and mixed commercial/residential areas than the residential area. Plastic litter constituted 40–78% of items by count and 52–64% by mass and most plastic items were single-use packaging such as sweet wrappers, chip packets and softdrink bottles. Industrial pellets (“nurdles”) washed out of the industrial outlet during every rain event. Compared to similar surveys of the three stormwater outlets in 1996, litter densities by count halved in the industrial and residential areas but increased threefold in the commercial/residential area. The proportion of plastics was broadly similar to 1996 except in the residential area where the proportion of plastic by mass increased from 20% to 83%. When extrapolated across the City of Cape Town, I estimate that 70–630 tonnes of plastic are released from stormwater outlets in Cape Town each year. While this is orders of magnitude less than predicted by global models, it is nevertheless clear that stormwater outlets are a significant source of litter, particularly plastics, into the sea. Intercepting this waste before it reaches the sea would greatly reduce litter loads in coastal waters around Cape Town.

Introduction

Approximately 80% of the litter entering the global oceans is predicted to come from land-based sources (Jambeck et al. 2015), much of which is thought to be transported by wind or water from large coastal cities (Jambeck et al. 2015; Axelsson & Van Sebille 2017). Jambeck et al. (2015) estimated that 4.8–12.7 million tonnes of plastic waste is released into the sea from coastal areas annually and identified poor waste management as one of the key determinants of how much waste is released. Due to its high per capita waste production (~2 kg per person per day) and poor waste management (an estimated 56% of waste is deemed to be mismanaged), Jambeck et al. (2015) ranked South Africa as the 11th worst contributor of litter from land-based sources and estimated that 0.09–0.25 million tonnes are released from South African coastal areas into the sea annually. However, their predictions are based on measures of per capita waste production, waste management strategies and human population rather than empirical field data. It therefore remains unclear how accurate their predictions are and these uncertainties likely contribute to the large mismatch between the 5–12 million tonnes of plastic predicted to be released into the sea from land-based sources annually (Jambeck et al. 2015) and the ~250 000 tonnes found floating at sea (Eriksen et al. 2014).

The main way in which litter is transported from coastal cities to the sea is via urban run-off through stormwater drainage systems (either directly into the sea, or into rivers draining urban areas). Stormwater systems are designed to quickly transport large volumes of rainwater from streets in order to prevent flooding. However, these systems can also collect and move large quantities of waste from gardens, roofs, footpaths, streets and parking lots. In Europe, it is common for stormwater and sewer systems to be combined so that stormwater is screened and treated in wastewater treatment plants (WWTPs) before being released into the natural environment (Armitage 2007). However, in South Africa and many other parts of the world, stormwater and sewer systems are separated (although poor maintenance does mean that sewage and stormwater run-off can mix occasionally; Armitage 2007). Urban stormwater run-off in South Africa is therefore rarely screened or processed before flowing into the natural environment. Any street litter or litter that has accumulated in the stormwater system is therefore likely to be washed directly out to sea, or into wetlands, rivers or canals where it can be picked

up (often at great financial cost), become stuck in riparian vegetation, buried in sediments, washed up on river banks or eventually be carried out to sea (Lattin et al. 2004; Armitage 2007; Ivar do Sul et al. 2014; Rech et al. 2014).

Several studies conducted in South Africa in the 1990s and early 2000s showed that stormwater systems were significant sources of litter into the environment (Arnold & Ryan 1999; Armitage & Rooseboom 2000a; Marais et al. 2004; Armitage 2007). In Cape Town, litter loads ranged from 0.2–104 $kg \cdot ha^{-1} \cdot year^{-1}$ in residential areas and from 23–111 $kg \cdot ha^{-1} \cdot year^{-1}$ in the Central Business District (Marais et al. 2004). Another study conducted in Cape Town in 1996 estimated low levels of litter in run-off from an upper-income residential area (6.4 $items \cdot ha^{-1} \cdot day^{-1}$, 12.7 $g \cdot ha^{-1} \cdot day^{-1}$), slightly higher loads in a mixed commercial/residential area (29.2 $items \cdot ha^{-1} \cdot day^{-1}$, 23.8 $g \cdot ha^{-1} \cdot day^{-1}$) and much higher litter loads in an industrial area (731.2 $items \cdot ha^{-1} \cdot day^{-1}$, 383.3 $g \cdot ha^{-1} \cdot day^{-1}$) (Arnold & Ryan 1999). When organic matter was excluded, plastics constituted 74–90% of litter (20–69% by mass) and were the most common type of litter in all three catchments (Arnold & Ryan 1999). While the authors presented their findings to local communities and municipalities and an efficient passive screening system was designed (Armitage & Rooseboom 2000b; Armitage & Rooseboom 2000c; Armitage 2007), no significant mitigation measures were implemented. To assess whether these stormwater systems are still releasing large quantities of litter into Table Bay, I resurveyed the three outlets sampled by Arnold & Ryan (1999) in 1996 by placing nets over each outlet during storm events. I estimate how much litter is released into the environment from each of these catchments, to identify what kinds of litter are found and to assess whether the proportion of plastics has increased since 1996. I also crudely extrapolate the total amount of plastic waste generated by Cape Town each year and compare this to predictions used to estimate African and global marine plastics inputs (Jambeck et al. 2015, 2018).

Methods

Site description

Cape Town is a winter (May–September) rainfall area and storms tend to be short and intense, with most rain falling over a few hours or days. Water flow through the stormwater drainage system is therefore sporadic and large volumes of litter and vegetation can accumulate within the system during dry periods. When it rains, large volumes of water carrying heavy loads of litter and organic material wash out of the stormwater outlets. This makes sampling urban run-off difficult as nets must have a small enough mesh size to trap litter, must be strong enough to withstand high water pressure and at the same time must not cause flooding. Arnold & Ryan (1999) therefore restricted their sampling to relatively small catchment areas (2–6 ha).

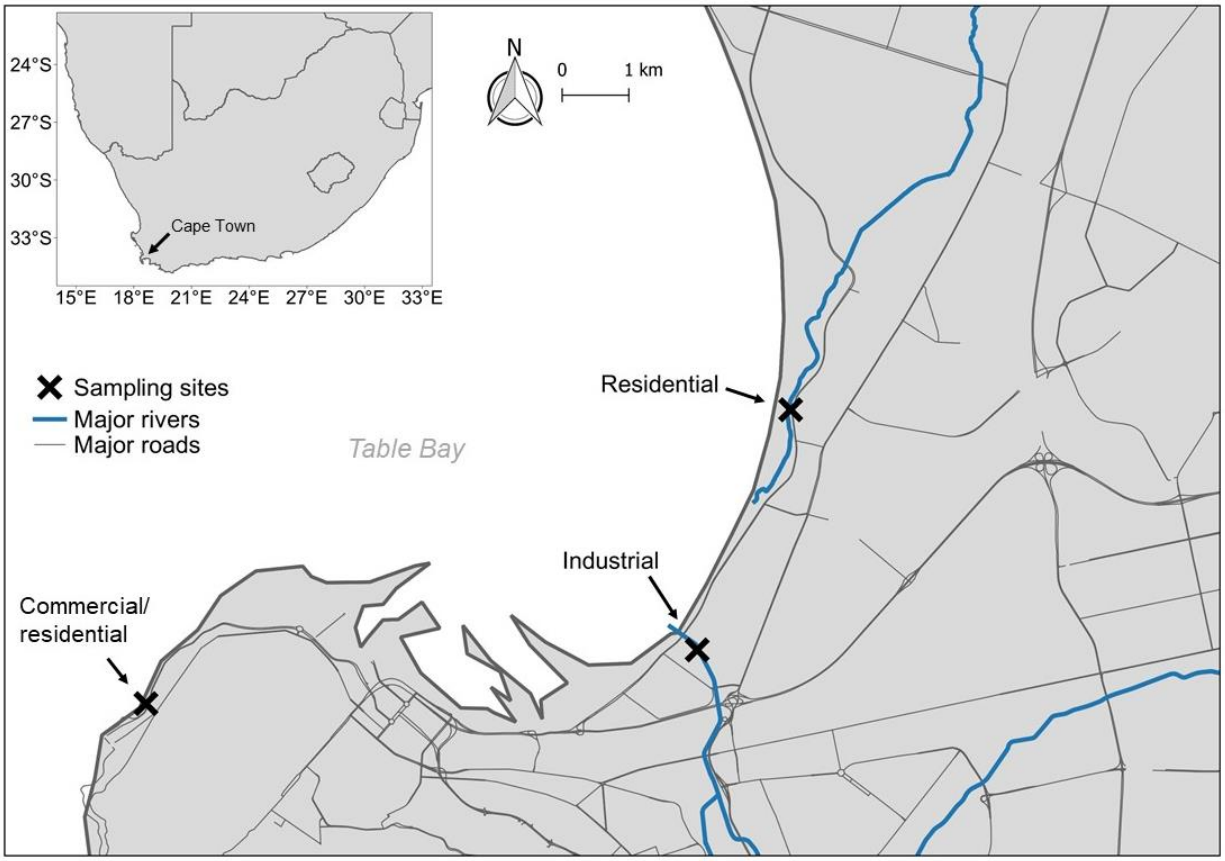

Figure 2.1: Maps of the study area showing the three stormwater outlets in the City of Cape Town.

Field protocol

I resampled the three stormwater outlets surveyed by Arnold & Ryan (1999), which together represent the main land-use areas of Cape Town (Figure 2.1). The largest catchment area is in Sea Point (6.24 ha, outlet location: 33.9172°S 18.3859°E), 20% of which drains commercial shopping areas while the rest is comprised of middle- to high-income high-density residential accommodation (mostly high-rise blocks of flats). This mixed commercial/residential stormwater outlet flows directly onto a ~40 m wide beach and into Table Bay. The industrial catchment (2.5 ha, outlet location: 33.9104°S 18.4751°E) is in Paarden Eiland which is characterised by a mix of light industrial factories and wholesale outlets. The industrial stormwater outlet flows into the Salt River approximately 500 m from where the river drains into Table Bay. This section of the Salt River is canalised and, while there are some small patches of vegetation within the canal, it is unlikely that much of the litter entering the canal will become trapped in the vegetation and most will be washed out to sea, especially during periods of high flow. The residential catchment area is in Milnerton (4.7 ha, outlet location: 33.8780°S 18.4904°E) which is a medium- to high-income, low-density area (free-standing houses). This stormwater outlet flows into the Diep River approximately 1 500 m from the river mouth, which also drains into Table Bay. However, this river has not been canalised and so some litter may become buried in bottom sediments or trapped in vegetation.

I secured nets over the three outlets by locking them to stainless steel bolts inserted into the wall on either side of the outlet. The outlets were circular pipes and were approximately 0.5 m wide in the residential and industrial sites and 0.9 m wide in the mixed commercial/residential site. I used the same polypropylene netting and net frames as used by Arnold & Ryan (1999) except in the mixed commercial/residential area where I used a slightly larger net frame (47 cm diameter in 1996, 67 cm diameter in 2018/19). I used the larger frame in the mixed commercial/residential area as the smaller frame used by Arnold & Ryan (1999) did not completely cover the outlet and I wanted to ensure that I caught all of the litter. The nets had a 7.5 × 7.5 mm mesh, although smaller litter items were often caught when they became trapped by vegetation and other larger litter items. I deployed the nets for 17 days in winter (August–September 2018 and May–September 2019) of which six were dry days (i.e. no rainfall) while the rest constituted seven separate

storms (average of 24.7 ± 18.2 mm of rain per event, range 11–65 mm). Four rain events lasted one day, two lasted two days and one lasted three days. For each rain event, I deployed the nets at least six hours before the onset of the storm, checked them daily for the duration of the storm to ensure that they were not overly full, and emptied them once the storm had passed.

All material caught in the nets was returned to the University of Cape Town where it was oven dried at approximately 50°C for 24–48 hours until it was completely dry. Thereafter it was sorted and weighed to the nearest 0.1 g. I classified litter items according to the type of material: plastics, cigarette butts, rubber, glass, metal, cloth, worked wood and other (paint pieces, wax and other miscellaneous items). I further classified all plastic items according to their function and type: single-use packaging (e.g. food wrappers, polystyrene packing chips and take-away clam shells, bags, bottles, lids), disposable items (e.g. earbud sticks, straws, cable ties), user items (e.g. toys, stationery), industrial pellets, miscellaneous hard pieces/fragments and other (items whose origin and/or use were unidentifiable such as pieces of glue, pieces of synthetic fluff/foam or plastic sheets).

Statistical analyses

I used Generalised Linear Models (GLMs; R Core Team 2019) to assess which factors determine the number of litter items and the mass of litter caught during each rain event. For the number of litter items, I fitted the GLM with a quasi-Poisson distribution to account for the data being over-dispersed. For the total mass of debris (organic matter and litter items) I used a Gamma distribution to account for the semi-continuous data. Both models included the area of each catchment as an offset. I considered site type (residential vs industrial vs mixed commercial/residential), the amount of rain which fell during each rain event (mm) and the number of days since last rain as independent variables. Rainfall was measured in Rondebosch (33.9689°S 18.4846°E), 6–10 km from the three study catchments.

To assess if there has been a change in the types of litter washing out of the stormwater outlets, I compared the proportion of material types as well as the origin of plastics found in 2018-19 to that found in 1996 by Arnold & Ryan (1999) which I recalculated from their raw data. Arnold & Ryan (1999) deployed the nets for five

days in the industrial catchment, seven days in the residential catchment and for 22 days in the mixed commercial/residential catchment.

To extrapolate how much plastic litter washes out of the City of Cape Town annually, I used two slightly different approaches. For the first method, I used plastic loads measured per rain event (n=7) while for the second I used plastic loads measured per number of days of rain (n=11). For the first method, I multiplied the mass of plastic per rain event (excluding organic matter and non-plastic litter) caught in the three catchment areas by the average number of rain events experienced by the City over the past five years (39.4 ± 4.7 times as measured at the Newlands weather station [33.9596°S 18.4500°E] from 2015–2019, taken from CSAG 2019). I then multiplied the mass·ha^{-1}·yr^{-1} in each of the catchments by the total area each catchment type covers in the settled area of the City of Cape Town metropolitan area, of which 52% is residential (30 110 ha), 10% is industrial (5 836 ha) and 4% is commercial (2 066 ha; City of Cape Town Data Portal 2018). The remaining 34% of the area of Cape Town is designated as open spaces such as Table Mountain National Park and other recreational areas which I excluded from the analyses as these are unlikely to act as significant sources of litter. To get a total estimate for the City of Cape Town, I then added the estimated plastic litter in run-off from each of the three land-use types. To get a lower estimate, I multiplied the mass of plastic per rain event by the lower 95% confidence interval of the average number of rain events in Cape Town per year (mean – 2 × standard error). Similarly, to get an upper estimate I used the upper confidence intervals (mean + 2 × standard error). For the second method, I followed a similar protocol except litter loads were measured as mass·ha^{-1}·days of rain^{-1} and used the average number of days where more than 5 mm of rain fell over the past five years (measured at the Cape Town Yacht Club weather station by SAWS for 2015–2019).

Litter loads in urban run-off are often much higher in low-income, high-density residential areas than in high-income, low-density residential areas (Marais et al. 2004; Armitage 2007). I did not sample in low-income residential areas and therefore incorporated estimates of litter loads in low-income areas in Cape Town using data from Marais et al. (2004). They sampled litter in various suburbs of Cape Town over a two-year period by placing traps in catchpit openings. They found the highest litter loads in a low-income, high-density residential area (Ocean View, 104 $kg \cdot ha^{-1} \cdot year^{-1}$ of which 16 $kg \cdot ha^{-1} \cdot year^{-1}$ was plastic) which was 18 times higher than in a medium-income residential area (Summer Greens, 5.8 $kg \cdot ha^{-1} \cdot year^{-1}$, 1 $kg \cdot ha^{-1} \cdot year^{-1}$ of plastic) and 520 times higher than in a high-income residential area (Welgemoed, 0.2 $kg \cdot ha^{-1} \cdot year^{-1}$, no plastic was found here). Based on this, I estimated litter loads in low-income areas as 18 (best case scenario) and 520 times (worst case scenario) higher than I found in Milnerton (0.2 $kg \cdot ha^{-1} \cdot yr^{-1}$ including all litter, 0.1 $kg \cdot ha^{-1} \cdot yr^{-1}$ of plastic). Given that litter loads in Milnerton were similar to Welgemoed, I estimated litter loads in medium- to high-income areas as 29 times higher than in Milnerton based on the ratio between Summer Greens and Welgemoed (Marais et al. 2004). I then estimated the annual flux of litter out of Cape Town for a range of scenarios by modifying the proportion of the residential area of Cape Town that is low-, medium- and high-income.

Results

Litter densities

Organic matter (mainly vegetation, but also dead animals, faeces, etc.) constituted 79% of material caught by mass and made up a larger proportion of litter in the commercial/residential (86 ± 4%) and residential (88 ± 13%) catchments where there is an abundance of trees and gardens than in the industrial catchment (51 ± 22%) where there are few trees and no gardens. Excluding organic matter, 30 048 litter items (mean across all three sites = 169.2 ± 187.7 $items \cdot ha^{-1} \cdot day^{-1}$) weighing 18.4 kg (mean across all three sites = 106.5 ± 124.2 $g \cdot ha^{-1} \cdot day^{-1}$) were caught in 2018-19. Litter was found in the nets during all seven rain events but only one litter item was found on dry days: a single sweet wrapper in the mixed commercial/residential area. The number of items caught during rain events ranged from a single cigarette butt

(0.21 g) in the residential area on 22 June 2019 (22 mm of rain) to 4 183 items caught in the industrial area on 17 August 2018 (17 mm of rain). The largest mass of litter (4.3 kg) was caught in the industrial area on 22 June 2019 (22 mm of rain). All litter caught in the industrial area on this day was covered in a thin film of black oil which suggests that stormwater outlets also act as sources of other pollutants into Cape Town's waterways. The oil also may have aided the retention of small litter items on this day.

Overall, litter densities were higher in the industrial (373 items·ha^{-1}·day^{-1}, 244 g·ha^{-1}·day^{-1}) and mixed commercial areas (132 items·ha^{-1}·day^{-1}, 74 g·ha^{-1}·day^{-1}), than in the residential area (3.2 items·ha^{-1}·day^{-1}, 1.5 g·ha^{-1}·day^{-1}) (Table 2.1, Table 2.2, Table 2.3). The amount of rain and number of days since last rainfall were not significant factors in explaining the variation in either the number of litter items or mass of debris (Table 2.1, Table 2.2). Compared to Arnold & Ryan (1999), litter densities by count decreased by ~50% in the industrial and residential areas but increased by 355% in the mixed commercial/residential area (Table 2.3). When measured by mass, litter densities in the residential area decreased by 88%, and by 36% in the industrial area, but increased by 208% in the commercial/residential area (Table 2.3). The large increase in the litter loads in the mixed commercial/residential catchment may be explained in part by the larger net frame which I used as well as the higher proportion of days on which Arnold & Ryan (1999) caught litter in the industrial (all five days) and residential (six out of seven days) areas than in the commercial/residential area, where they caught litter on only 9 of 22 days (41%). Excluding the 13 days on which no litter was caught, litter loads in 1996 were 71 items·ha^{-1}·day^{-1} and 58 g·ha^{-1}·day^{-1}. While this still suggests an increase in litter densities in this catchment between 1996 and 2018-19, the increase is more modest (86% by count and 28% by mass).

Litter composition

Overall, plastics were the most common type of material both numerically (64% of all litter) and by mass (52% of litter) while worked wood made up the second largest proportion by mass (18%). Most plastics were single-use packaging such as pieces of polystyrene, chip packets and bottles (66% by count, 62% by mass).

In the industrial area, 78% of litter was plastic (52% by mass) of which most were pieces of single-use packaging (58% by count, 54% by mass) (Figure 2.2, Figure 2.3). Of the packaging, 72% (23% by mass) was polystyrene, most of which were small pieces (<10 mm and <5 g) and appeared to be pieces of take-away trays, clam shells, coffee cups and packing chips. Industrial pellets ("nurdles") used in the manufacturing of other plastic items were found in the industrial catchment during every rain event (mean: 598.4 ± 1 129.6 pellets·rain event^{-1}, range 7–3 121, with the highest number on 6 August 2018). Pellets accounted for 34% of plastic items by count in this catchment and were doubtless underestimated due to their small size (2–4 mm diameter) allowing most to pass through the relatively coarse-mesh net. In the mixed commercial/residential area, most litter items were plastic (49% by count, 52% by mass) or cigarette butts (37% by count, 13% by mass). Of the plastic, most were single-use packaging (82% by count, 74% by mass) (Figure 2.3). Similarly, plastics (40% by count, 64% by mass) and cigarette butts (42% by count, 19% by mass) were the most common type of litter found in the residential area (Figure 2.2). Again, the most common type of plastic was single-use packaging (54% by count, 52% by mass) followed by miscellaneous hard plastic pieces (34% by count, 39% by mass) (Figure 2.3).

The largest change in the composition of plastics between 1996 and 2018-19 was found in the residential area where the proportion of plastics by mass increased from 16% in 1996 to 64% in 2018-19 (Figure 2.2). This was driven by an increase in the proportion of miscellaneous hard pieces (7% by mass in 1996, 39% in 2018-19) (Figure 2.3). However, the proportion of plastics by count at this site remained relatively similar (41% in 1996, 40% in 2018-19). In the mixed commercial/residential area, the proportion of plastics by count increased from 40% in 1996 to 49% in 2018-19, driven by a near-doubling of the proportion of plastic single-use packaging (48% in 1996 to 82% in 2018-19 by count) (Figure 2.2, Figure 2.3). Conversely, the proportion of plastics by mass in the mixed commercial/residential area decreased slightly (from 57% to 52%). In the industrial area, the proportion of plastics decreased slightly between 1996 (85% by count, 54% by mass) and 2018-19 (78% by count, 52% by mass) (Figure 2.2).

Table 2.1: GLM results showing the effect (± standard error) of site type, amount of rain (mm) and number of days since last rainfall on the number of litter items caught in the nets during seven rain events. Catchment size was included as an offset and estimates are presented on the log-scale. The model explained 66% of the variation.

	Estimate (± SE)	**t-value**	**P**
Site type [a]			
Industrial	4.122 ± 1.548	2.663	**0.017**
Mixed	3.994 ± 1.539	2.595	**0.019**
Rain (mm)	-0.012 ± 0.012	-0.999	0.333
Days since last rain	0.035 ± 0.057	0.620	0.544
Intercept	3.690 ± 1.554	2.374	**0.030**

[a] Estimates indicate expected litter densities in the given site relative to the reference site (residential)

Table 2.2: GLM results showing the effect (± standard error) of site type, amount of rain (mm) and number of days since last rainfall on the total mass of debris including and excluding organic debris. Catchment size was included as an offset and estimates are presented on the log-scale. The model explained 37% of the variation when organic debris was included and 72% when it was excluded.

	Estimate (± SE)	**t-value**	**P**
Including organic debris			
Site type [a]			
Industrial	1.463 ± 0.596	2.457	**0.026**
Mixed	2.569 ± 0.493	5.207	**<0.001**
Rain (mm)	-0.007 ± 0.016	-0.437	0.668
Days since last rain	-0.015 ± 0.084	-0.174	0.864
Intercept	6.770 ± 0.533	12.701	**<0.001**
Excluding organic debris			
Site type [a]			
Industrial	4.523 ± 0.489	9.251	**<0.001**
Mixed	4.245 ± 0.405	10.481	**<0.001**
Rain (mm)	0.007 ± 0.013	0.522	0.609
Days since last rain	-0.024 ± 0.069	-0.348	0.733
Intercept	2.767 ± 0.438	6.323	**<0.001**

[a] Estimates indicate expected litter mass in the given site relative to the reference site (residential)

Table 2.3: Densities of litter caught in nets placed over stormwater outlets in three different land-use types: residential, industrial and mixed commercial/residential, in Cape Town, South Africa, in 2018-19, compared to the total litter load in 1996 (from Arnold & Ryan 1999).

	n·ha^{-1}·day^{-1}			Mass g·ha^{-1}·day^{-1}		
	Resid.	**Comm.**	**Indust.**	**Resid.**	**Comm.**	**Indust.**
Plastic	1.3	64.5	291.1	1.0	38.4	126.0
Single-use packaging	0.7	53.0	168.1	0.5	28.4	67.6
Bags/packets	0.1	10.6	10.8	0.0	6.4	9.4
Bottles/tubs	0.0	0.5	1.3	0.1	8.5	17.0
Cigarette wrapping	0.1	7.7	5.2	0.0	0.9	0.9
Foamed plastics	0.2	6.0	121.1	0.1	1.7	15.4
Food wrap	0.2	18.3	17.1	0.2	6.8	11.3
Lids/lid rings	0.0	1.6	4.4	0.0	2.0	6.8
Other packaging	0.0	7.6	6.5	0.0	1.2	5.4
Tape	0.0	0.7	1.8	0.0	0.9	1.4
Disposable	0.1	3.6	5.7	0.1	3.4	17.0
User	0.0	1.1	2.4	0.0	3.2	30.2
Industrial pellets	0.0	0.0	98.6	0.0	0.0	2.6
Misc. hard pieces	0.4	6.5	16.0	0.4	2.2	8.0
Other	0.0	0.2	0.3	0.0	0.1	0.4
Cigarette butts	1.4	47.8	45.0	0.3	9.1	9.9
Rubber	0.0	0.3	0.4	0.0	0.2	0.4
Glass	0.0	0.3	4.8	0.0	5.2	3.4
Metal	0.0	2.6	3.3	0.0	3.2	12.8
Cloth	0.0	0.3	0.4	0.0	4.5	4.8
Card/paper	0.5	13.2	23.6	0.3	5.2	10.9
Wood	0.0	1.8	1.9	0.0	0.9	73.9
Other	0.0	0.5	2.3	0.0	6.9	2.4
TOTAL 2018-19	**3.2**	**132**	**373**	**1.5**	**74**	**244**
TOTAL 1996	**6.4**	**29**	**731**	**13**	**24**	**383**
% change	**-50**	**+355**	**-49**	**-88**	**+208**	**-36**

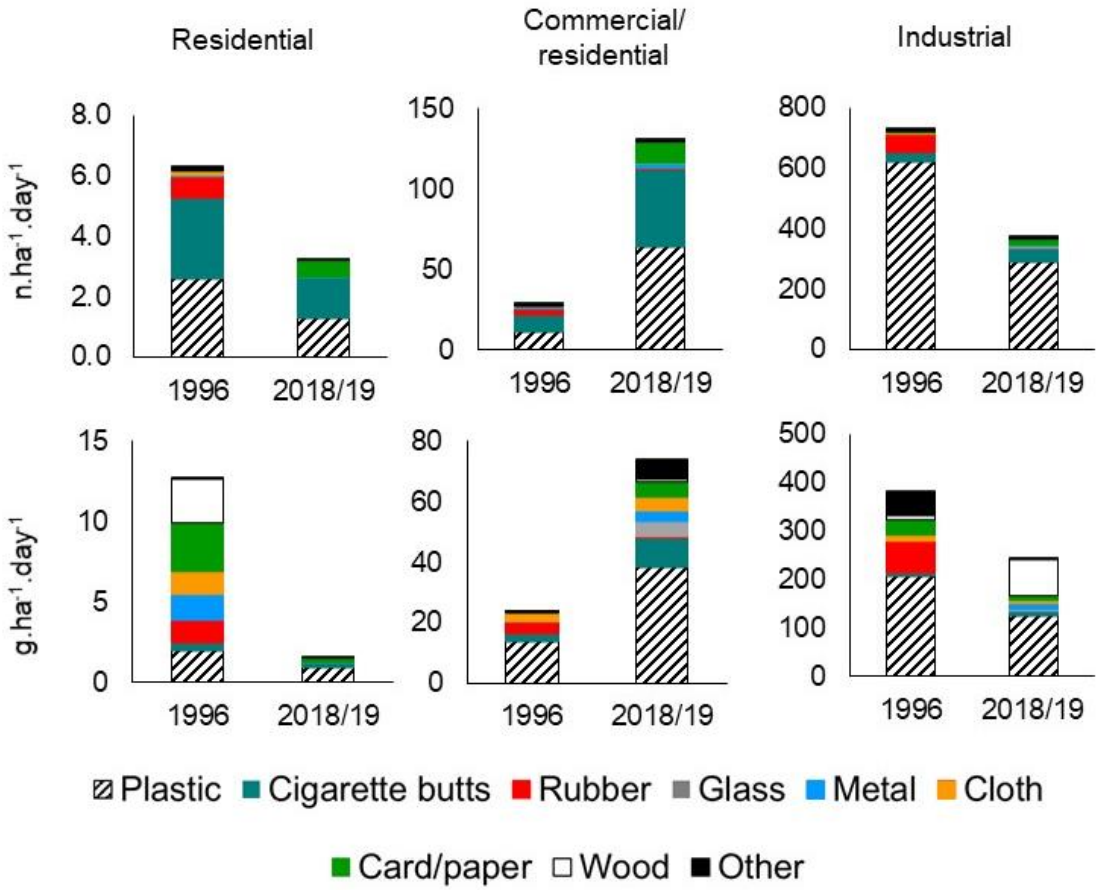

Figure 2.2: Densities of litter and the proportions of different materials by count ($n \cdot ha^{-1} \cdot day^{-1}$) and mass ($g \cdot ha^{-1} \cdot day^{-1}$) caught in nets deployed over three stormwater outlets in a residential, industrial and mixed commercial/residential area in Cape Town in winter 1996 and 2018-19. Note the different scales on the y-axes.

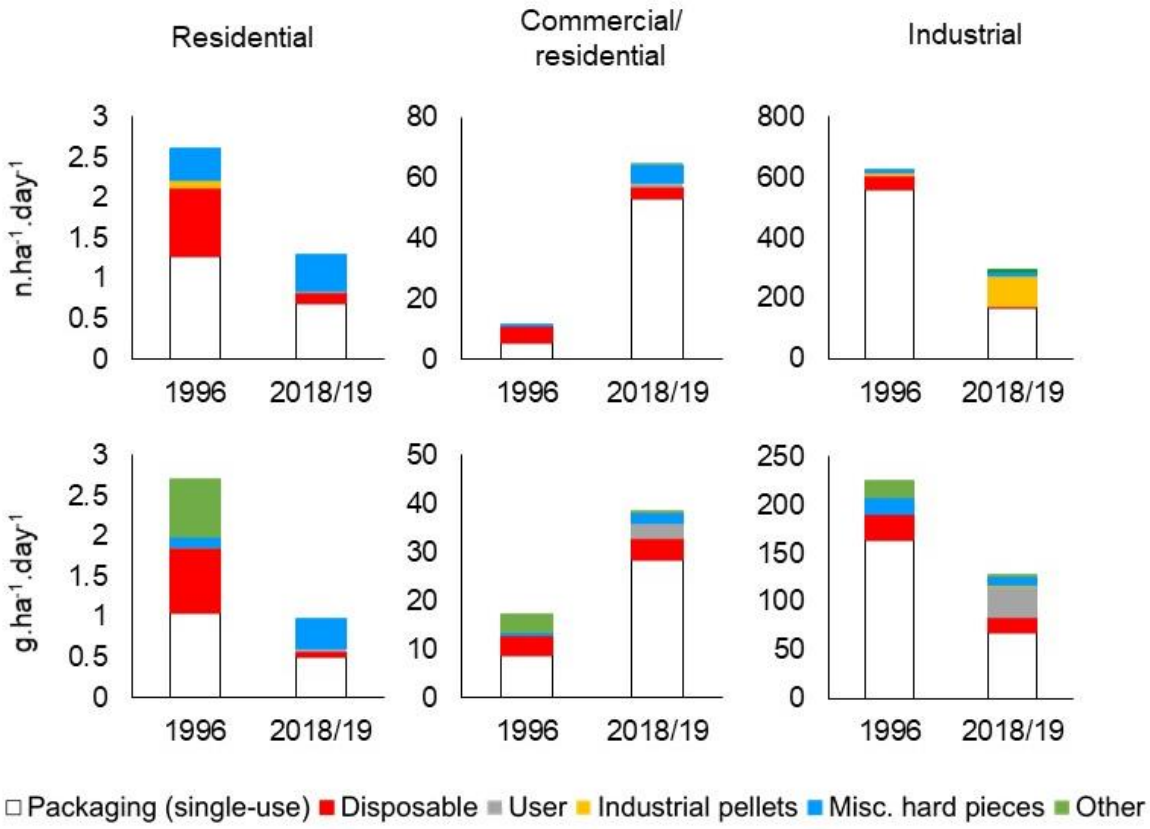

Figure 2.3: Densities of plastic and proportions of each plastic type by count ($n \cdot ha^{-1} \cdot day^{-1}$) and mass ($g \cdot ha^{-1} \cdot day^{-1}$) caught in nets deployed over three stormwater outlets in a residential, industrial and mixed commercial/residential area in Cape Town in winter 1996 and 2018-19. Note the different scales on the y-axes.

Amount and mass of litter released annually

Assuming the best scenario (that litter in low-income residential areas is ~18 times higher than in Milnerton), the annual flux of plastic into the sea from Cape Town is estimated at 120–140 tonnes$\cdot yr^{-1}$ when calculated according to the number of rainfall events per year and 40–50 tonnes$\cdot yr^{-1}$ when using the number of rain days. Together, these give an average estimate of 70–100 tonnes$\cdot yr^{-1}$ (Table 2.4). However, Marais et al. (2004) suggest that litter loads in low-income residential areas may be as much as 520 times higher than in middle- to high-income areas. Under this scenario, the annual flux of plastic increases to 420–900 tonnes$\cdot yr^{-1}$ when measured per rain event and 150–350 tonnes$\cdot yr^{-1}$ using numbers of days of rain. When taken together, 70–630 tonnes$\cdot yr^{-1}$ are estimated to wash out of Cape Town annually (Table 2.4).

Table 2.4: Annual plastic litter loads (tonnes·yr^{-1}) in urban run-off from Cape Town estimated using data presented in this chapter and that from Marais et al. (2004). A range of estimates are presented based on varying ratios of low-, medium- and high-income residential areas. Estimates presented here are the average of two methods, the first of which quantified annual loads based on the number of rainfall events per year while the second used number of days of rain (>5 mm).

Residential income ratios	**Worst scenario [a]**	**Best scenario [b]**	**Marais et al. (2004)**
25% low, 50% medium, 25% high	320 (280–360)	80 (70–90)	180
33% low, 33% medium, 33% high	410 (360–450)	80 (75–90)	220
50% low, 25% medium, 25% high	570 (510–630)	90 (80–100)	300

[a] Assuming litter loads in the low-income residential are 520 times higher than in the medium-income area

[b] Assuming litter loads in the low-income residential are 18 times higher than in the medium-income area

Discussion

Beach clean ups around the South African coast conducted since 1994 showed that plastic pollution was highest close to large urban centres, suggesting that most macro- and larger microplastic litter derives from local, land-based sources (Ryan et al. 2018; Ryan 2020). In Cape Town, a recent survey of litter on five beaches found that most litter items showed signs of weathering and having been transported by water (Chitaka & Von Blottnitz 2019). The authors therefore concluded that the litter was unlikely to have been left by beach goers, but rather to have been transported via stormwater outlets and rivers into the sea and washed back up on local beaches (Chitaka & Von Blottnitz 2019). The results of this chapter support the notion that urban run-off from the City of Cape Town is a significant source of litter into the sea, with particularly high litter loads found in the industrial and mixed commercial/residential areas.

Litter composition

The most common type of plastic in all three catchments was single-use packaging, particularly food packaging such as chip packets, sweet wrappers and softdrink bottles, as well as foamed plastics like polystyrene trays and clam shells. Similarly,

food packaging is often one of the most common types of litter found on beaches, both in Cape Town (Ryan et al. 2014a; Chitaka & Von Blottnitz 2019) and other parts of the world (e.g. Ivar do Sul et al. 2011; Mobilik et al. 2014; Munari et al. 2016; Nelms et al. 2017) and it was recently estimated that 32% of plastic packaging produced globally is leaked into the natural environment (World Economic Forum 2016).

Other prominent litter items found in my study were cigarette butts and industrial pellets, both of which increased in abundance since 1996. The high abundance of cigarette butts in urban run-off is particularly worrying as discarded cigarette butts release toxic chemicals such as arsenic, nicotine and heavy metals into the environment, many of which are harmful to aquatic organisms (Micevska et al. 2006; Moriwaki et al. 2009; Slaughter et al. 2011). Industrial pellets are also ingested by many marine animals (e.g. Ryan 1987; Robards et al. 1995; Gregory 2009) and can act as a transport medium for legacy persistent organic pollutants such as PCBs and DDT (Mato et al. 2001; Rochman et al. 2013). Industrial pellets made from polyethylene and polypropylene float in sea water and can be dispersed over long distances (Schumann et al. 2019); the negative effects of this pollution are therefore not only seen on a local scale, but further afield as well. The plastics industry launched education programmes in the early 1990s to reduce the loss of industrial pellets during the manufacturing and transport process (e.g. Operation Clean Sweep, www.opcleansweep.org) and there has been a subsequent reduction in the abundance of pellets on beaches in New Zealand (Gregory & Ryan 1997) and a decrease in the proportion of pellets ingested by short-tailed shearwaters (*Ardenna tenuirostris*) in the North Pacific (Vlietstra & Parga 2002) and the mass of pellets ingested by northern fulmars (*Fulmarus glacialis*) stranded on beaches in Holland (Van Franeker et al. 2005). Similarly, there has been a marked decrease in the proportion of pellets ingested by seabirds (Ryan 2008) and juvenile loggerhead turtles (*Caretta caretta*) (Ryan et al. 2016) off southern Africa. However, despite the apparent decrease in the abundance of pellets in the area (Ryan et al. 2020), the presence of pellets in urban run-off in Cape Town shows that not all plastic manufacturers are effectively containing spillages.

Temporal change in litter densities

The population of Cape Town increased by 80% between 1996 and 2018-19 (World Population Review 2020). It is therefore reassuring that litter densities decreased by ~50% in the industrial and residential catchments during this period. It is unlikely that street cleaning efforts have increased over the past few years; the decrease in litter loads in these two catchments is therefore more likely due to an increased awareness about the problem of littering. Conversely, litter loads were three times higher in the mixed commercial/residential catchment although this increase was more modest (86% by count and 28% by mass) when only days on which litter was caught are included. To ensure I caught all litter, I used a slightly larger net frame in the mixed commercial/residential catchment than Arnold & Ryan (1998) which might explain some of the observed increase. Alternatively, it is more likely that this is due to changes in the social demographics of the area, the proliferation of informal vendors, an increase in population density as well as a decrease in the frequency of street cleaning (Armitage 2007).

Comparison to other studies

The proportions of materials and litter loads for the different areas of land-use sampled in this chapter are broadly similar to other studies. For example, Allison & Chiew (1995) sampled stormwater run-off in Coburg, Melbourne, Australia and showed that organic matter comprised 85% of debris by mass in stormwater run-off in a residential area, 67% in a mixed commercial/residential area and 36% in an industrial area. Similarly, I found that the organic matter component was highest in the residential area (88 ± 13%), followed by the mixed commercial/residential area (86 ± 4%) and lowest in the industrial area (51 ± 22%). When organic debris was excluded, Allison & Chiew (1995) found the annual litter load for the residential area to be 0.5 kg·ha^{-1}·yr^{-1} which is similar to my study (0.05–0.15 kg·ha^{-1}·year^{-1}) and the high- (Welgemoed, 0.2 kg·ha^{-1}·year^{-1}) and medium-income (Summer Greens, 5.8 kg·ha^{-1}·year^{-1}) residential areas sampled by Marais et al. (2004). However, litter was several orders of magnitude higher in the low-income, high-density residential area of Ocean View (104 kg·ha^{-1}·year^{-1}) (Marais et al. 2004). Marais et al. (2004) also sampled in a light industrial site (Montague Gardens) and found 49 kg·ha^{-1}·year^{-1},

which is of the same order of magnitude as what I found in Paarden Eiland (9–23 kg·ha^{-1}·year^{-1}). By comparison, Cornelius et al. (1994) found much lower litter loads in an industrial area of Auckland, New Zealand (0.9 kg·ha^{-1}·year^{-1}). Litter loads in the mixed commercial/residential area of Sea Point (3–7 kg·ha^{-1}·year^{-1}) were similar to that found in a commercial area of Auckland, New Zealand (1.3 kg·ha^{-1}·year^{-1}) (Cornelius et al. 1994) but lower than was found in Springs, a mixed commercial/industrial area near Johannesburg, South Africa (82 kg·ha^{-1}·year^{-1}) (Armitage et al. 1998), as well as the commercial Central Business District of Cape Town where litter loads varied between 42–111 kg·ha^{-1}·year^{-1} in 2000 and 23–59 kg·ha^{-1}·year^{-1} in 2001 (Marais et al. 2004).

Comparison to global models

Using estimates of per capita waste production, the proportion of this waste that is plastic, the efficiency of national waste management strategies and the human population within 50 km of the coast, Jambeck et al. (2015) estimated that roughly 90,000–250,000 t of plastic were washed into coastal seas from land-based sources in South Africa in 2010. Beach surveys around the South African coast have shown that plastic is concentrated around the four major coastal urban centres of Cape Town, Port Elizabeth, East London and Durban (Ryan et al. 2018; Ryan 2020). This empirical data matches model predictions of where plastic from these coastal cities is likely to wash ashore (Collins & Hermes 2019). It can therefore be inferred that most of the plastic entering coastal areas from land-based sources comes from these cities. Approximately 63% of the South African coastal population live in these four urban centres, of which 40% live in Cape Town (Collins & Hermes 2019). Cape Town should thus account for at least 25% of the 90,000–250,000 t of plastic thought to enter coastal seas from South Africa (i.e. 23,000–63,000 t·yr−1 according to Jambeck et al. 2015). However, my sampling of stormwater outlets, coupled with the data presented by Marais et al. (2004), suggests that the annual flux of plastic from Cape Town is b1,000 t·yr^{-1} (Table 4). This much more modest estimate of coastal plastic inputs is consistent with estimates of the amount of plastic stranding along the South African coast. Chitaka & von Blottnitz (2019) found that litter accumulated at five beaches in Cape Town at an average daily rate of 18 ± 17 (range: 1.3–44)

$kg \cdot km^{-1} \cdot day^{-1}$ of which 95–99% was plastic. If crudely extrapolated along the 300 km of coast in the City of Cape Town (Department of Environmental Affairs 2016) and assuming 20–30% of litter released out of Cape Town strands on local beaches (Collins & Hermes 2019), this equates to some 2,300–2,500 t of litter washing out of Cape Town annually. Of course, some litter strands farther away, but the density of stranded litter decreases rapidly with distance from Cape Town, and litter standing stocks on beaches are much lower than expected from the Jambeck et al. (2015) global model prediction (Ryan 2020). Together, these results suggest that the annual flux of litter from land to sea in Cape Town is at least an order of magnitude lower than predicted by Jambeck et al. (2015). My findings thus help to explain the large discrepancy between the millions of tonnes of plastic predicted to wash into the sea from global land-based sources each year (Jambeck et al. 2015) and the more modest ~250,000 t estimated to be floating in the world's seas (Eriksen et al. 2014) without invoking rapid sinking of most floating litter to account for the 'missing' plastic (Koelmans et al. 2017).

Estimates of litter loads in urban run-off are however dependent on the mesh size used in litter traps. I used a 7.5 x 7.5 mm mesh which may have underestimated the abundance of smaller items. However, it was not possible to use a smaller mesh size as the large litter loads which accumulate during dry periods combined with the high water pressure exerted on the nets during rain events would likely have broken the nets. Furthermore, while the mesh was coarse, some small items were caught in the net (e.g. pellets were found washing out of the industrial catchment during each rain event) suggesting that even a coarse mesh size can give some indication of the abundance of smaller litter items. On the other hand, even if only macrodebris (>20 mm) were caught, these account for >90% of the mass of plastics both on beaches (Ryan 2020) and at sea (Lebreton et al. 2018) suggesting that a finer mesh size is unlikely to improve estimates of annual litter loads by more than 10%.

Solutions

Many possible solutions to curb plastic pollution in Cape Town have been suggested by other researchers, most extensively by Armitage (2007). Many of these can be implemented relatively easily. One possible solution is to focus on reducing the

amount of litter entering the drainage system. This can be done by improving street cleaning practices (e.g. increasing the frequency of street cleaning efforts and ensuring street cleaners avoid sweeping litter into stormwater drains), placing more bins in areas with particularly high litter loads or installing grates over catch pit openings. Education is also a powerful tool, for example Marais et al. (2004) found that litter loads in stormwater catch pits in the low-income area of Ocean View were significantly reduced over a two-year period and attributed this to the local community becoming more sensitive to the problem of littering (Armitage 2007). Alternatively, litter traps designed to trap litter once it has already entered the stormwater system can be installed (Armitage 2007). Some traps have been installed in areas of Cape Town, for example a simple litter boom recently installed in the Black River and the grating over the stormwater outfall in the harbour. However, there are many more places where similar traps can be implemented, for example there is no boom downstream of the industrial stormwater outlet which I sampled and so all the litter washing out of the outlet will likely be carried out to sea. Another solution would be to transition from a linear economy – where plastic packaging is used once and discarded – to a more circular economy model where plastic items such as packaging are designed to either be used more than once or are recycled (Jambeck et al. 2018). If these measures are implemented, the amount of litter entering Table Bay via urban run-off can be drastically reduced.

Conclusion

I resurveyed three stormwater outlets initially surveyed in 1996 by Arnold & Ryan (1999) to assess whether urban run-off is a significant source of litter into Table Bay. Litter loads were ~50% lower in the industrial and residential catchments than in 1996 but increased threefold in the mixed commercial/residential catchment. Most litter was single-use plastic packaging and cigarette butts although large quantities of industrial pellets were found washing out of the industrial stormwater outlet. Despite litter loads being much lower than predicted by global models (Jambeck et al. 2015), it is clear from the results of this chapter that urban run-off is a significant source of litter into the sea.

Chapter 3

Longitudinal assessment of micro- and macroplastic pollution in the Orange-Vaal River system, South Africa

The neuston net used to sample for larger microplastics seen here in the Upper Vaal River. Photo: K. van Schie

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Author contributions: EAW, VP and PGR planned the field work; EAW and VP conducted the field work; EAW processed all microplastic samples in the lab; EAW analysed the data with assistance from PGR; EAW wrote the draft; PGR assisted with manuscript edits/preparation; VP contributed to drafts.

Abstract

Much of the plastic waste entering the sea is thought to be transported from land by rivers. However, most studies to date have been conducted in Europe, North America and Asia, whereas little is known about plastic pollution in African rivers. To address this knowledge gap and to improve our understanding of how plastics are transported by rivers, I collected surface water samples from 33 bridges on the Orange-Vaal River, the largest river system in South Africa. I sampled each site at the end of the wet season (April 2018) and at the end of the dry season (November 2018). At each site, I collected bulk water samples to test for microplastic and microfibre (0.025–1 mm) pollution and conducted observations for macrodebris (>50 mm). Where possible, I also sampled for larger microplastic fragments (5–25 mm) using a neuston net. Microfibres and microplastics were found at every site (1.7 ± 5.1 L^{-1}, >99% fibres) and accounted for 99% of the number of items found. Microfibres and microplastics were particularly abundant in the arid, sparsely populated lower reaches of the Orange River during the period of low flow prior to the wet season flush. By comparison, macrodebris and larger microplastics were orders of magnitude less abundant (observations: 0.0002 ± 0.0007 items·m^{-2}; neuston net: 0.34 ± 0.93 items·m^{-2}). However, at sites where larger items were found, they comprised most of the mass of litter. Larger plastics were found mostly at sites in the upper reaches of the densely populated and highly urbanised Vaal River. The results of this chapter suggest that, while the Orange-Vaal River system may be a source of microfibres to the Atlantic Ocean, larger plastic items typically only travel short distances from source and are retained in the upper reaches of the Vaal River. The Orange-Vaal River system therefore does not appear to be a major source of plastics into the Atlantic Ocean, at least under regular flow conditions.

Introduction

At-sea dumping of plastics has been banned for 30 years (MARPOL Annex V 1988), yet waste plastic is still a major pollutant of marine environments (Bergmann & Klages 2012; Van Cauwenberghe et al. 2013; Van den Beld et al. 2017; Lebreton et al. 2018; Ostle et al. 2019) and continues to be identified as a threat to marine biodiversity (Derraik et al. 2002; Moore 2008; Deudero & Alomar 2015). While accidental spills and illegal dumping from ocean-going vessels still occur (Ryan et al. 2019a; Schumann et al. 2019), it has been estimated since the 1990s that 80% of the plastic entering coastal seas originates from land (Jambeck et al. 2015). Much of this plastic is thought to be transported into the sea by rivers. For example, it was recently estimated that 0.4–2.4 million tonnes of waste plastic is released into coastal seas by the world's rivers annually (Lebreton et al. 2017; Schmidt et al. 2017). However, there are large uncertainties associated with estimates from these global models, which might contribute to the much greater estimates of plastic entering the sea than is estimated to occur in marine environments (Koelmans et al. 2017; Lebreton et al. 2019). Uncertainties in global river inputs are mostly due to the paucity of field measurements, particularly from rivers in developing countries (Blettler et al. 2018; Van Emmerik et al. 2018; Blettler & Wantzen 2019). For example, Lebreton et al. (2017) and Schmidt et al. (2017) based their global emission models on a handful of litter studies conducted mostly in North America, Asia and Europe. Neither used any empirical data collected from rivers in Africa, despite the apparent growing importance of Africa as a major source of plastics into the sea (Jambeck et al. 2018). It therefore remains unclear how accurately global models such as those of Lebreton et al. (2017) and Schmidt et al. (2017) predict the flux of waste plastics from land to sea via rivers, particularly in Africa (Blettler et al. 2018).

Plastics can enter a river catchment via various pathways (Figure 3.1). Plastics can be blown or washed into a river from landfills, streets or via stormwater outlets and are thus often most abundant close to urban centres with large human populations (Moore et al. 2011; Cable et al. 2017). Macroplastics (>5 mm) can eventually break down into microplastics (0.025–5 mm) due to UV radiation and mechanical abrasion (Corcoran et al. 2009). Microplastics, particularly microbeads

from personal care products and microfibres from clothing (although these can also be of natural origin) (Stanton et al. 2019), can also be released into rivers from wastewater treatment plants (WWTPs) and are therefore often most abundant close to urban centres where WWTPs tend to be concentrated (Leslie et al. 2017; McCormick et al. 2016). However, microplastics and microfibres can also be transported in the atmosphere over long distances and have been found in remote areas far from large cities (Free et al. 2014; Barrows et al. 2018a; Allen et al. 2019; Bergmann et al. 2019). Microplastics, and especially microfibres, are prevalent in sewage sludge, which is often used to fertilise croplands and which can be washed into rivers (e.g. Nizzetto et al. 2016a). In some agricultural areas, thin polyethylene sheets are placed over the soil to limit weed growth; these break down into small fragments within a few months and are often washed into rivers (Rillig 2012; Zhang et al. 2019). Some waste plastics in rivers are transported into coastal seas (Rech et al. 2014; Lebreton et al. 2017; Schmidt et al. 2017) but some are retained within river catchments either by being buried in bottom sediments (Hurley et al. 2018; Nel et al. 2018), stranded on river banks (Lattin et al. 2004; Rech et al. 2014) or becoming entangled in riparian vegetation (Ivar do Sul et al. 2014). However, these might only be temporary sinks, with plastics washed out to sea during periods of high flow or floods (Veerasingam et al. 2016; Hurley et al. 2018).

Most sampling of plastic pollution in rivers has been done by towing nets with mesh sizes of ~300 µm (e.g. Faure et al. 2015; Mani et al. 2015; Baldwin et al. 2016). While these nets are ideal for sampling larger microplastics, their coarse mesh size means they underestimate the abundance of smaller microplastics, particularly microfibres (Barrows et al. 2017; Lahens et al. 2018; Ryan et al. 2019b), whereas their small frames mean that they fail to sample larger macroplastics adequately (Blettler et al. 2018; Lebreton et al. 2018; Van Emmerik et al. 2018; Blettler & Wantzen 2019). This is problematic because microfibres tend to be the most abundant form of plastic pollution, while macroplastics make up the largest proportion of waste plastics by mass (Eriksen et al. 2014; Blettler et al. 2018; Lahens et al. 2018). Global models based on existing literature therefore likely underestimate the amount and mass of plastic entering coastal seas via rivers (Van Emmerik et al. 2018). For example, Lebreton et al. (2017) base their global model on studies which sampled with nets with mesh sizes of 330–3 200 µm.

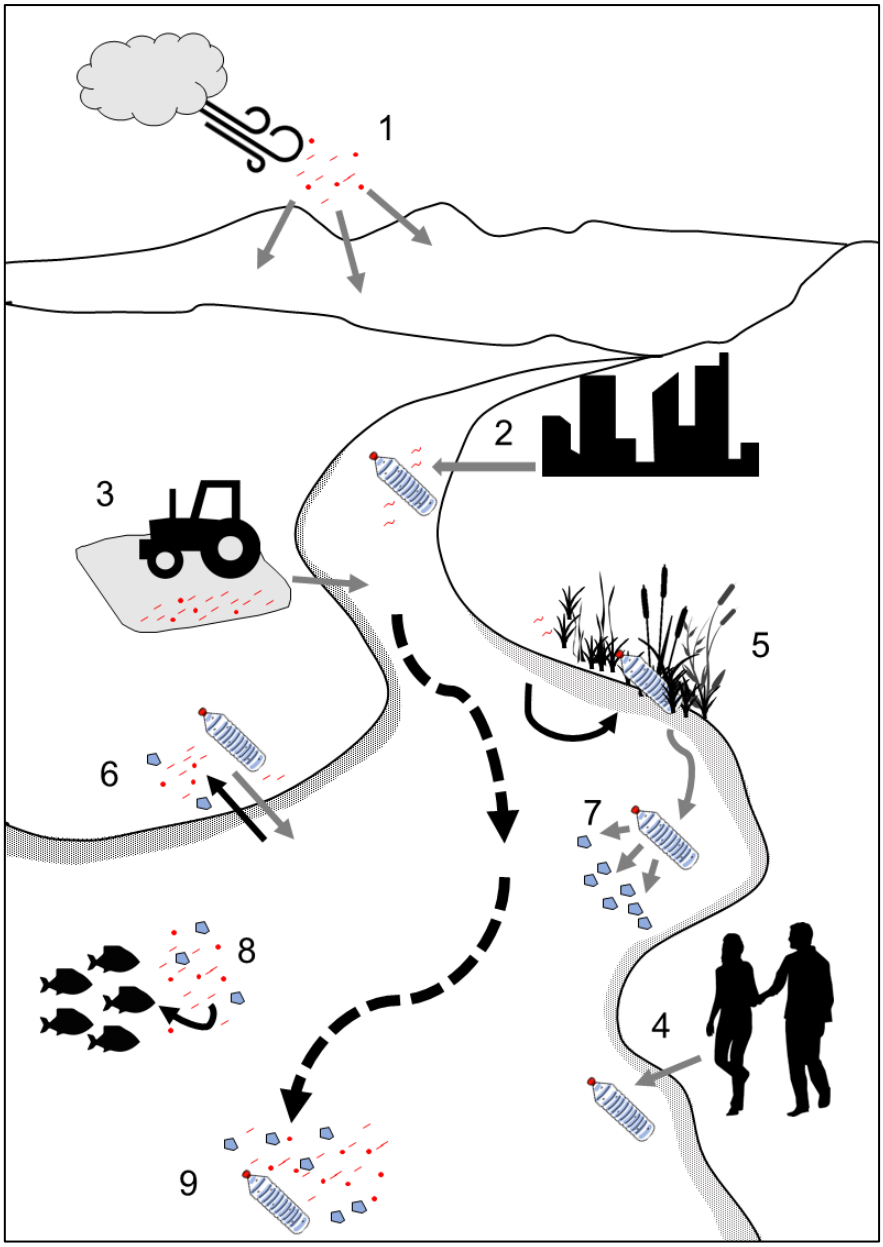


Figure 3.1: Schematic diagram showing the movement of plastics through a river catchment. The main sources of plastics into a river are atmospheric fallout of microplastics and microfibres (1), landfills, stormwater drains and wastewater treatment plants which tend to be most concentrated around urban centres (2), sewage sludge applied to agricultural croplands which can be washed into rivers (3), pieces of thin polyethylene sheets placed over agricultural soils to limit weed growth and which break down within a few months (3) and direct littering by recreational users or illegal dumping of plastic trash into rivers or onto their banks (4). Once plastics have entered the river they can be retained in riparian vegetation (5) or wash up on the riverbanks and become deposited in sediments (6), although this plastic can be washed back into the river during periods of high flow. Larger macroplastics can eventually break down into smaller fragments due to UV radiation and mechanical abrasion (7) and smaller fragments, microplastics and microfibres can be ingested by invertebrates, birds, mammals and fishes (8). Plastics not retained within the river catchment may eventually be washed out to sea (9).

A more robust method for assessing levels of microplastic and microfibre pollution is to filter bulk surface water samples (1–10 L) through filters with smaller mesh sizes (0.7–48 µm) (Barrows et al. 2017; Lahens et al. 2018; Ryan et al. 2019b). Microplastics and microfibres are often orders of magnitude more abundant in these bulk water samples compared to samples collected using a neuston net (Lahens et al. 2018). On the other hand, sampling for floating macroplastics requires large areas to be sampled, given their much lower densities than microplastics, and this generally requires either fixed nets left *in situ* for long periods (Gasperi et al. 2014) or direct observations of floating litter (Ryan 2013; Ryan 2014; Ryan et al. 2014b; González-Fernández & Hanke 2017; Van Emmerik et al. 2018).

To improve our understanding of plastic pollution in African rivers, I report levels of microplastic, microfibre and macroplastic pollution in the Orange-Vaal River system, the largest river system in South Africa. While most rivers are populated along their entire banks, the Orange River flows mostly through rural agricultural areas and its catchment is sparsely populated, particularly for the 500 km upstream of its mouth (although there is intensive irrigated agriculture along much of its banks). In contrast, the Vaal River, which is the largest tributary of the Orange River, drains some of the most densely populated and heavily urbanised areas of South Africa, including the southern part of the Johannesburg-Pretoria conurbation. This river system therefore offers an unusual opportunity to assess how plastics are transported by rivers as litter found near the mouth is likely to have originated inland in the Vaal River catchment. To assess to what extent the Orange-Vaal River transports plastics to the Atlantic Ocean, I quantify levels of plastic pollution along the lengths of both rivers and compare this to the 0.095 (0.037–0.355) tonnes of macroplastic predicted to be released from the Orange River into the Atlantic Ocean annually (Lebreton et al. 2017). I expect plastics to be most abundant in the highly urbanised and densely populated upper and middle reaches of the Vaal River. However, if plastics are evenly distributed downstream from this area, it would suggest that there is effective transport over long distances.

Methods

Field data

I collected samples from 33 bridges spanning the lengths of the Orange and Vaal rivers (Table S1), including two sites on the Klip River, one on the Blesbokspruit River and one on the Suikerbosrand River, all of which flow out of the heavily urbanised areas south of Johannesburg (Figure 3.2). The Blesbokspruit River flows into the Suikerbosrand River near the town of Heidelberg (26.6003°S 28.2900°E) which then, along with the Klip River, joins the Vaal River in Vereeniging (26.6708°S 27.9548°E) (Figure 3.2). I sampled each site twice: at the end of the wet season (April–May 2018) and at the end of the dry season (October–November 2018) (Figure S1). I chose bridges based on their accessibility and safety (e.g. I did not sample from major highways or from particularly narrow bridges) and all sampling was conducted during daylight hours. The bridges ranged from 1.5–20 m above the water. River width ranged from 11–333 m at the end of the wet season and 10–285 m at the end of dry season.

I grouped the sampling sites into six sections relative to major dams or natural features (Figure 3.2, Table S1). I classified sites upstream of the Vaal Dam as the Upper Vaal (n = 6), those between the Vaal and Bloemhof dams (which included sites on the Klip and Suikerbosrand rivers) as the Middle Vaal (n = 9), and bridges downstream from the Bloemhof Dam to the confluence with the Orange River as the Lower Vaal (n = 5). On the Orange River, I classified sites above the confluence of the Orange and Vaal rivers as the Upper Orange (n = 5), those between the confluence and Augrabies Falls as the Middle Orange (n = 5), and those between Augrabies and the river mouth as the Lower Orange (n = 3) (Figure 3.2, Table S1).

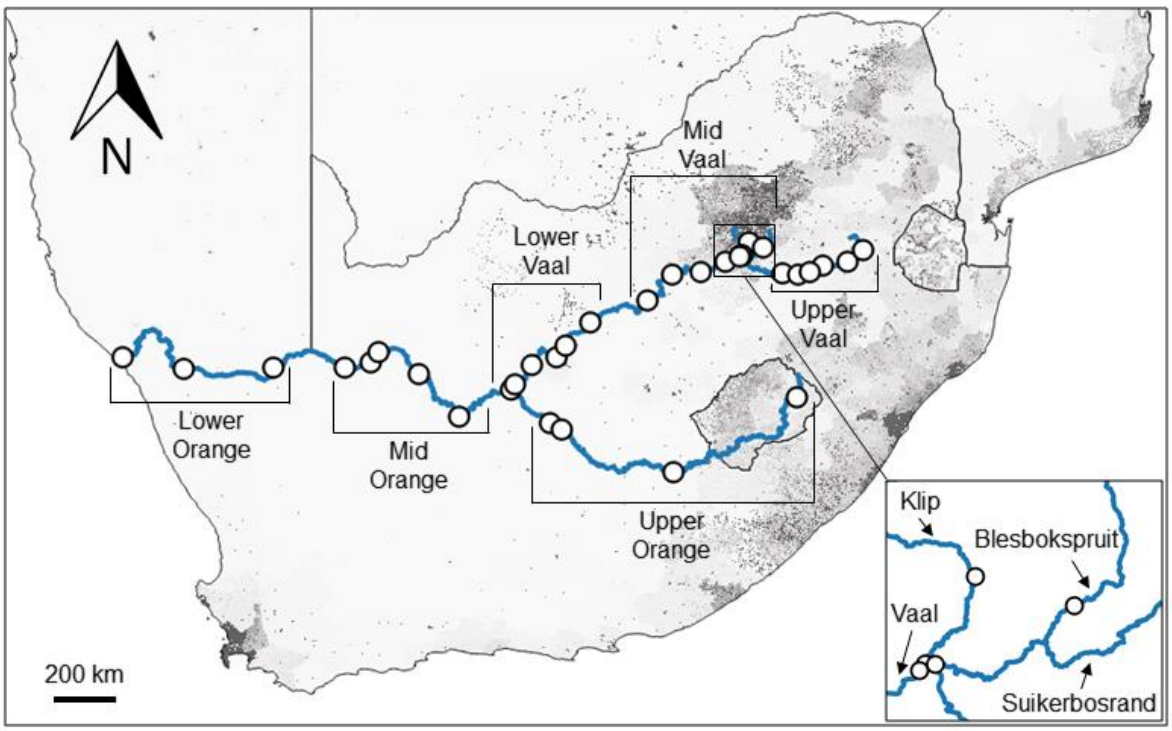


Figure 3.2: Sampling sites on the Orange-Vaal River system, South Africa. Sites were grouped into sections (lower, middle and upper) based on proximity to major features such as dams and the confluence of the Orange and Vaal rivers. The grey shading in the background indicates population density (Worldpop 2019), which peaks around sampling sites in the Upper and Middle Vaal River (darkest grey shading = 1.17 people per pixel). The insert shows sampling sites on the Klip, Blesbokspruit and Suikerbosrand rivers which drain the densely populated area south of Johannesburg.

The Middle Vaal River is the most heavily populated section of river, particularly the sampling sites on the Klip, Blesbokspruit and Suikerbosrand rivers, which are within 100 km of Johannesburg, South Africa's largest city (Figure 3.2, Table 3.1, Table S1). These sites all have more than 3 million people living within a 50 km radius and drain a large industrial area south of Johannesburg (Figure 3.2; Worldpop 2013). In contrast, sites on the lower and upper sections of the Vaal River are an order of magnitude less densely populated and have fewer WWTPs within 50 km upstream (Figure 3.1, Table 3.1, Table S1). The Orange River catchment is even less densely populated than that of the Vaal River, particularly the arid Lower Orange River where there are no WWTPs and fewer than 10 000 people living within 50 km of the sampling bridges (Figure 3.2, Table 3.1, Table S1).

Table 3.1: Number of sites, average number of wastewater treatment plants (WWTPs) 50km upstream of each site and average human population within 50 km of each site per river section of the Orange and Vaal rivers.

	# sites	**WWTP [a]**	**Human population [b]**
Orange River			
Upper Orange	5	2.3 ± 1.7	30 183 ± 20 265
Middle Orange	5	1.2 ± 0.4	51 513 ± 43 397
Lower Orange	3	0	4 623 ± 1 298
Vaal River			
Upper Vaal	6	3.0 ± 1.8	222 805 ± 53 663
Middle Vaal	9	5.4 ± 1.9	2 943 307 ± 2 636 506
Lower Vaal	5	2.2 ± 1.3	161 986 ± 129 029

[a] Van Deventer et al. 2018

[b] Worldpop 2013

To assess levels of microplastic and microfibre pollution, I collected replicate 10 L water samples (n = 9) from all 33 bridges. I collected three samples from the middle of each bridge and three from within 20 m of each bank when the river was wider than 100 m and from within 5 m of each bank when the river was narrower than 100 m. I used a galvanised steel bucket and a sisal (brown natural fibre) rope to fill a 10 L clear plastic container which I then gravity filtered through a 25 µm nylon mesh filter. I placed the filters in clean metal foil envelopes which I dried at 20 °C and stored until further processing at the University of Cape Town (UCT). To reduce the chance of atmospheric contamination, I triple rinsed all equipment and nylon filters with river water prior to use. Due to a faulty oven, 14 filters collected during the wet season were destroyed. I therefore based analyses on 5–9 replicates per site.

To sample for larger microplastics, I deployed a neuston net at bridges where the river was at least 50 cm deep (to avoid the net frame touching the bottom) and the water was flowing fast enough to flow through the net (minimum flow rate sampled = 0.002 $m \cdot s^{-1}$), but not faster than 3 $m \cdot s^{-1}$ so as not to damage the net. This limited sampling to 14 sites in the wet season (Upper Orange n = 2, Middle Orange n = 5, Lower Orange n = 2, Upper Vaal n =3, Middle Vaal n = 4, Lower Vaal n = 2) and 10 sites in the dry season (Middle Orange n = 3, Lower Orange n = 1, Upper Vaal n

= 1, Middle Vaal n = 5). The net has a mesh size of 300 µm, is 60 cm wide and samples the top 30 cm of the water column. I deployed the net in replicate (n = 2–4 per site) for 7–15 minutes, depending on how long it took to fill with debris. Ideally, I would have deployed the net once at each side of the river (within 20 m of the bank) and twice in the middle of the river, but this was seldom possible due to the river being too shallow. At two sites (Middle Orange n = 1, Lower Orange n = 1), the bridge was too high to safely deploy the net, and so I took a sample within 1 km downstream of the bridge. Here, I waded into the middle channel of the river and deployed the net for two 15-minute periods using a 3–4 m long rope. I used a flowmeter (General Oceanics Model 2030RC) attached to the inside of the net frame to estimate the volume of water flowing through the net. After each deployment, I rinsed the contents of the net into the cod-end using water filtered through a 25 µm filter. I then emptied the cod-end onto a 330 µm metal mesh sieve from which I rinsed the sample into a small plastic sampling jar or ziplock bag. I visually sorted the samples and picked out all anthropogenic litter items using forceps which I stored in 70% ethanol until further processing in the lab.

To estimate densities of macrodebris, I conducted observations at bridges where there was visible river flow (n = 28 bridges in the wet season and n = 20 bridges in the dry season) following a similar protocol used to estimate macrolitter densities at sea (Ryan 2013; Ryan 2014; Ryan et al. 2014b). Observers stood in the middle of the bridge facing upstream and recorded all litter items seen for two 15-minute periods. Where the river was >100 m wide, they only viewed 50 m on either side. Most litter was detected with the naked eye, but regular scans of the river were made with 10 x 32 binoculars to detect smaller and/or more distant debris. Anthropogenic material was identified using binoculars, although some submerged items could not be identified. Observers recorded the colour of the item, the type of material (plastic, wood, etc.), its buoyancy (positive [i.e. above the surface], negative [i.e. below the surface] or neutral [i.e. on the surface]), the approximate size of the item and the perpendicular distance from the observer. To estimate the ratio of natural to anthropogenic material floating in the river, natural items (e.g. leaves, feathers, etc.) also were recorded.

To further estimate the amount of litter delivered by the Orange River into the Atlantic Ocean, an additional sampling trip was conducted to the bridge at Alexander Bay, near the mouth of the Orange River, in August 2018 (middle of the dry season, Figure S1). This followed the same sampling protocol: I collected replicate (n = 9) bulk water samples, deployed the neuston net for four 15-minute tows and conducted observations for macrolitter. However, to increase the chance of recording macroplastic items in the river, two observers simultaneously scanned one half of the river for ten 15-minute periods (i.e. the river was observed for 150 minutes by two independent observers).

Sample processing

To ensure consistency, I processed all bulk water samples. Some filters (n = 45) were saturated with sediment, making it difficult to see microplastics and microfibres. I therefore used density separation to remove the sediment. First, I used Milli-Q water to rinse the contents of the filter into a clean glass beaker to which I added 100 ml of saturated sodium chloride solution (1.2 $g \cdot ml^{-1}$) Thompson et al. 2004). To reduce the chance of contamination, I pre-filtered the sodium chloride solution through a 25 µm nylon mesh filter (Nel et al. 2018). Using a clean metal spoon, I then vigorously stirred the contents of the beaker for 2 minutes after which I left it to settle for 10 minutes. I then filtered the supernatant through a 25 µm filter which I finally examined under a stereomicroscope. To decrease the chance of contamination, I triple-rinsed all equipment with Milli-Q water prior to use and covered all equipment with metal foil when not in use.

I observed all filters under a stereomicroscope (8X–50X magnification) and counted all fibres and plastic fragments. I measured their width and length using the eyepiece graticule and noted their colour. Microfibres were identified by their uniform width, bright artificial colours, lack of cellular structures and smooth edges (Barrows et al. 2018b); plastic pieces by their bright artificial colours and sharp edges (Hidalgo-Ruz et al. 2012). It should be noted that recent research has shown that many microfibres found in the natural environment are of natural origin (wool, cotton or cellulose) rather than plastic (Stanton et al. 2019; Suaria et al. 2020). While I

attempted to identify the fibres found in my bulk water samples using a Raman spectroscope at the UCT Electron Microscope Unit, the spectroscope was fitted with a green laser (532 nm wavelength), causing the samples to fluoresce to such an extent that it was not possible to acquire a clear spectrum. To fix this, a red laser is required (785nm wavelength) but I did not have access to one. Effort will be made to identify the fibres at a later stage.

I used a triple-rinsed 25 µm filter as a procedural blank to assess atmospheric contamination in the lab. I placed the blank next to the microscope and exposed it to the air while processing each sample. I only found microfibres on the procedural blanks; these I counted and subtracted from the total count of microfibres in the corresponding sample.

I air-dried items found in the neuston net, weighed them to the nearest 0.1 mg and classified them according to their colour, material (e.g. card, plastic, etc.) and type (balls/beads, fibres, films, pieces of flexible packaging, hard pieces, pellets, polystyrene pieces, pieces string/twine).

Total amount of plastic at each site

To assess overall pollution, I calculated the total abundance and mass of plastic found at each bridge by combining measures of plastic found in bulk water, neuston net and macroplastic samples expressed in terms of number (items·m^{-2}) and mass (mg·m^{-2}). To estimate the masses of microfibres, I calculated each fibres' volume by assuming a cylindrical shape and using its length and width measured with the graticule of the stereomicroscope. To estimate mass, I then multiplied the volume of each fibre by a density of 1.35 g·cm^{-3} (the average density of the most common fibre polymers, namely wool, cotton and polyester; Stanton et al. 2019). For microplastic pieces, I multiplied the volume (length × breadth × height) by a density of 1 g·cm^{-3}. To ensure that the density of microplastics in bulk water samples was directly comparable to neuston net and macroplastic samples, I extrapolated the total number and mass of microfibres per litre to total number and mass per m^2 by assuming that the bucket sampled the top 30 cm of the water column, the same

depth sampled by the neuston net (i.e. the abundance and mass of microplastics and microfibres per litre was multiplied by 300 to give the density and mass per m^2).

For neuston net samples, I calculated the linear distance trawled during each deployment by converting the number of rotations of the flow meter to metres using the formula:

$$Distance\ trawled\ (m) = (flowmeter\ revolutions\ x\ 26873)/999999$$

following the manufacturers specifications (General Oceanics Inc. 2019). To make measures of net plastics directly comparable to macroplastic observations, I then multiplied the distance trawled by the net width (0.6 m) to give the surface area sampled.

For macroplastic observational data, I assumed that all items within 50 m of either side of the observer and within the top 30 cm of the water column were seen. To verify the second assumption, I lowered a weighted 21.5 cm wide Secchi disk from the middle of the bridge (the same point from which the observer conducted their observation) using a rope and measured the depth at which the disk was no longer visible from directly above. At the end of the wet season, the Secchi disc was visible to an average depth of 0.51 ± 0.23 m and to 0.58 ± 0.28 m at the end of the dry season. The mass of each item observed was estimated from weights of similar items sampled during surveys of street, river and beach litter collected elsewhere in South Africa. Each item was given a plausible upper and lower mass estimate to account for possible error in this process, with the average of these two values used as the best estimate. I calculated the total area observed for macrolitter by multiplying the flow rate of the river (based on the flow meter deployed with the neuston net or from DWS [2019a]) by the duration of observation (seconds) and by the transect width (the width of the river for sites where the river was < 100 m wide, or by 100 m where the river was >100 m).

Data analyses

The number of items·L^{-1} in bulk water samples, items·m^{-2} in net tows and items·m^{-2} seen during observations were not normally distributed (Shapiro-Wilk normality tests; all p-values < 0.001). I therefore used non-parametric Mann-Whitney tests to test for differences in microplastic and microfibre concentrations in bulk water samples collected from the sides versus the middle of the river. I used non-parametric Kruskal-Wallis tests to test for differences in microplastic and microfibre concentrations in samples collected at the river mouth at the end of the wet season (April 2018), end of the dry season (November 2018) and in the middle of the dry season (August 2018). Where significant, I used a *post-hoc* Dunn's test to test for differences between cases and applied a Bonferroni correction to decrease the chance of committing Type 1 errors. I used linear correlations to test if there was a relationship between the amount of plastic caught in the neuston net and that caught in bulk water samples and seen in macroplastic observations.

To test for differences in microfibre and microplastic concentrations between the different river sections and across seasons, I fit a Generalised Linear Mixed Model (GLMM) using the *lme4* package in R (Bates et al. 2015; R Core Team 2019). I used a negative binomial distribution to account for the data being over-dispersed and modelled the number of items·10 L^{-1} (c.f. per litre) to ensure that the response variable was an integer. I included site as a random effect and ran the model using the "bobyqa" optimiser. I used the same GLMM for the neuston net samples except I modelled the number of items caught per 15-minute tow. For the macrodebris observations, I used a hurdle model as the data were zero-inflated and right-skewed. This model has two steps: first I fit a Generalised Linear Model (GLM) with a binomial distribution which models whether litter was seen in a given observation (i.e. zero values vs. non-zero values). I then modelled replicates where litter was seen (i.e. non-zero values) using a GLM fit with a quasi-Poisson distribution to account for the data being over-dispersed. I used river section and season as explanatory variables. I also modelled the total mass of plastics (bulk water, neuston net and observation data combined) as a function of river section and season using a GLMM fit with a Gamma distribution to account for the data being semi-continuous. I included site as a random term.

To assess how much variation there was in microplastic and microfibre concentrations at each site, I calculated coefficients of variation (CV = mean/SD) for each site which I then averaged across all sites. I did not do this for neuston net or macroplastic samples due to the large number of samples where no plastic was found (i.e. the CV would be zero).

Only microfibres were found in samples collected at the Orange River mouth. I therefore estimated the mass of microfibres released into the Atlantic Ocean each year by multiplying the average mass of fibres·m^{-3} found at the end of the wet season (3.10 mg·m^{-3}) by the average river flow (m^3·day^{-1}) measured over the past five years (2015–2019) between January–June, the average mass of fibres found in August (0.52 mg·m^{-3}) by the average river flow between July–September and the average mass of fibres found at the end of the dry season (0.35 mg·m^{-3}) by the river flow between October–December. I used river flow data measured at Sendelingsdrift (28.125°S, 16.890°E), approximately 100 km upstream from the mouth of the river (DWS 2019b). No data were available for Sendelingsdrift between January and mid-May 2018. For this period, I estimated the flow rate as 40% less than the flow rate measured at Upington (28.471°S 21.234°E), approximately 570 km upstream of Sendelingsdrift. This was based on the flow rate being on average 40.0 ± 15.7% lower at Sendelingsdrift than Upington for the other four years (2015–2017 and 2019). I calculated an upper estimate as above except I used the upper 95% confidence interval (mean mass of fibres + 2 x standard error) of the mass of fibres found at the end of the wet season (3.98 fibres·m^{-3}), August (0.65 fibres·m^{-3}) and the end of the dry season (0.42 fibres·m^{-3}). To calculate a lower estimate, I multiplied the 95% lower confidence interval of the mass of fibres (mean mass – 2 x standard error) measured at the end of the wet season (2.22 fibres·m^{-3}) by the average flow rate between April–June, and the average lower 95% confidence interval in August and at the end of the dry season (0.0003 fibres·m^{-3}) by the average flow rate between January–March and July–December.

I conducted all analyses in R version 3.6.0 (R Core Team 2019) and report results as mean ± 1 SD unless otherwise indicated.

Results

Bulk water samples

Overall, 10 075 potential microplastics were found, of which 99.4% were microfibres (n = 10 016) and only 0.6% (n = 59) were fragments. Fibre contamination in the lab was low; procedural blanks accumulated an average of 0.08 ± 0.30 microfibres·filter^{-1} (median = 0 microfibres·filter^{-1}) and all fibres found on the blanks were blue. After controlling for contamination, the average number of microfibres and microplastics per site was 1.71 ± 5.10 items·L^{-1} and the median 0.5 items·L^{-1}. Microfibres were detected at all sites and in 90.3% of replicates at a mean abundance of 2.3 ± 7.2 microfibres·L^{-1} in the wet season and 1.4 ± 2.6 microfibres·L^{-1} in the dry season. Fragments were only found at seven sites at the end of the wet season (0.2 ± 0.2 fragments·L^{-1}) and five sites at the end of the dry season (0.2 ± 0.2 fragments·L^{-1}) (Table 3.2). Microfibres weighed an estimated average of 0.0002 ± 0.0008 mg and were mostly blue (n = 9737, 97.2%) while fragments were on average 0.0009 ± 0.0017 mg and were green (n = 28, 43%), white (n = 16, 25%), blue (n = 11, 17%), yellow (n = 6, 9%), orange (n = 2, 3%) and pink/red (n = 2, 3%). No significant difference in microfibre and microplastic concentrations was found between samples collected from the sides of the river compared to the middle (Mann-Whitney test: W = 40278, p = 0.12Coefficients of variation within sites were high (112 ± 41%, range = 67–209%) indicating that bulk water samples provide only a rough estimate of the amount of microfibre and microplastic pollution at a site.

Microfibres and microplastics were slightly more abundant in the wet (2.08 ± 6.86 items·L^{-1}) than dry season (1.34 ± 2.53 items·L^{-1}), although the GLMM predicted the opposite (Table 3.3). In the wet season, microfibres were most abundant at sites in the Lower Orange River (14.6 ± 17.1 items·L^{-1}), where they were 6–49 times more abundant than in the other sections of the rivers (Figure 3.3A) but decreased to 1.33 ± 1.35 items·L^{-1} in the dry season, resulting in microfibres and microplastics being much more homogenous throughout both rivers in the dry season compared to the wet season (Figure 3.3A). Overall, the GLMM showed significantly higher concentrations of microfibres and microplastics in the Middle and Lower Orange River than the Upper Orange River (Table 3.3).

Only microfibres were found at Alexander Bay, the mouth of the Orange River. They were more abundant in April 2018 (end of the wet season, 19.48 ± 13.49 items·L^{-1}) than in August 2018 (middle of the dry season, 1.92 ± 1.12 items·L^{-1}; Dunn's test: $Z = 2.82$, $p < 0.05$) or in November 2018 (end of the dry season, 0.90 ± 0.49 items·L^{-1}; Dunn's test: $Z = 4.40$, $p < 0.001$), with no significant difference between August and November (Dunn's test: $Z = 1.58$, $p = 0.35$). Based on the average concentrations of microfibres found at Alexander Bay, and assuming fibres are evenly distributed throughout the water column, it is estimated that 2.5 (1.2–3.5) tonnes of microfibres wash out of the Orange River annually.

Neuston net trawls

Due to low water levels, net sampling was only possible at 14 sites at the end of the wet season and 10 sites at the end of the dry season. Three macrolitter objects were found in a single net sample collected at the end of the wet season at the lower site on the Klip River (a card cigarette box, a piece of plastic cigarette wrapper and a plastic lollipop stick). Although they comprised <0.1% of net litter items, they weighed 3.66 g and accounted for 88.1% of the mass of litter found in that sample and 49.7% of the litter in all net samples. Excluding these items, 3 324 litter items (0.34 ± 0.93 items·m^{-2}) weighing 3.704 g (0.31 ± 0.90 mg·m^{-2}) were found in the neuston trawls of which >99% were plastic (89% by mass, Table 3.4). Although pieces of paper were the most abundant non-plastic litter items, paraffin wax made up most of the mass of the non-plastic items (Table 3.4). The plastic items were mostly pieces of flexible packaging and hard fragments (Table 3.4) and were mostly clear (n = 1 537, 46.4%), white (n = 656, 19.8%) and blue (n = 478, 14.4%). On average, items weighed 1.14 ± 8.28 mg, with 82.4% (n = 2 739) weighing less than 1 mg and 96.9% (n = 3 222) less than 5 mg.

Table 3.2: Densities of microfibres and microplastic pieces found in bulk water samples collected in the lower, middle and upper reaches of the Orange and Vaal rivers in South Africa at the end of the wet and dry seasons.

	Mean $N \cdot L^{-1}$		Mean $mg \cdot m^{-3}$	
	Wet	**Dry**	**Wet**	**Dry**
Fibres	**2.3 ± 7.2**	**1.4 ± 2.6**	**0.6 ± 1.6**	**0.3 ± 0.5**
Upper Orange	0.6 ± 0.4	1.0 ± 1.2	0.2 ± 0.1	0.3 ± 0.3
Middle Orange	2.3 ± 4.4	1.7 ± 2.0	0.5 ± 1.1	0.4 ± 0.5
Lower Orange	17.1 ± 17.4	1.3 ± 1.3	2.9 ± 4.1	0.4 ± 0.5
Upper Vaal	0.4 ± 0.3	2.3 ± 3.2	0.3 ± 0.6	0.5 ± 0.7
Middle Vaal	0.5 ± 0.4	1.3 ± 3.7	0.4 ± 1.0	0.2 ± 0.6
Lower Vaal	0.7 ± 0.7	0.8 ± 0.8	0.3 ± 0.4	0.2 ± 0.2
Pieces	**0.2 ± 0.2**	**0.2 ± 0.2**	**0.2 ± 0.3**	**0.1 ± 0.1**

Table 3.3: GLMM results showing the effect (± standard error) of season and river section on the mean abundance of microplastics (items·10 L^{-1}) found in bulk water samples collected from 33 bridges spanning the lengths of the Orange and Vaal rivers. Estimates are presented on the log-scale.

	Estimate (± SE)	**z-value**	**P**
Season (dry) [a]	0.275 ± 0.110	2.495	**0.013**
River section [b]			
Middle Orange	0.990 ± 0.418	2.367	**0.018**
Lower Orange	2.102 ± 0.503	4.183	**<0.001**
Upper Vaal	0.322 ± 0.402	0.801	0.423
Middle Vaal	-0.006 ± 0.372	-0.017	0.987
Lower Vaal	0.039 ± 0.419	0.093	0.926
Intercept	1.699 ± 0.302	5.618	**<0.001**

[a] Estimate indicates expected microfibre and microplastic count in the dry season relative to the wet season

[b] Estimates indicate expected microfibre and microplastic count in the given river section relative to the reference section (Upper Orange)

Net plastics and other litter items were more evenly distributed than macroplastics, being found at 15 of 18 sites (83%) sampled at the end of the wet season and 9 of 10 sites (90%) at the end of the dry season (Figure 3.3). The GLMM showed significantly higher concentrations of litter in the wet season (0.38 ± 1.06 items·m^{-2}) compared to the dry season (0.27 ± 0.69 items·m^{-2}) (Table 3.5). In the wet season, plastics were most abundant in net tows conducted in the middle (0.90 ± 1.64 items·m^{-2}) and lower (0.71 ± 0.28 items·m^{-2}) sections of the Vaal River, and in the Middle Vaal River at the end of the dry season (0.54 ± 0.93 items·m^{-2}) (Table 3.5). The abundance of net plastics from the Middle Vaal River was due to high concentrations along the Klip River (Figure 3.3C). Pollution at both sites on the Klip River was 1–3 orders of magnitude higher than at other sites sampled. Overall, the GLMM showed significantly higher concentrations of neuston net litter in the Upper and Middle Vaal River than the Upper Orange River (Table 3.5).

Table 3.4: The abundance and total mass (g) of items caught in the neuston net deployed from bridges on the Orange and Vaal rivers at the end of the wet and dry seasons in 2018.

Type of litter	n	% n	Mass (g)	% mass
Plastic items	**3 315**	**99.7**	**3.277**	**88.5**
Flexible packaging	2 629	79.1	2.227	60.1
Hard pieces	601	18.1	0.722	19.5
Balls/beads	50	1.5	0.056	1.5
Fibres	16	0.5	0.001	0.03
Polystyrene	14	0.4	0.243	6.5
String/twine	3	0.09	0.016	0.4
Industrial pellet	1	0.03	0.002	0.1
Softdrink label	1	0.03	0.011	0.3
Non-plastic items	**9**	**0.3**	**0.428**	**11.5**
Pieces of paper	5	0.2	0.023	0.6
Paraffin wax	3	0.1	0.401	10.8
Rubber bristle	1	0.03	0.004	0.1
TOTAL	**3 324**		**3.704**	

Table 3.5: GLMM results showing the effect (± standard error) of season and river section on the mean abundance of litter caught in the neuston at 33 bridges spanning the lengths of the Orange and Vaal rivers. Estimates are presented on the log-scale.

	Estimate (± SE)	**z-value**	**P**
Season (dry) [a]	-0.635 ± 0.298	-2.131	**0.033**
River section [b]			
Middle Orange	-0.565 ± 1.175	-0.481	0.630
Lower Orange	-1.811 ± 1.517	-1.194	0.233
Upper Vaal	-2.878 ± 1.439	-2.001	**0.045**
Middle Vaal	2.040 ± 1.179	1.731	0.083
Lower Vaal	-0.890 ± 1.407	-0.633	0.527
Intercept	1.212 ± 0.994	1.220	0.223

[a] Estimate indicates expected microfibre and microplastic count in the dry season relative to the wet season

[b] Estimates indicate expected microfibre and microplastic count in the given river section relative to the reference section (Upper Orange)

Due to low water levels at the mouth of the Orange River, no neuston tows were conducted at Alexander Bay at either the end of the wet or dry season. However, in August, water levels were high enough to deploy the net and a single red piece of hard plastic weighing 0.0049 g was found in one of the four net tows (0.05 items·m^{-2} and 0.0003 mg·m^{-2}). The site closest to the mouth at which the net could be deployed at the end of the wet season was Vioolsdrift (~260 km from the mouth), and at Onseepkans at the end of the dry season (~460 km from the mouth). No plastic was found at either of these sites during these deployments, although eight pieces of plastic (6 hard pieces, 1 flexible packaging and 1 polystyrene; 0.18 ± 0.19 items·m^{-2} and 0.39 ± 0.55 mg·m^{-2}) were found at Onseepkans at the end of the wet season.

Macrodebris observations

Only 76 macrolitter items (mean = 0.0002 ± 0.0007 items·m^{-2}) were seen during observations, of which 97% (n = 74) were plastic (Table 3.6). Most items were neutrally buoyant (n = 66, 87%) while seven (9%) were positively buoyant and three

(4%) were negatively buoyant. On average, items weighed an estimated 7.6 ± 15.7 g (lower estimate = 5.7 ± 12.4 g, upper estimate = 9.6 ± 19.2 g) giving an estimate of 0.8 ± 2.9 $mg \cdot m^{-2}$ (range 0.5 ± 2.2 to 1.1 ± 4.4 $mg \cdot m^{-2}$). Most items were miscellaneous plastic pieces, bags/packets, food packaging or pieces of polystyrene (Table 3.6).

Macrolitter items were only seen at 35% of sites (n = 20) at the end of the dry season and at 14% of sites (n = 28) at the end of the wet season (Figure 3.3B). Even at sites where macrolitter items were seen, organic litter was 10–100 times more abundant than macrolitter. Overall, the hurdle model predicted higher concentrations of macrolitter at the end of the dry season than the end of the wet season (Table 3.7).

Table 3.6: The abundance and estimated lower and upper mass of macrodebris items seen during observations conducted from bridges on the Orange and Vaal rivers at the end of the wet and dry seasons in 2018.

Type and colour of litter	**n**	**% n**	**Mass (g) [a]**	**% mass**
Plastic items	**74**	**97**	**576 (430–722)**	**99**
Miscellaneous pieces	18	24	141 (89–194)	24
Bags or packets	16	21	51 (39–63)	9
Food packaging	15	20	40 (28–52)	7
Polystyrene pieces	10	13	85 (67–103)	15
Plastic bottles	5	7	199 (167–230)	34
Hard plastic pieces	4	5	26 (16–35)	4
Plastic bottle tops	4	5	7 (5–10)	1
Plastic earbud sticks	1	1	0.2 (0.2–0.3)	0.04
Plastic medicine tub	1	1	28 (20–35)	5
Non-plastic items	**2**	**3**	**3.4 (2.2–4.6)**	**0.6**
Card cigarette box	1	1	1.5 (0.5–2.5)	0.3
Metal bottle top	1	1	1.9 (1.7–2.1)	0.3
TOTAL	**76**		**579 (432–727)**	

[a] Best estimate (lower–upper estimates)

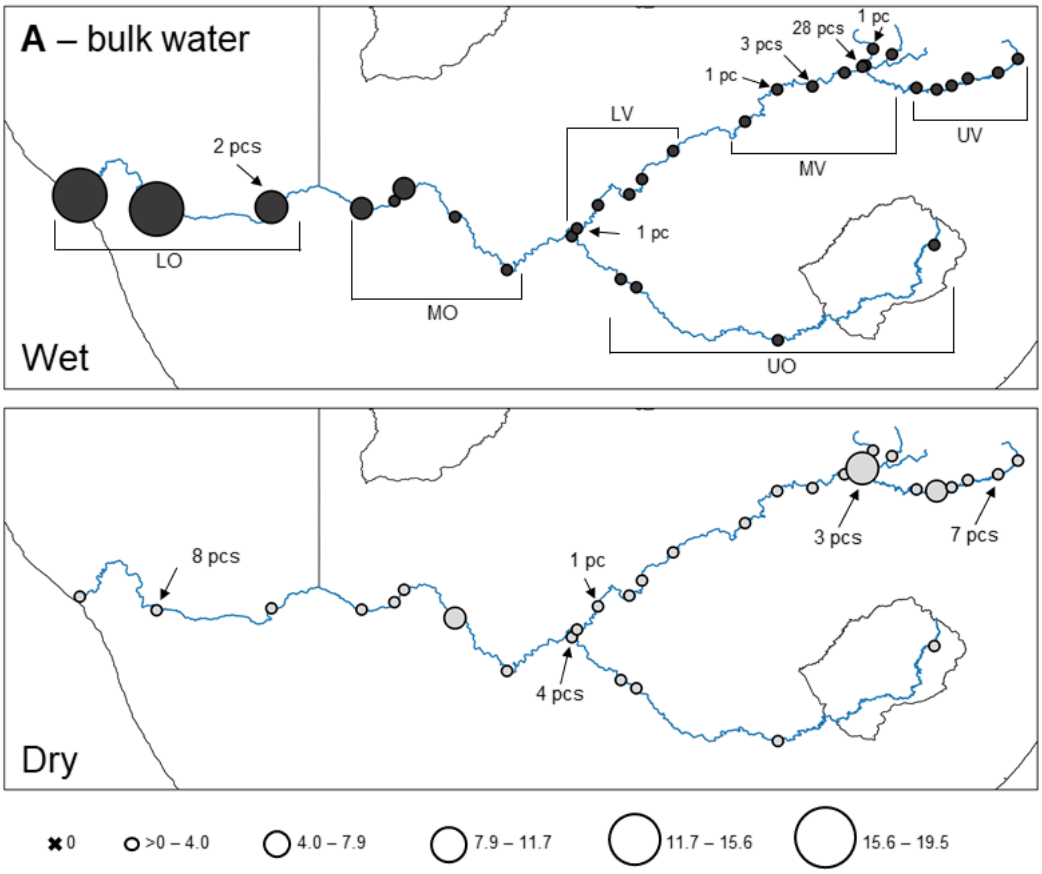
Macrolitter items were most abundant at the lower site on the Klip River, closest to the confluence with the Vaal River (site "S1 Klip" in Figure 3.5), where 59% of all macrolitter items were seen (Figure 3.3C). This resulted in macrolitter being most abundant in the Middle Vaal River in both seasons (mean wet season = 0.0006 ± 0.0015 items·m^{-2}, mean dry season = 0.0008 ± 0.0011 items·m^{-2}) and significantly more macrolitter in the Middle Vaal than the Lower Vaal River (Table 3.7). No macrolitter were observed in the Lower Orange River in either season, but a single plastic sweet wrapper was spotted at the mouth of the Orange River in August 2018, when sampling effort was increased. The most downstream site at which macrolitter items were seen at the end of the wet season was on the Vaal River near Douglas, 15 km upstream from the confluence of the Orange and Vaal rivers (~1 100 km from the mouth of the Orange River). At the end of the dry season, the site at which macrolitter were seen closest to the mouth of the river was near Upington, approximately 700 km upstream from the mouth (Figure 3.3C).

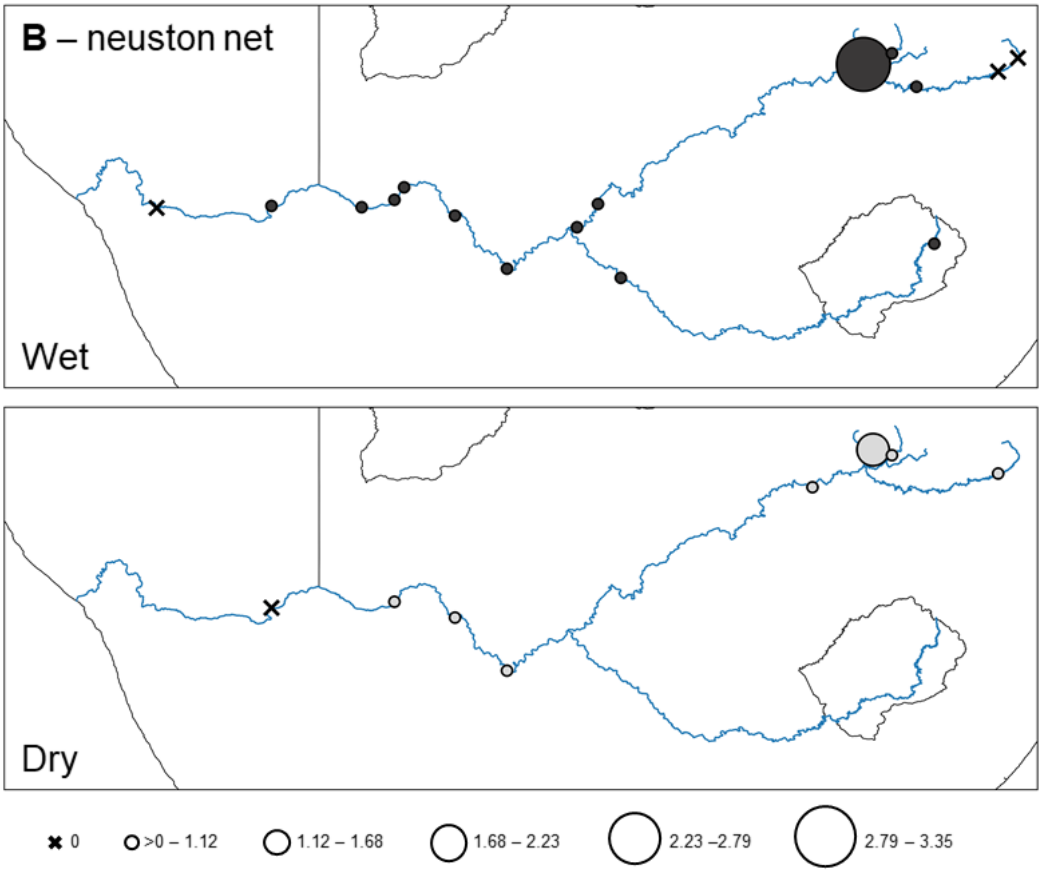
Table 3.7: Hurdle model results showing the effect (± standard error) of season and river section on the mean abundance of macrolitter seen during 15-minute observations conducted from 33 bridges spanning the lengths of the Orange and Vaal rivers. Estimates are presented on the log-scale.

	Estimate (± SE)	**t-value**	**P**
Season (dry) [a]	-1.620 ± 0.262	-6.173	**<0.001**
River section [b]			
Upper Orange	1.697 ± 1.150	1.476	0.168
Middle Orange	2.313 ± 1.228	1.884	0.086
Upper Vaal	0 ± 1.562	0	1
Middle Vaal	2.806 ± 1.121	2.504	**0.029**
Intercept	0 ± 1.105	0	1

[a] Estimate indicates expected macrolitter count in the dry season relative to the wet season

[b] Estimates indicate expected macrolitter count in the given river section relative to the reference section (Lower Vaal). No macrolitter items were seen in the Lower Orange River and it was therefore not included in the model


A – bulk water
2 pcs
LO
MO
LV
MV
UV
UO
1 pc
3 pcs
28 pcs
1 pc
1 pc
Wet
8 pcs
1 pc
4 pcs
3 pcs
7 pcs
Dry
0
>0 – 4.0
4.0 – 7.9
7.9 – 11.7
11.7 – 15.6
15.6 – 19.5


B – neuston net
Wet
Dry
0
>0 – 1.12
1.12 – 1.68
1.68 – 2.23
2.23 –2.79
2.79 – 3.35

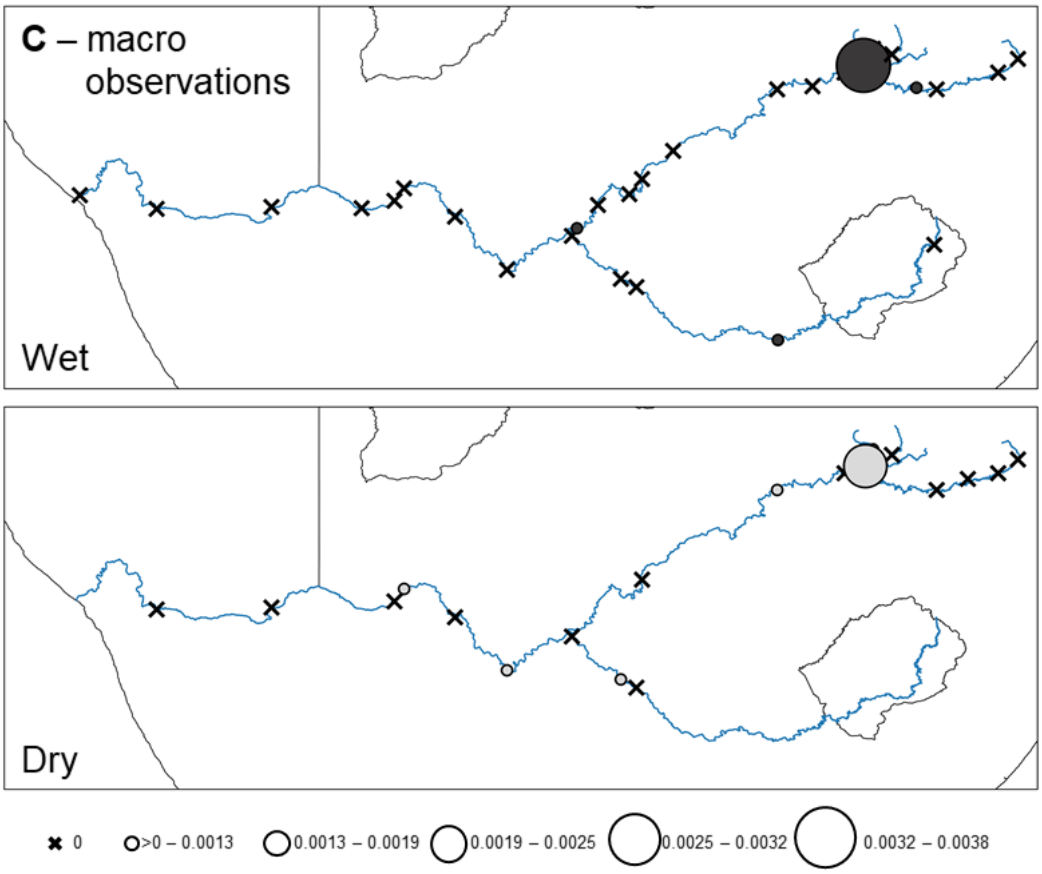


Figure 3.3: Average concentrations of plastics and other litter items found in (A) bulk water samples (items·L^{-1}), (B) neuston net samples (items·m^{-2}) and (C) macrolitter observations (items·m^{-2}) collected from bridges on the Orange and Vaal rivers in the wet (dark grey circles) and dry (light grey circles) seasons. The circles are proportional to the average number of items found at that site (note the different scales for A, B and C) and an "x" indicates sites where no plastic was found. The different river sections have been indicated in panel A. The arrows in panel A indicate sites where microplastic pieces (c.f. microfibres) were found. For bulk water samples, microfibres found in samples collected from Alexander Bay (the mouth of the Orange River) during August have been added to the dry season map.

Total amount of litter at each site

There was no correlation between the number of microfibres and microplastics and the number of items found in the net ($r = -0.12$, $p = 0.56$) or the number of macrolitter ($r = -0.10$, $p = 0.48$) items sampled at the same site/season (Figure 3.4). When extrapolated to the number of items per m^2, microplastics and microfibres in bulk water samples (528 ± 1 075 items·m^{-2}) were three orders of magnitude more abundant than net litter (0.34 ± 0.93 items·m^{-2}) and six orders greater than macrolitter (0.0002 ± 0.0007 items·m^{-2}). However, the mass of microfibres and microplastics (0.12 ± 0.19 mg·m^{-2}) was if anything less than that of net litter (0.31 ± 0.90 mg·m^{-2} excluding the three macrodebris items caught in the net or 0.66 ± 3.82 mg·m^{-2} including them) and macrolitter (0.78 ± 3.23 mg·m^{-2}). The average mass of macrolitter was skewed by the large number of sites where no macrolitter were observed. At sites where macrolitter items were recorded, their average density was 4.35 ± 6.71 mg.m^{-2}, an order of magnitude greater than the mass of microfibres and microplastics (Figure 3.4, Figure 3.5). At these sites, macrolitter made up 87% (± 24%) of the mass of litter at the end of the dry season and 80% (± 36%) at the end of the wet season (Figure 3.5). Again, there was no correlation between the mass of litter in bulk water and net samples ($r = -0.14$, $p = 0.48$) or between the mass of litter in bulk water samples and macrolitter ($r = -0.13$, $p = 0.38$) (Figure 3.4). There was a significant correlation between the mass of macrodebris and the mass of net litter in the wet season ($r = 0.98$, $p < 0.001$), but not in the dry season ($r = 0.11$, $r = 0.80$) (Figure 3.4C). The strong correlation between macrolitter and net samples at the end of the wet season was driven by the high abundance of litter at the lower site on the Klip River (labelled site in Figure 3.4C). The GLMM showed no difference in the mass of litter in the wet and dry seasons and a significantly higher mass of litter in the Lower Orange River than the Upper Orange River (Table 3.8).

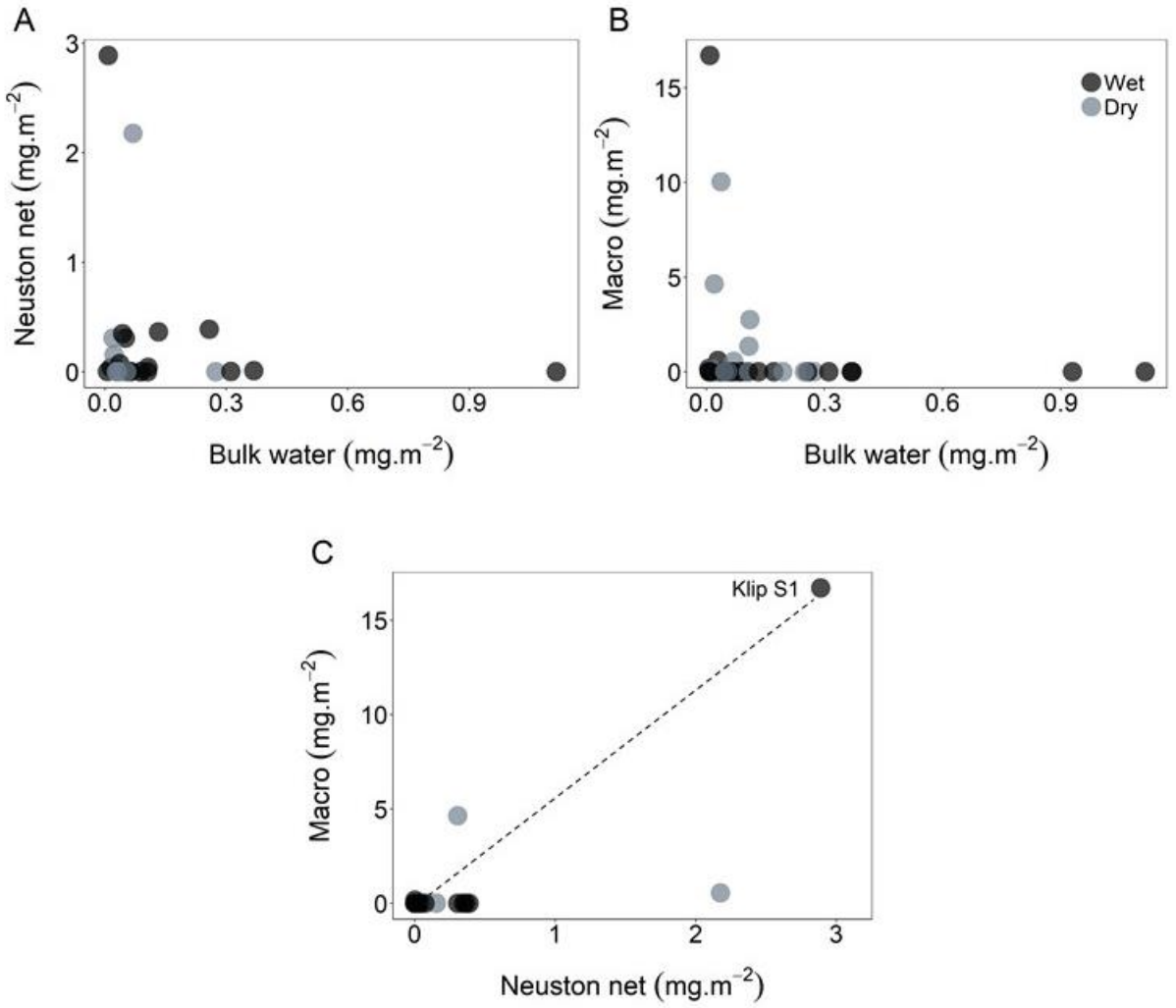


Figure 3.4: Plots of the mass of plastics (mg·m^{-2}) in (A) bulk water versus neuston net samples, (B) bulk water versus macrolitter samples and (C) macrolitter versus neuston net samples. The only significant correlation was between macrolitter and neuston net samples in the wet season which was driven by the high mass of litter in both types of samples at site S1 on the Klip River (labelled Klip S1).

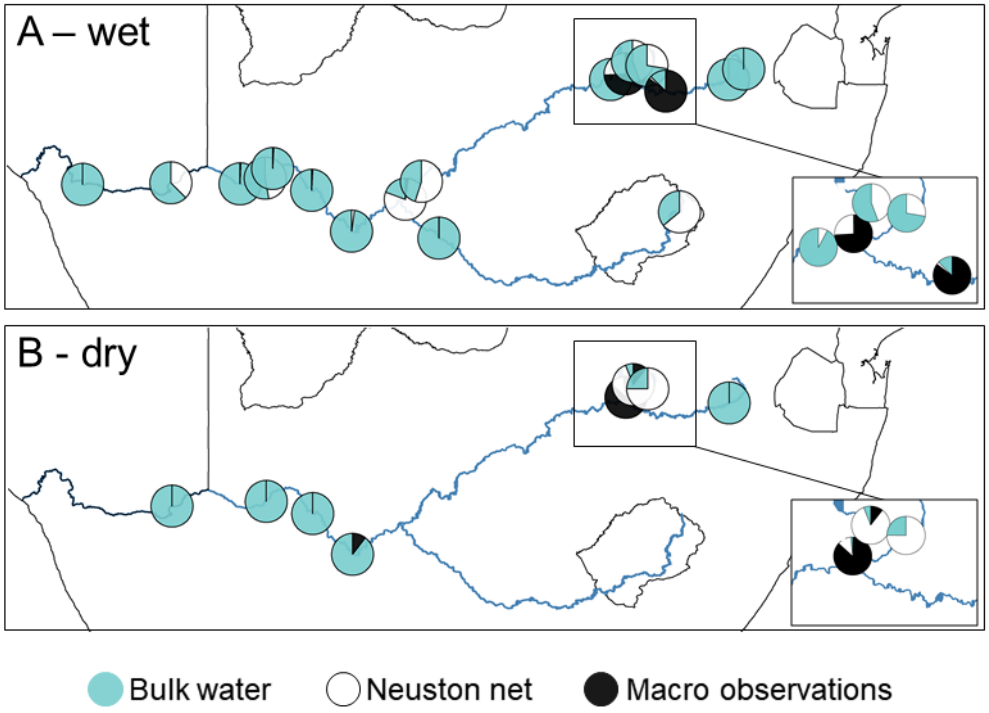


Figure 3.5: Sites where bulk water, neuston net and observation samples were collected in the wet (A) and dry (B) seasons on the Orange-Vaal River system, South Africa. Pie charts show the proportion of the total mass of plastic (excluding the three macrodebris items caught in the net) made up by each type of sample.

Table 3.8: GLMM results showing the effect (± standard error) of season and river section on the mean mass of litter ($g \cdot m^{-2}$) at 33 sites spanning the lengths of the Orange and Vaal rivers. Estimates are presented on the log-scale.

	Estimate (± SE)	**t-value**	**P**
Season (dry) [a]	0.017 ± 0.157	0.109	0.913
River section [b]			
Middle Orange	0.383 ± 0.607	0.630	0.529
Lower Orange	1.442 ± 0.704	2.049	**0.040**
Upper Vaal	-0.064 ± 0.583	-0.110	0.912
Middle Vaal	0.870 ± 0.537	1.621	0.105
Lower Vaal	0.0300 ± 0.609	0.049	0.961
Intercept	-2.676 ± 0.436	-6.138	**<0.001**

[a] Estimate indicates expected mass of litter in the dry season relative to the wet season

[b] Estimates indicate expected mass of litter in the given river section relative to the reference section (Upper Orange)

Discussion

Africa is home to some of the largest and most turbid rivers in the world. Large parts of some of these river catchments are densely populated and waste within these areas is often poorly managed (Jambeck et al. 2018). These rivers may therefore act as significant sources of waste plastics and other litter into coastal seas, with three of the top 20 most polluting rivers for plastics globally inferred to be in Africa (Lebreton et al. 2017). However, little research on plastic pollution in African freshwater ecosystems has been published (Blettler et al. 2018; Jambeck et al. 2018). To date, microplastics have only been recorded in the African Great Lakes (Biginagwa et al. 2016) and in river sediments and estuaries in South Africa (Naidoo et al. 2015; Nel et al. 2018; De Villiers 2019). The main aim of this chapter was to assess large-scale patterns of plastic pollution along the Orange and Vaal rivers and to estimate the amount of litter delivered to the sea. The results show that microfibres are pervasive pollutants of surface waters in the Orange-Vaal River system, being found at even the most remote sampling sites from the headwaters of the Orange River in Lesotho to its mouth.

Microfibres were particularly abundant in the arid lower reaches of the Orange River in April 2018, before the summer flood waters reached the area, but their concentrations had decreased significantly by August and November 2018. Water levels in the lower reaches of the Orange River were still low when samples were collected in April due to low rainfall in the lower catchment of the river coupled with high levels of water abstraction for agriculture. Heavy rains had fallen in the upper reaches of the Orange and Vaal rivers over the wet season, but this water only reached the Lower Orange River after April (Figure S1). It therefore appears that microfibres accumulate in the Lower Orange River during periods of low flow and presumably are washed out to sea once flood waters from the upper reaches of the catchment flush the area (Barrows et al. 2018a; Hurley et al. 2018; Van Emmerik et al. 2019).

In contrast to microfibres, macrolitter and larger microplastics sampled with the neuston net were most abundant at sites in the highly urbanised and densely populated Middle Vaal River and were largely absent at sites below the confluence of the Orange and Vaal rivers. Gauteng and adjacent densely populated areas thus

act as a major source of plastics into the Vaal River, but these larger items are being retained near point sources either by being buried, washed up on riverbanks or becoming trapped in riparian vegetation (Figure 3.1). These trapped items may then be picked up and sent to landfill or washed downstream during periods of high flow (Hurley et al., 2018).

The concentrations of microfibres found in the Orange-Vaal River system (mean = 1.7 ± 5.1 items·L^{-1}) are similar to those found in rivers in the USA, for example in the Gallatin River watershed (mean = 1.2 items·L^{-1}, Barrows et al. 2018a), the Hudson River (mean = 0.98 items·L^{-1}, Miller et al. 2017 and mean = 1.6 items·L^{-1}, McNeish et al. 2018) and the Muskegon River (mean = 2.9 items·L^{-1}, McNeish et al. 2018). Di & Wang (2018) also report similar levels of contamination in the Yangtze River, China (4.7 ± 2.8 items·L^{-1}), but microplastics and microfibres were orders of magnitude more abundant in the Saigon River, Vietnam (172–519 items·L^{-1}, Lahens et al. 2018). On the other hand, the abundance of net plastics was slightly higher on the Orange and Vaal rivers (mean = 0.34 ± 0.93 items·m^{-2}) than in the Laurentian Great Lakes (mean = 0.043 ± 0.116 items·m^{-2}, Eriksen et al. 2013) but lower than in the Yangtze River, China (mean = 3.4–13.6 items·m^{-2}, Zhang et al. 2015).

It is, however, difficult to directly compare the levels of plastic pollution reported in this chapter to other rivers due to a lack of standardised methodology. For example, Miller et al. (2017), Barrows et al. (2018a) and McNeish et al. (2018) filtered bulk water samples (1–2 L) through 0.45 µm filters while Di & Wang (2018) used a 48 µm mesh, compared to 25 µm in my study. On the other hand, Lahens et al. (2018) filtered bulk water samples through a 2.7 µm mesh while Zhang et al. (2015) used a neuston net with mesh size 112 µm, both of which were finer mesh sizes than what were used in my study and which might explain the higher concentration of plastics found in the Saigon and Yangtze rivers. Different studies also collect water from different depths. For example, neuston nets generally sample the top ~30 cm of the water column, and Miller et al. (2017), Barrows et al. (2018a) and McNeish et al. (2018) collected bulk water samples from just below the surface. Conversely, Di & Wang (2018) collected water using a pump from 1 m below the surface. A recent study showed that microplastics are distributed throughout the

water column (Liedermann et al. 2018) and future studies may therefore benefit from collecting samples from multiple depths as this would greatly improve our understanding of how microplastics are transported in rivers (Moore et al. 2011). A further consideration is that the volume of water filtered can also influence how many microplastics and microfibres are found, particularly in rivers with low levels of pollution (Barrows et al. 2018a; Ryan et al. 2019b). The large variances I observed in bulk water counts within sites suggest that even filtering 10 L of water – which is a greater volume than most previous studies – only provides a rough estimate of differences in plastic and microfibre pollution between sites.

One of the most effective ways of preventing waste plastics entering a river is to identify specific point sources of plastic pollution and then implement mitigation measures to stem the flow of plastics from these sources. For example, if littering is the main source of macroplastics into a river, anti-litter campaigns can teach communities about the impacts of littering (Kiessling et al. 2019) and placing traps on storm-water run-off can prevent litter from entering rivers (e.g. Armitage 2007). However, it is often difficult to identify specific sources given the large intervals between sampling sites (Barrows et al. 2018a). Identifying specific sources typically requires sampling at a smaller spatial scale around potential sources, for example McCormick et al. (2016) showed that microfibres and microplastics were significantly more abundant at sites directly downstream of a WWTP than directly upstream of the WWTP and concluded that the WWTP was acting as a significant source of plastics into the river. The presence of microfibres at even the most remote sampling sites in the Orange and Vaal rivers suggests that some fibres may be transported in the atmosphere (Dris et al. 2016; Allen et al. 2019; Bergmann et al. 2019). However, the accumulation of microfibres and microplastics in the lower reaches of the Orange River during periods of low flow suggests that they are also transported by the river and so may originate in the more densely populated areas farther upstream. Future research should sample on a finer spatial and temporal scale to improve our understanding of where these microfibres and microplastics originate.

By comparison, larger microplastics and macroplastics were most abundant in the Klip and Blesbokspruit rivers that drain urbanised areas south of Johannesburg. Most of the plastics caught in the net were pieces and fragments of larger items such

as plastic food packaging and therefore likely originate from larger items breaking down *in situ* (Figure 3.1). A useful approach which would greatly improve our understanding of how larger plastic items enter these rivers would be to survey the riverbanks and catchments, which would show if littering and/or dumping are major sources (Kiessling et al. 2019). Another approach would be to deploy nets over stormwater outlets, particularly during rain events (Chapter 2). Chemical contaminants accumulate on streets and in stormwater systems during the dry season and therefore are most concentrated in the first flush of rainwater but level off as a storm progresses (Lee et al. 2002). Plastics presumably show a similar pattern and might therefore be most abundant in rivers at the onset of a storm.

It is clear from these results that neuston nets underestimate the abundance and mass of plastic pollution in rivers. Microfibres in particular were orders of magnitude more abundant in bulk water samples than plastics caught in the neuston net, which agrees with other studies (Barrows et al. 2017; Lahens et al. 2018). It is also clear that neuston nets underestimate the abundance of macroplastics, catching only three macroplastic items at a single site. The results of this chapter therefore support the findings of Van Emmerik et al. (2018) who conducted observations for macroplastics on the Saigon River, Vietnam and showed that the global model used by Lebreton et al. (2017), which was based on studies which sampled with neuston nets, underestimated the amount of plastic in the river by at least half. While researchers are starting to collect bulk water samples in rivers (e.g. Barrows et al. 2018a; Lahens et al. 2018), very few estimates of macroplastics in rivers exist and those that have been published mostly sampled using surface booms (e.g. Gasperi et al. 2014). These can be expensive to deploy, need to be emptied on a regular basis and probably only sample a fraction of the macrolitter load. Conducting observations for macroplastics is a more cost-effective method and can be more easily standardised to allow comparison between different river systems (González-Fernández & Hanke 2017; Van Emmerik et al. 2018).

In conclusion, the aim of this chapter was to assess whether the Orange-Vaal River system transports litter from the densely populated inland areas south of Johannesburg to the sea and therefore to what extent it acts as a land-based source of litter into the Atlantic Ocean. As expected, larger plastic items were most

abundant in the highly urbanised and densely populated upper and middle reaches of the Vaal River and therefore do not appear to travel far from source, at least under normal flow conditions. The Orange-Vaal River system therefore does not appear to be a major source of large plastic items into the sea. Only a single item was seen during macrodebris observations and a single hard plastic piece was caught in the net at the Orange River mouth. There was too little data to give a realistic estimate of the amount of macroplastics or net plastics released by the Orange River into the Atlantic Ocean, but litter loads are likely very low. This lends support to the global model predictions of Lebreton et al. (2017) which predicted only 0.095 (0.037–0.355) tonnes of macroplastic litter in the Orange River. Their global models did not, however, consider microfibres. I only found particularly high concentrations of microfibres near the mouth of the Orange River during the period of low flow before the wet season flush and these were likely transported out to sea when flood waters from the upper reaches of the catchment flushed the area. Assuming microfibres are evenly distributed throughout the water column, it is estimated from my three sampling events at the mouth that some 2.5 (1.2–3.5) tonnes of microfibres wash out of the Orange River annually which are likely carried northwards towards Namibia by the Benguela current and may be deposited on local beaches.

Chapter 4

Little evidence that dams in the Orange-Vaal River system trap floating microplastics or microfibres

Equipment used to sample for microplastics on the banks of the Bloemhof Dam.

Photo: K. van Schie

This chapter has been published:

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Author contributions: EAW, VP and PGR planned the field work; EAW and VP conducted the field work; EAW processed all microplastic samples in the lab; EAW analysed the data with assistance from PGR; EAW wrote the draft; PGR assisted with manuscript edits/preparation; VP contributed to drafts.

Abstract

Rivers are thought to be major sources of plastics into coastal seas, but it is unclear whether dams retain floating plastics, thus reducing the number of plastic items reaching the sea. To test if dams trap microfibres and microplastics, I compared microfibres and microplastic (0.025–1 mm) counts in 10 L water samples collected from sites above and below the walls of five major dams on the Orange-Vaal River system in South Africa. I also conducted neuston net tows at each site on the dams to test if larger microplastics (1–25 mm) accumulate behind the dam walls. I sampled each site twice: at the end of the wet season when dam flood gates were open, and at the end of the dry season when flood gates were closed. Most manufactured items were microfibres and densities of plastics in 10 L water (0.21 ± 0.27 items·L^{-1}) and net (0.04 ± 0.16 items·m^{-2}) samples were modest. For 10 L samples, there was a significant interaction between dam and season, with dams on the Vaal River, close to South Africa's industrial heartland, having higher microfibre and microplastic concentrations during dry than wet conditions, whereas the opposite pattern occurred at dams farther downstream or on the less industrialised Orange River catchment. The total mass of microplastic (bulk water and neuston net samples combined) did not differ between dams in the dry season but was highest in the rural Gariep Dam in the wet season due to the accumulation of microplastics at two sites. Overall, there was no effect of site location (above vs. below dam walls) on microplastic abundance nor was there a strong correlation between plastic contamination and distance to the dam wall. The results of this chapter therefore provide little evidence to suggest that dams trap floating microfibres and microplastics. However, the data were noisy (average CV = 184%) and so provide only a rough estimate of differences in contamination among sites.

Introduction

Waste plastics and microfibres are pervasive pollutants of freshwater ecosystems that have been detected in rivers and estuaries in North and South America (Ivar do Sul & Costa 2013; McCormick et al. 2014), Asia (Lin et al. 2018), Africa (Nel et al. 2018), Europe (Mani et al. 2015) and the United Kingdom (Stanton et al. 2019). Plastic pollution may pose a threat to freshwater organisms and fisheries due to entanglement, smothering or ingestion, and ingested microfibres, despite being mostly of natural origin (e.g. cotton and wool), can release harmful chemicals into the gut of an organism (Ladewig et al. 2015; Stanton et al. 2019). However, research on riverine plastic and microfibre pollution has largely focused on the extent to which rivers are sources of pollution into coastal seas (Blettler et al. 2018). Lebreton et al. (2017) estimated that 1.15–2.41 million tonnes of plastic is released into the oceans by rivers annually. Their model projections are based on seven studies, six of which sampled for floating plastics using manta or neuston nets or stationary conical driftnets with mesh sizes ranging between 0.33–3.2mm. They included the presence or absence of dams in their models and concluded that dams may trap approximately 65% of waste plastics entering rivers (Lebreton et al. 2017). However, they noted that their model did not consider differences in retention rates between dams or among different size classes of plastics because there are few estimates of the extent to which dams trap floating plastics, thus preventing them from being released into the sea (Blettler et al. 2018).

Dams retain large amounts of suspended sediments (Syvitski et al. 2005). For example, the construction of the Hòa Bình Dam in Vietnam reduced the downstream flux of sediments by half (Dang et al. 2010) and the Three Gorges Dam in China traps some 172 million tonnes of sediment each year (Hu et al. 2009). It is thus likely that they also trap some plastics and microfibres, especially polymers with higher densities. Mani et al. (2015) suggest that weirs on the Rhine River may act as sinks for plastics, and Zhang et al. (2015) showed that plastics accumulate behind the wall of the Three Gorges Dam. However, no studies have tested whether dams trap plastic or microfibre particles by comparing samples collected above and below dam walls. Given that there are approximately 2.8 million dams, with more than half of the larger rivers in the world having been dammed (Lehner et al. 2011), quantifying

whether dams trap plastic and microfibres will improve our understanding of how they are transported by rivers (Blettler et al. 2018).

Urban centres are major sources of waste plastics entering waterways via direct dumping, stormwater run-off or litter being blown off streets or rubbish dumps (Cable et al. 2017; Moore et al. 2011; Figure 4.1). Similarly, wastewater treatment plants (WWTPs), which are generally more concentrated in urbanised areas with large human populations, often release large quantities of microplastics and microfibres (0.025–5 mm) into rivers due to a lack of an effective filtration system to remove particles of this size (Eerkes-Medrano et al. 2015, but see Talvitie et al. 2017). Given that urban centres are major sources of plastics and microfibres into rivers, it is likely that dams closer to urban areas will have higher levels of plastic and microfibre pollution than dams in more rural areas, especially if dams retain plastics and microfibres within their reservoirs. However, even rural dams may contain high levels of plastics, particularly microplastics and microfibres. Sewage sludge (Li et al. 2018; Nizzetto et al. 2016a) and plastic mulch (Steinmetz et al. 2016) are known to be significant sources of microplastics and microfibres into agricultural soils which can be washed into rivers or dams when it rains. Furthermore, air-borne microplastics and microfibres can be carried long distances from point source areas, resulting in microplastics and microfibres being found in even the most remote habitats (Bergmann et al. 2019; Dris et al. 2016; Zhang et al. 2016). Upon entering a dam, plastic particles or fibres with a higher density than water (1 $g \cdot cm^{-3}$), such as polyethylene terephthalate (PET, 1.4 $g \cdot cm^{-3}$), unexpanded polystyrene (PS, 1.04 $g \cdot cm^{-3}$) and polyvinyl chloride (PVC, 1.18–1.70 $g \cdot cm^{-3}$), as well as natural microfibres made of wool (1.3 $g \cdot cm^{-3}$) or cotton (1.5 $g \cdot cm^{-3}$), are likely to sink due to the decrease in water velocity and turbulence (DeLassus & Whiteman 1999). These may accumulate in bottom sediments or exit the dam via the bottom sluices which are often kept open to generate hydropower, for downstream irrigation and to maintain river functioning (Hoellein et al. 2014; Kowalski et al. 2016; Figure 4.1). Conversely, plastics with a lower density than water such as low- and high-density polyethylene (LDPE, 0.92–0.923 $g \cdot cm^{-3}$; HDPE, 0.941–0.967 $g \cdot cm^{-3}$) and polypropylene (PP, 0.903 $g \cdot cm^{-3}$) may wash up on the banks of the dams where they can be buried by sediment, blown inland or go back into the dam (blown by the wind or washed off if dam levels rise; DeLassus & Whiteman 1999). In the dry season, when the water

levels drop below the level of the dam wall, floating plastics may accumulate behind dam walls (Zhang et al. 2015) while in the wet season, when water levels are high, these floating pieces may pass over dam walls which would result in a more homogenous spread of plastics throughout the catchment area.

The Orange-Vaal River system in South Africa provides an ideal system in which to assess the effect of dams on plastic transport down rivers. While the banks along the Orange River are irrigated, there is little urbanisation and human populations within its catchment are relatively low (Figure 4.2A). Conversely, its largest tributary, the Vaal River, drains the industrial heartland of South Africa and is densely populated. It is therefore not surprising that the Vaal River (and particularly the Upper Vaal) is one of the most polluted rivers in the country (Wepener et al. 2011). However, little is known about levels of plastic contamination in either the Orange or Vaal rivers. The Orange River is one of the most turbid in Africa and delivers large amounts of sediment to the Atlantic Ocean (Compton & Maake 2007). It may therefore act as a conduit of waste plastics and microfibres to the ocean, especially if plastics and microfibres entering the Vaal River flow into the Orange River. However, five major dams have been built in this system: the Gariep and Van De Kloof Dams on the Orange River and the Grootdraai, Vaal and Bloemhof dams on the Vaal River (Figure 4.2), which may act as sinks for floating plastics and microfibres. In this chapter, I report levels of microplastic and microfibre (0.025–25 mm) pollution in each of the five dams. I compare microplastic and microfibre abundance at sites above and below the dam walls and consider whether particles accumulate behind the walls by comparing densities at sites close to the walls to those further away. If dams do trap microplastics and microfibres, I predict that microplastics and microfibres will be more abundant at sites above than below dam walls, at sites close to dam walls, and in the dams with densely populated catchments (Grootdraai and Vaal dams) compared to dams with catchments containing lower human population densities (Bloemhof, Gariep and Van Der Kloof dams).

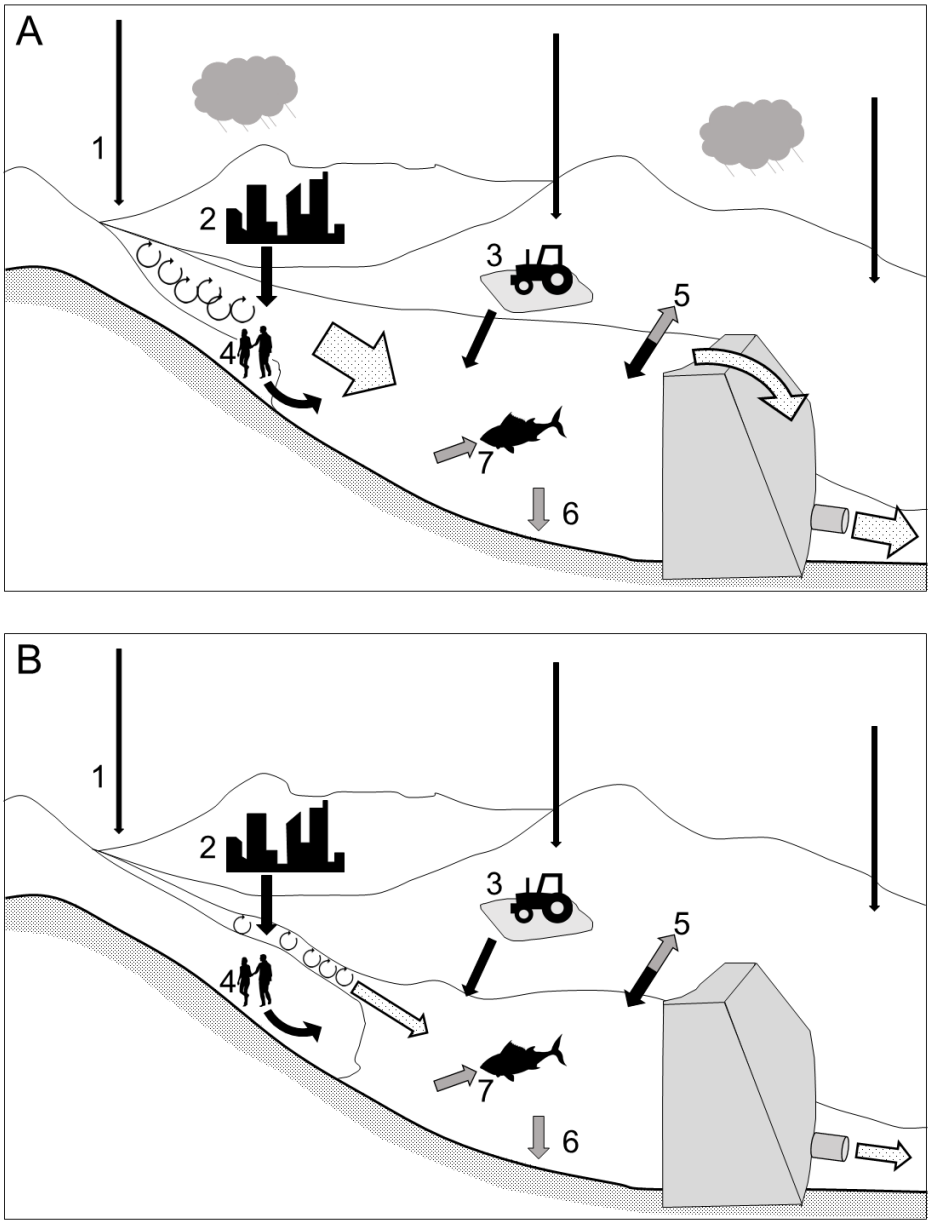


Figure 4.1: Schematics showing how plastics are likely to move through a river and dam catchment area in the wet season (A) when the dam is overflowing and the dry season (B) when water is only flowing through bottom sluices. The main sources (black arrows) of plastic pollution are atmospheric fallout (1), urban centres from which waste can be blown or released via stormwater run-off (2), run-off from agricultural croplands (3) and direct littering on the banks of the dam or river which can be blown into the water (4). The main sinks for plastics (grey arrows) are being washed up and retained on the banks of the dam (5), although these plastics can also be blown back into the dam or inland, or resuspended if dam levels rise, and plastics that sink and accumulate in bottom sediments (6). Plastics may also be ingested by biota (7). The stippled arrows show water flowing into and out of the dams and the circular arrows show that water velocity and turbulence are higher in the river than in the dam.

Methods

Field data

I collected bulk water and neuston net samples from each of the five major dams on the Orange and Vaal rivers (Table 4.1, Figure 4.2). The Vaal Dam has the most densely populated catchment of the five dams and is less than 100 km downstream from the centre of Johannesburg, the largest urban centre in South Africa. The Grootdraai Dam has the second most densely populated catchment and is approximately 160 km from Johannesburg and less than 100 km from Ermelo, the capital of the Mpumalanga Province. The Bloemhof Dam and the two dams on the Orange River have less densely populated catchments and are situated in rural agricultural areas far from large urban centres (Table 4.1, Figure 4.2A).

Table 4.1: Storage capacity, distance upstream of the confluence of the Orange and Vaal rivers, human population and number of wastewater treatment plants (WWTPs) within 50 km of each dam. VDK = Van Der Kloof Dam.

	Orange River		**Vaal River**		
	Gariep	**VDK**	**Grootdraai**	**Vaal**	**Bloemhof**
Storage (million m^3) [a,b]	5 500	3 200	350	2 600	1 200
Distance (km)	315	185	1 040	840	400
Human population [c]	95 000	43 000	406 000	2 100 000	176 000
WWTPs [d]	3	3	5	8	3

[a] DWAF 2019

[b] DWS 2019b

[c] Worldpop 2013

[d] Van Deventer et al. 2018

I sampled each site at the end of the wet season (11–27 April 2018) when all dams were overflowing and resampled at the end of the dry season when flood gates were closed and water was only flowing from the bottom sluices (8–21 November 2018). One site on the Van Der Kloof Dam was not accessible in the dry season.

To test for microplastic and microfibre pollution, I collected bulk water samples from below each of the dam walls (downstream of the outlet and within 500 m of the dam wall) and at 3–4 sites above the dam walls (Table S2). I collected replicate bulk water samples (n = 3–6) at each site by wading to knee-height and filling a white 10 L plastic container, which was triple-rinsed with river water prior to filling. During the wet season, I collected three replicates per site; however, I increased this to six replicates per site in the dry season due to high variation between replicates. I gravity-filtered each sample through a triple-rinsed 25 µm nylon mesh filter, which I then placed in a clean metal foil envelope, dried at 20°C and stored at room temperature until I could examine them under a stereomicroscope at the University of Cape Town (UCT). Due to a faulty oven, 10 bulk water filters from the wet season trip were destroyed (below the Gariep Dam wall [n = 3], Gariep Dam beach 1 [n = 2], Van Der Kloof Dam beach 3 [n = 2], below the Van Der Kloof Dam wall [n = 2] and Van Der Kloof Dam beach 4 [n = 1]). I excluded these sites from the analyses.

To test the water for larger microplastics, I towed a neuston net at each site above the dam wall. The neuston net samples the top 30 cm of the water column, is 60 cm wide and has a mesh size of 300 µm. I waded waist-deep with the neuston net at a constant speed for 15 minutes adjacent to the shore. Initially, I waded in one direction and then doubled back so that the same stretch of shoreline was sampled at least twice, but sometimes three times at sites with shorter beaches. I completed two tows per site, covering a mean distance of 403 ± 47 m per tow. The neuston net was attached to ropes and was towed at least 3–4 m behind the person to minimise the chance of sampling within their wake. After each tow, I used water filtered through a 25 µm filter to rinse the contents of the neuston net into the cod-end, which I then emptied onto a 330 µm metal mesh sieve and finally rinsed into small plastic sampling jars. I visually sorted the samples by picking out any plastic pieces with forceps and stored these in 70% alcohol until further processing in the lab at UCT.

Additionally, I also collected bulk water and neuston net samples at the end of the wet season using a motorised boat. On the Gariep Dam, I collected bulk water samples by boat from within 500 m of the dam wall (n = 6; 30.6222°S 25.5089°E), 5 km from the wall (n = 3; 30.6144°S 25.5618°E) and 10 km from the wall (n = 6; 30.6171°S 25.6028°E), and I collected neuston net samples from within 500 m of the wall (n = 2) and 10 km from the wall (n = 2). On the Vaal Dam, I collected bulk water

(n = 3 at each site) and neuston net (n = 2 at each site) samples from within 500 m of the wall (26.8858°S 28.1165°E) and 2 km from the wall (26.8860°S 28.1375°E). Sampling from the boat was similar to shoreline samples but I collected bulk water samples by lowering a triple-rinsed metal bucket from the boat with a sisal rope (brown natural fibre) which I used to fill the 10 L container. I collected three to six replicates (30–60L) at each site. I also towed the neuston net for 15 minutes behind the boat at a constant speed of approximately 0.4 $m \cdot s^{-1}$. I towed the net at least 10 m behind the boat to ensure that the neuston net was outside of the boat's wake.

Sample processing

In the lab, I examined all bulk water filters under the stereomicroscope. Some bulk water filters (n = 16) were saturated with sediment, making identification of microplastics and microfibres difficult. To remove the sediment, I used density separation. I washed the contents of these filters into a glass beaker using Milli-Q water and added 100 ml of saturated sodium chloride solution (1.2 $g \cdot cm^{-3}$, Thompson et al. 2004). To reduce the chance of contamination, I pre-filtered the sodium chloride solution through a 25 µm nylon mesh filter (Nel et al. 2018). I then stirred the contents of the beaker vigorously for 2 minutes using a metal spoon and then allowed it to settle for 10 minutes which I deemed long enough as most sediment was coarse sand and sedimented out quickly. Thereafter, I filtered the supernatant through a 25 µm filter which I examined under the stereomicroscope. To decrease the chance of contamination, I triple-rinsed all equipment with Milli-Q water prior to use.

I counted the number of microplastics and microfibres per filter under the stereomicroscope. I identified microfibres based on standard criteria recommended by Barrows et al. (2018): uniform width, uniform artificial colour (e.g. bright blue and red) and lack of visible cellular structures. I identified plastic fragments by their unnatural colours and sharp edges (Hidalgo-Ruz et al. 2012). To assess the level of atmospheric contamination in the lab, I placed a procedural blank (triple-rinsed 25 µm filters) next to the microscope and exposed it to the air each time I opened a new sample. I counted the number of fibres on each procedural blank and subtracted this from the total count for the corresponding sample.

I air-dried the neuston net samples and classified each item according to its colour and type (flexible packaging, fibre, hard piece/fragment, paint, paper, industrial plastic pellet or polystyrene) and weighed each item to the nearest 0.1 mg. I found one whole crisp packet (>25 mm) in the second tow at Grootdraai beach 2 in the wet season. This single macroplastic item made up 17.7% of the total mass of neuston net plastics and I excluded it from further analyses.

Data analyses

The number of items·L^{-1} in bulk water samples and items·m^{-2} in neuston net tows were not normally distributed (Shapiro-Wilk normality test; bulk water: $W = 0.783$, $p < 0.0001$, neuston net: $W = 0.285$, $p < 0.001$). I therefore used non-parametric Mann-Whitney tests to compare microplastic and microfibre contamination in bulk water samples between sites collected within dams to those collected below the dam walls. I also used Mann-Whitney tests to test if microplastics and microfibres were more or less abundant in bulk water and neuston net samples collected in the wet season by boat versus those collected from the shore. Here, to control for variation between dams, I analysed samples from the Vaal and Gariep dams separately (i.e. I compared boat samples to shore samples collected in the same dam). I used linear correlations to see if there was a relationship between the distance to the dam wall and the concentration of microplastics and microfibres in bulk water and neuston net samples collected at a site. To test if floating microplastics and microfibres accumulate behind dam walls, I used non-parametric Kruskal-Wallis tests to compare the abundance of microplastics and microfibres collected by boat at sites within 500 m of the wall to sites further away. Again, I analysed samples from the Vaal and Gariep dams separately (i.e. I compared boat samples to shore samples collected in the same dam). Where significant, I used a *post-hoc* Dunn's test to show significant differences among cases and I applied a Bonferroni correction to decrease the chance of committing a Type 1 error. I also used Kruskal-Wallis and *post-hoc* Dunn tests to assess whether there were large-scale differences in microplastic and microfibre abundances between the dams. Here, I only included sites on the dams. Where applicable, I did each of these analyses with seasons combined and then for each season separately. To assess how much variation there

was in microplastic and microfibre abundance at each site, I calculated coefficients of variation (CVs) for each site and then averaged these across all sites.

I used a generalised linear mixed model (GLMM) to assess whether dam, season and site location (above vs. below dam wall) played a role in explaining microplastic and microfibre abundance in bulk water samples at a given site. I fitted the model using the *lme4* package in R (Bates et al. 2015; R Core Team 2019). I used a negative binomial distribution to account for the data being over-dispersed and modelled the number of items·10 L^{-1} (c.f. per litre) to ensure that the response variable was an integer. Due to the nested nature of the data (multiple sites within each dam), I included a random effect of site. I also included an interaction term between season and dam. I ran the model for 100 000 iterations using the "bobyqa" optimiser and then calculated the least squares mean abundance of microplastics and microfibres for sites above and below the dam walls and compared these using the "cld" function in the *multcomp* package in R (Hothorn et al. 2008; R core team 2019). I applied a Bonferroni correction to reduce the chances of committing a Type 1 error.

To assess overall pollution, I calculated the total mass of microplastics and microfibres found at each site on the dam by combining densities found in the bulk water and neuston net samples. First, I estimated the mass of fibres by calculating the volume of each fibre (assuming a cylindrical shape and using the length and width of each fibre measured using the graticule of the stereomicroscope) and then multiplying it by a density of 1.35 g·cm^{-3} (the average density of wool, cotton and polyester which are the most common fibre polymers, Stanton et al. 2019). To make measures of contamination in bulk water and neuston net samples comparable, I extrapolated the total mass of microfibres (measured per litre) to total mass·m^{-2} by assuming that the bucket and 10 L container sampled the top 30 cm of the water column, the same depth sampled by the neuston net (i.e. I multiplied the density of microfibres per litre by 300 to give the density per m^2).

I used a hurdle model to assess if distance to the dam wall, dam and season explained significant amounts of the variation in the total mass of microplastics and microfibres at a given site. This model has two parts and was chosen because the data were zero-inflated, right-skewed and semi-continuous. Firstly, I fit a generalised

linear model (GLM) with a binomial distribution to the data. This part models whether microplastics and microfibres were found in a given replicate (i.e. zero values vs. non-zero values). Secondly, I modelled replicates where microplastics and microfibres were found (the non-zero data) using a GLM fit with Gamma distribution (to account for the right-skewed semi-continuous data). Here, I used distance to dam wall, dam and season as explanatory variables, as well as an interaction between dam and season. I conducted all analyses in R version 3.6.0 (R Core Team 2019) and report results ± 1 SD unless otherwise indicated.

Results

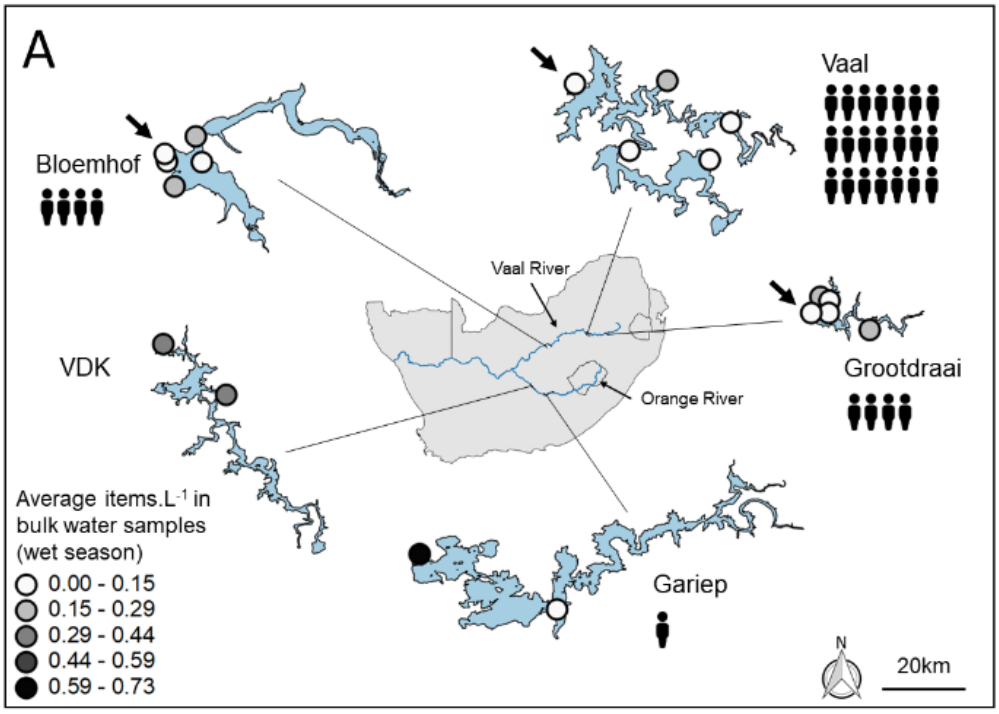
Bulk water samples

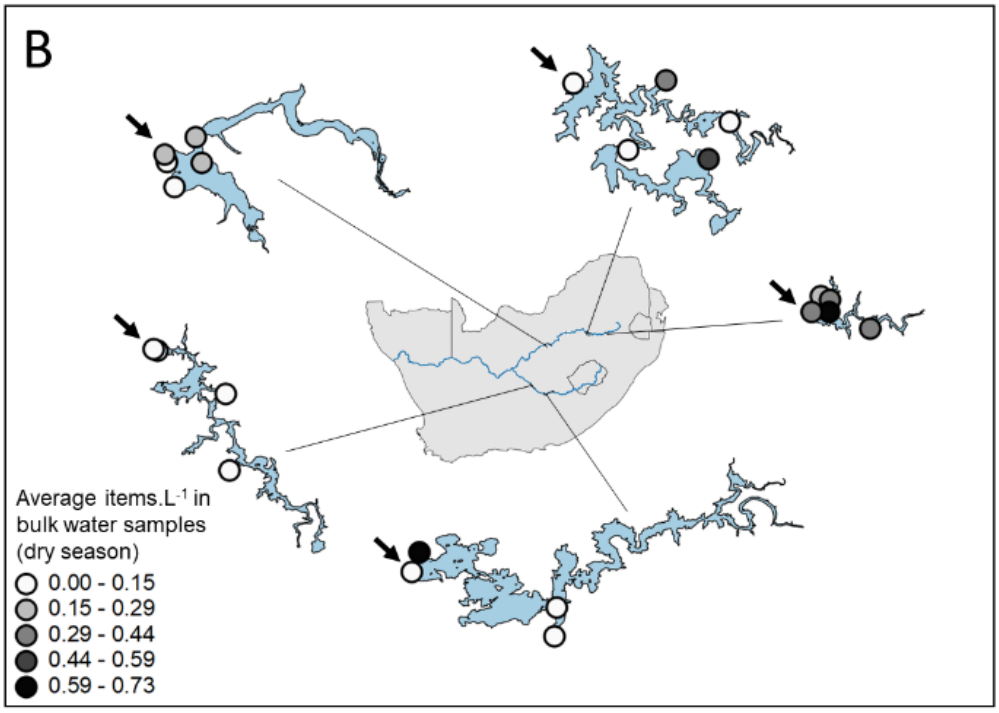
The bulk water samples contained 505 potential microplastics (mean = 0.23 ± 0.27 items·L^{-1}) of which 98% (n = 497) were fibres and only 2% (n = 8) were hard plastic fragments. As noted earlier, µ-FTIR or Raman spectroscopy has revealed that most microfibres (>90%) found in rivers (Stanton et al. 2019) and oceans (Suaria et al. 2020) are of natural origin (e.g. cotton or wool fibres) rather than plastic. I attempted to identify the fibres found in my bulk water samples using a Raman spectroscope at the UCT Electron Microscope Unit. However, the Raman spectroscope was fitted with a green laser (532nm wavelength) which caused the samples to fluoresce to such an extent that it was not possible to acquire a clear spectrum. To fix this, a red laser is required (785nm wavelength) but unfortunately I did not have access to one. Effort will be made to identify the fibres at a later stage.

Most fibres found in this study were blue (92%, n = 458), while the rest were red (7%, n = 34), purple (<1%, n = 2), transparent (<1%, n = 1), green (<1%, n = 1) and pink (<1%, n = 1). Of the hard fragments, 38% were blue (n = 3), 25% were white (n = 2), 25% were orange (n = 2) and 13% were green (n = 1).

The procedural blanks accumulated 0.25 ± 0.59 fibres·$filter^{-1}$ (median = 0). After controlling for atmospheric contamination, the average density was 0.21 ± 0.27 items·L^{-1} with slightly higher contamination in the wet season (0.26 ± 0.21 items·L^{-1}) than the dry season (0.21 ± 0.28 items·L^{-1}), but this difference was not significant (Mann-Whitney test: W = 3703.5, p = 0.51). Despite the increased number of

replicates collected in the dry season, the data for both seasons were strongly right-skewed and the standard errors were similar (SE wet season = 0.029, SE dry season = 0.023).





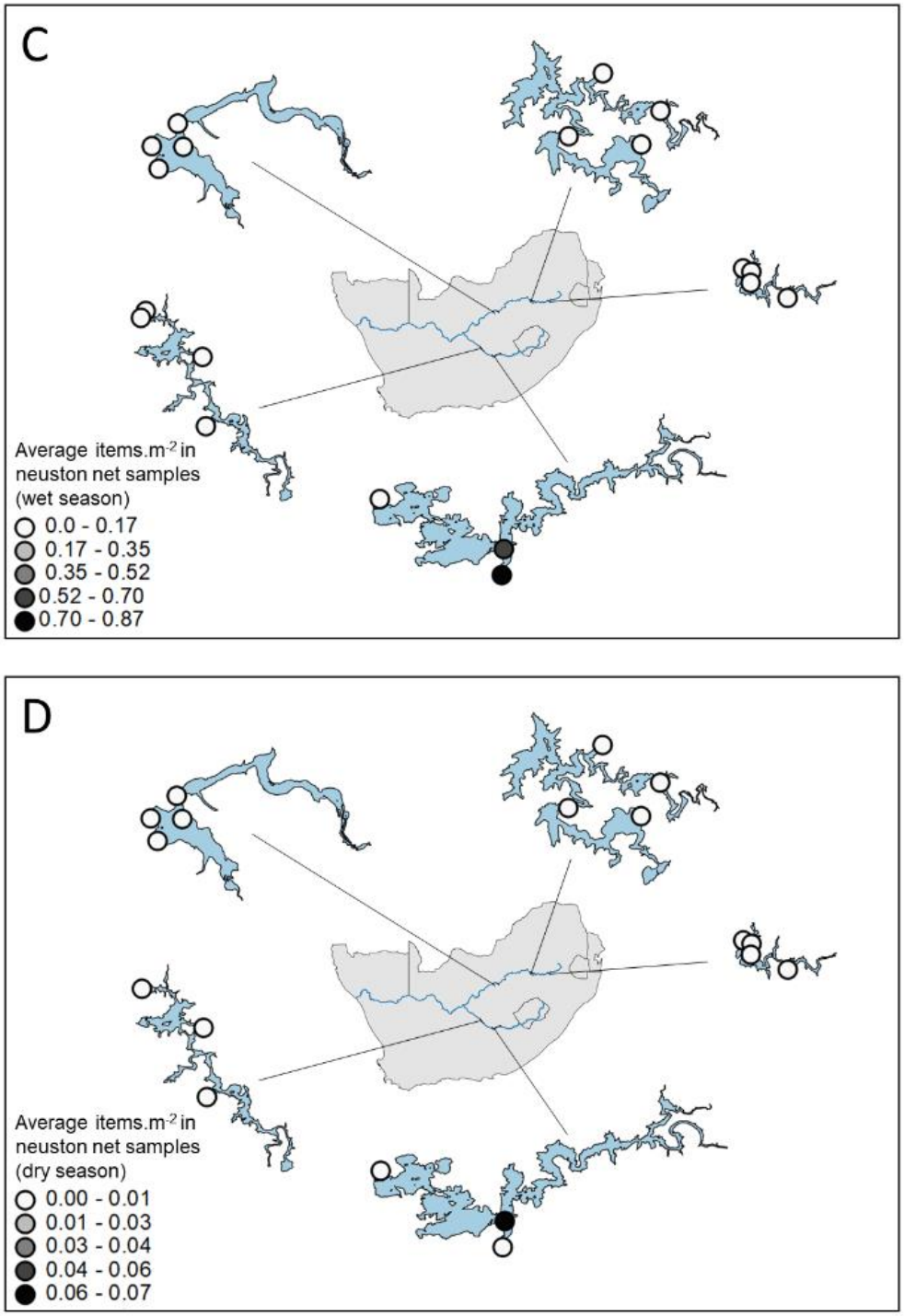


Figure 4.2: The distribution of sampling sites across five dams on the Vaal and Orange rivers in South Africa. The black arrows in panels A and B show sites below the dam walls. The colours of the circles indicate the average number of microplastic items found in bulk water (panels A and B) and neuston net (panels C and D) samples collected at each site in the wet (panels A and C) and dry (panels B and D) seasons. The human silhouettes next to each dam in panel A indicate the human population size within 50 km of each dam, where each silhouette represents 100 000 people (data from Worldpop 2013). The human population around the Van Der Kloof (VDK) Dam is <100 000 people.

The average densities of microplastics and microfibres were 2–4 times higher at sites on the dams than below the dam walls (Figure 4.2A, Figure 4.2B, Table S2). Densities of microplastics and microfibres were significantly higher at sites above than below the dam walls when seasons were combined (mean above: 0.21 ± 0.17 items·L^{-1}; mean below: 0.11 ± 0.06 items·L^{-1}; Mann-Whitney test: W = 3 646.5, p <0.05) and during the wet season (mean above: 0.13 ± 0.08 items·L^{-1}; mean below: 0.03 ± 0.06 items·L^{-1}; Mann-Whitney test: W = 233, p <0.05), but not during the dry season (mean above: 0.23 ± 0.22 items·L^{-1}; mean below: 0.14 ± 0.10 items·L^{-1}; Mann-Whitney test: W = 1 813.5, p = 0.302). During the dry season, microplastics and microfibres were more abundant at sites above the walls of the Gariep, Grootdraai and Vaal dams than at sites below their respective walls, but were slightly less abundant at sites on the Bloemhof and Van Der Kloof dams than below their walls (Figure 4.3, Table S2). Overall, the GLMM predicted higher concentrations at sites above than below the walls (Table 4.2, Figure 4.4). However, this effect of location (above vs. below the dam wall) was not significant, most likely due to the large variation in microplastic and microfibre concentrations between sites, as seen by the high CVs (mean CV per site = 124 ± 44%, range = 51–201%) and large spread of points in Figure 4.4. The model estimated the least squares mean for sites above the dam walls to be 0.155 ± 0.026 (SE) items·L^{-1}, which was not significantly different from the least squares mean estimate of 0.106 ± 0.035 items·L^{-1} for sites below the dam walls (p >0.05).

Microplastics and microfibres were most abundant at beach 3 on the Gariep Dam in the wet season (mean = 0.73 ± 0.49 items·L^{-1}) and beach 4 on the Grootdraai Dam in the dry season (mean = 0.73 ± 0.48 items·L^{-1}) (see black circles in Figure 4.2B, labelled points in Figure 4.5A and Table S2). Both these sites were situated in bays close to the dam walls, but there was no significant correlation between microplastic and microfibre concentration at a site and the distance from that site to the dam wall in either season (wet season: r = -0.29, p = 0.275; dry season: r = -0.20, p = 0.417) (Figure 4.5A).

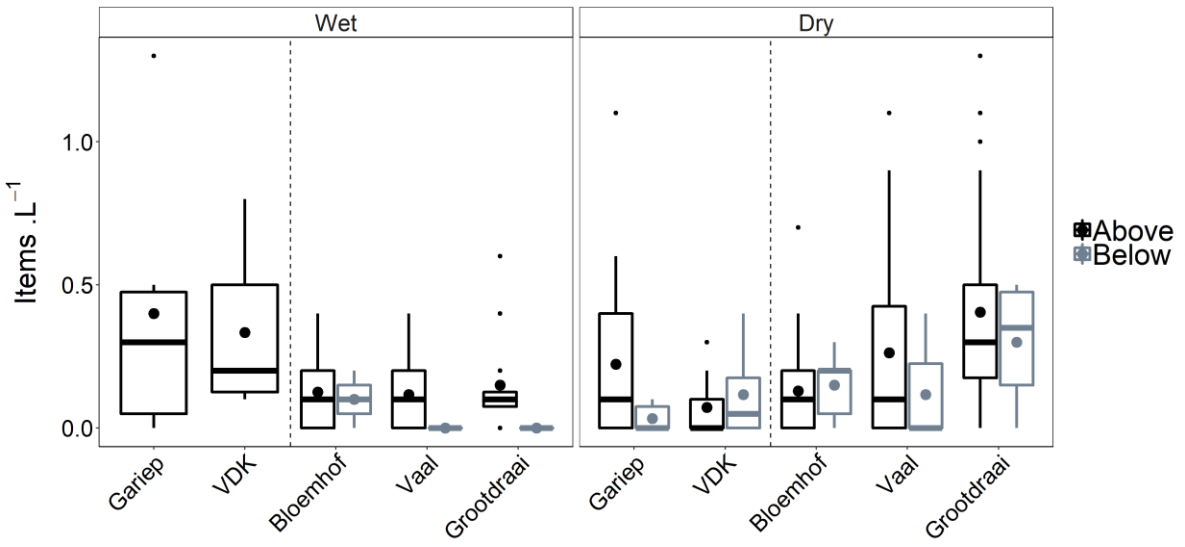


Figure 4.3: Boxplots of the concentration of microplastics and microfibres found in bulk water samples collected above and below the dam walls of each dam in the wet (left panel) and dry (right panel) season. The thick middle lines indicate the median and the large circles the mean concentration. The thin horizontal lines show the interquartile range while the small circles show outliers. The dashed vertical lines separate dams on the Orange River (Gariep and Van der Kloof [VDK]) from those on the Vaal River (Bloemhof, Vaal and Grootdraai). Samples collected from below the Orange River dams in the wet season were damaged and so were excluded from analyses.

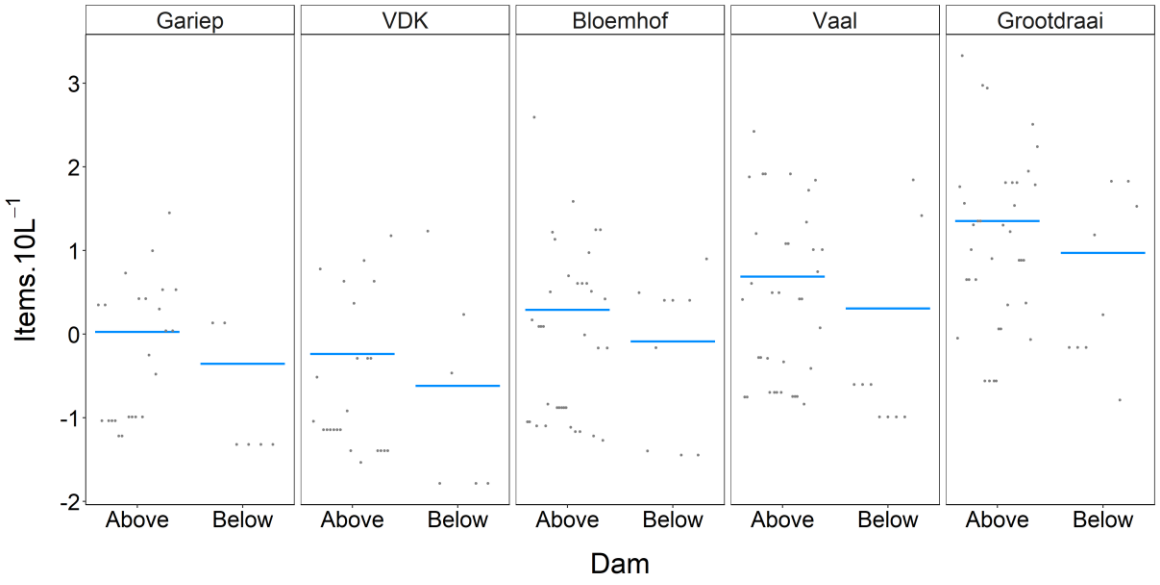


Figure 4.4: The average count of microplastics per 10 L as predicted by the GLMM at sites above and below the walls of the five dams in the wet and dry seasons.

Table 4.2: GLMM results showing the effect (± standard error) of season, dam, site location and an interaction between season and dam on mean abundance of microplastics (items·10 L^{-1}) found in bulk water samples collected from 24 sites at five dams on the Orange and Vaal rivers. Estimates are presented on the log-scale.

	Estimate (± SE)	**z-value**	**P**
Season (dry) [a]	0.09 ± 0.37	0.249	0.803
Site location (below) [b]	-0.38 ± 0.37	-1.046	0.296
Dam main effect [c]			
Vaal	-0.35 ± 0.58	-0.599	0.549
Grootdraai	0 ± 0.56	-0.011	0.992
Gariep	0.31 ± 0.67	0.460	0.645
VDK	1.32 ± 0.67	1.974	**0.048**
Interaction effect [d]			
Vaal	0.74 ± 0.54	1.373	0.170
Grootdraai	1.07 ± 0.52	2.066	**0.039**
Gariep	-0.57 ± 0.61	-0.943	0.345
VDK	-1.85 ± 0.73	-2.518	**0.012**
Intercept	0.20 ± 0.40	0.492	0.623

[a] Estimate indicates expected microplastic count in the dry season relative to the wet season

[b] Estimate indicates expected microplastic count below the dam wall relative to above the wall

[c] Estimates indicate expected microplastic count at the given dam relative to the reference dam (Bloemhof Dam)

[d] Estimates indicate expected microplastic count at the given dam in the dry season relative to the same dam in the wet season

At the end of the wet season, microplastic and microfibre concentrations were slightly higher in the two Orange River dams. Conversely, at the end of the dry season, microplastics and microfibres were most abundant in the densely populated Grootdraai and Vaal dams, particularly in the Grootdraai Dam where concentrations were 2–6 times higher than in the less densely populated Bloemhof, Gariep or Van Der Kloof dams (Figure 4.3). The high concentrations at sites in the Grootdraai Dam at the end of dry season resulted in a significant difference between dams (Kruskal-Wallis test: χ^2 = 16.44, p <0.005), with significantly more microplastics and microfibres in the Grootdraai Dam (mean = 0.40 ± 0.23 items·L^{-1}) than the Bloemhof (mean = 0.13 ± 0.07 items·L^{-1}) and Van Der Kloof dams (mean = 0.072 ± 0.07

items·L^{-1}) (Dunn's tests: Grootdraai–Bloemhof: Z = 2.945, p <0.05; Grootdraai–Van Der Kloof: Z = 3.732, p <0.005). However, the differences between dams was not significant at the end of the wet season (Kruskal-Wallis test: χ^2 = 5.291, p = 0.259). Overall, when seasons were combined, the concentration of microplastics and microfibres was 1.3–2.5 times higher in the Grootdraai Dam than the other dams, but this difference was not quite significant (Kruskal-Wallis test: χ^2 = 9.223, p = 0.056). The seasonal differences between dams resulted in a strong interaction between dam and season in the GLMM (Table 4.2). The model predicted microplastic and microfibre concentrations to be higher in the Orange River catchment in the wet season but higher in the densely populated Vaal and Grootdraai dams in the dry season. The Bloemhof Dam showed little difference between seasons (Figure 4.6).

Microplastics and microfibres were more abundant in samples collected by boat close to the dam wall than those further away from the wall on the Gariep Dam, but there was no significant pattern in the Vaal Dam, where sampling was more spatially constrained (Kruskal-Wallis tests: Vaal Dam: χ^2 = 0.20, p = 0.658; Gariep Dam: χ^2 = 6.58, p <0.05). On the Gariep Dam, microplastics and microfibres were almost twice as abundant close to the dam wall (mean = 0.55 ± 0.15 items·L^{-1}) than 10 km away (mean = 0.33 ± 0.25 items·L^{-1}) and 5 times more abundant than samples collected 5 km away (mean = 0.10 ± 0.17 items·L^{-1}). The difference between samples collected close to the dam wall and those collected 5 km away was significant (Dunn's test: Z = 2.50, p <0.05), but not significant between those collected close to the wall and 10 km away (Dunn's test: Z = 1.56, p = 0.36). On the Gariep Dam, microplastic and microfibre concentrations in boat samples were similar to those collected from the shore (mean boat: 0.37 ± 0.26 items·L^{-1}; mean shore: 0.40 ± 0.49 items·L^{-1}; Mann-Whitney test: W = 39.5, p = 0.695). Conversely, there were significantly more microplastics and microfibres found in boat samples than shore samples on the Vaal Dam (mean boat: 0.43 ± 0.13 items·L^{-1}; mean shore: 0.12 ± 0.21 items·L^{-1}; Mann-Whitney test: W = 5.5, p <0.05).

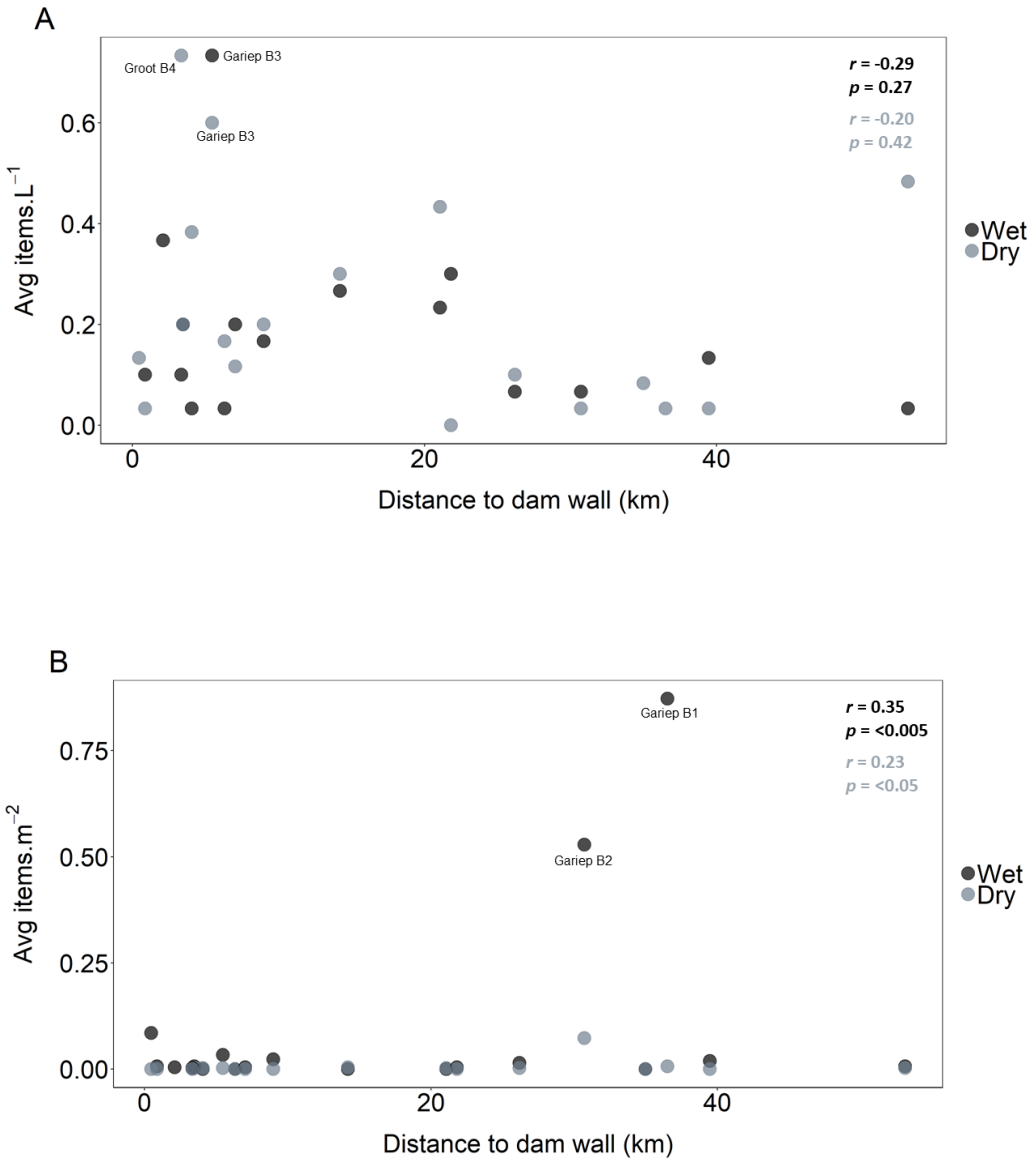


Figure 4.5: Mean number of plastics found in (A) bulk water and (B) neuston net samples at each site vs. distance from that site to the dam wall in the wet and dry seasons. Sites with unusually high levels of contamination are labelled: Groot B4 = beach 4 on the Grootdraai Dam, Gariep B1, B2, B3 = beaches 1, 2 and 3 on the Gariep Dam. Correlation coefficients and their respective p-values are shown in the top right corner.

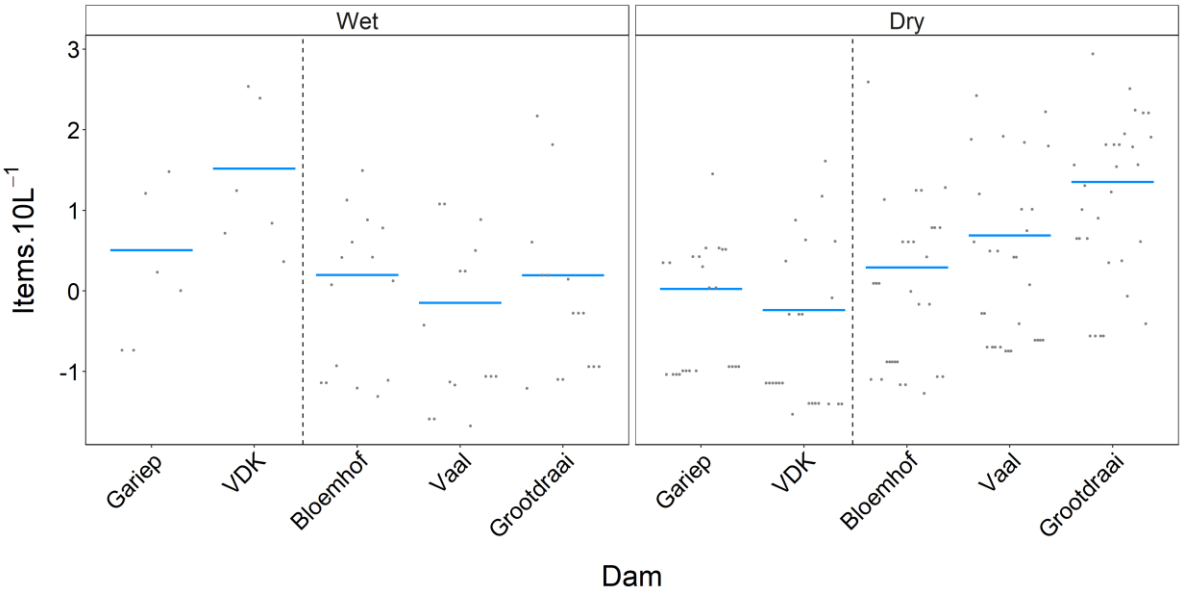


Figure 4.6: The average count of microplastics and microfibres (items·10 L^{-1}) as predicted by the GLMM at the five different dams for the wet and dry seasons. The dashed vertical lines separate dams on the Orange River (Gariep and Van der Kloof [VDK]) from those on the Vaal River (Bloemhof, Vaal and Grootdraai).

Neuston net samples

The neuston net samples contained 827 plastic items (mean = 0.04 ± 0.16 items·m^{-2}) and there was no significant correlation between the number (wet season: r = -0.15, p = 0.59; dry season: r = -0.21, p = 0.40) or mass (wet season: r = -0.09, p = 0.74; dry season: r = -0.16, p = 0.53) of particles found in bulk water and neuston net samples in either season (Figure 4.7). Most items found in the neuston samples were hard plastic pieces (85%, n = 703) while the rest were pieces of flexible plastic packaging (9%, n = 76), polystyrene (3%, n = 28), plastic fibres (1%, n = 10), plastic twine (<1%, n = 4), grey foam pieces (<1%, n = 3), white paper pieces (<1%, n = 2) and transparent plastic pellets (<1%, n = 1). Of the hard plastic pieces, most were transparent (39%, n = 272), white (35%, n = 243) and blue (15%, n = 102) and most of the flexible plastic packaging pieces were transparent (45%, n = 34) (see Table S3 for details). Large numbers of industrial pellets were found washed up at beach 2 on the Vaal Dam but no pellets were found in the neuston net samples collected at this site.

Overall, plastic contamination in neuston samples tended to be low with 17 of the 19 sites having a mean of less than 0.05 items·m^{-2}. No plastics were found at any

sites on either the Bloemhof or Van Der Kloof dams at the end of the dry season (Figure 4.8, Table S2). However, two sites on the Gariep Dam (beaches 1 and 2, black circles in Figure 4.2C, Figure 4.2D, outlying points in Figure 4.7, Table S2), both situated in a bay close to the town of Oviston, had particularly high levels of contamination. Contamination was more than seven times higher at these sites than at any other site, resulting in the overall mean number of plastics found at all sites being higher than the median (mean = 0.046 ± 0.166 items·m^{-2}, median = 0 items·m^{-2}). These two beaches were more than 30 km from the dam wall, resulting in a significant positive correlation between plastic contamination and distance to the dam wall in both seasons (wet season $r = 0.35$, $p < 0.005$; dry season $r = 0.23$, $p < 0.05$; Figure 4.7B).

The average density of plastics found in the neuston net samples collected on the Gariep Dam was 20–100 times higher than the other dams when seasons were combined and 20–230 times higher in the wet season. This was due to the high abundance of plastics at sites B1 and B2 on the Gariep Dam (Figure 4.8, Table S2). Overall, plastics were significantly more abundant in the Gariep Dam than the Bloemhof, Grootdraai and Van Der Kloof dams (Dunn tests, all p-values <0.05), but the difference between the Gariep and Vaal dams was not significant (Dunn's test, $Z = 2.574$, $p = 0.101$). Similarly, in the wet season, significantly more plastics were found in the Gariep Dam than the Bloemhof, Grootdraai and Van Der Kloof dams (Dunn tests, all p-values < 0.05), but not in the Vaal Dam (Dunn's test, $Z = 2.451$, $p = 0.142$). In the dry season, contamination at sites B1 and B2 on the Gariep Dam was still higher than all other sites, but was an order of magnitude lower than in the wet season (wet season: mean B1 = 0.872 ± 0.208 items·m^{-2}, mean B2 = 0.528 ± 0.026 items·m^{-2}; dry season: mean B1 = 0.006 ± 0.009 items·m^{-2}, mean B2 = 0.073 ± 0.056 items·m^{-2}). While contamination on the Gariep Dam was 13–17 times higher than the other dams, the only significant difference in the dry season was between the Gariep and Bloemhof dams (Dunn's test, $Z = 2.996$, $p < 0.05$), although the difference between the Gariep and Van Der Kloof dams was close to significant (Dunn's test, $Z = 2.803$, $p = 0.051$) (Figure 4.8).

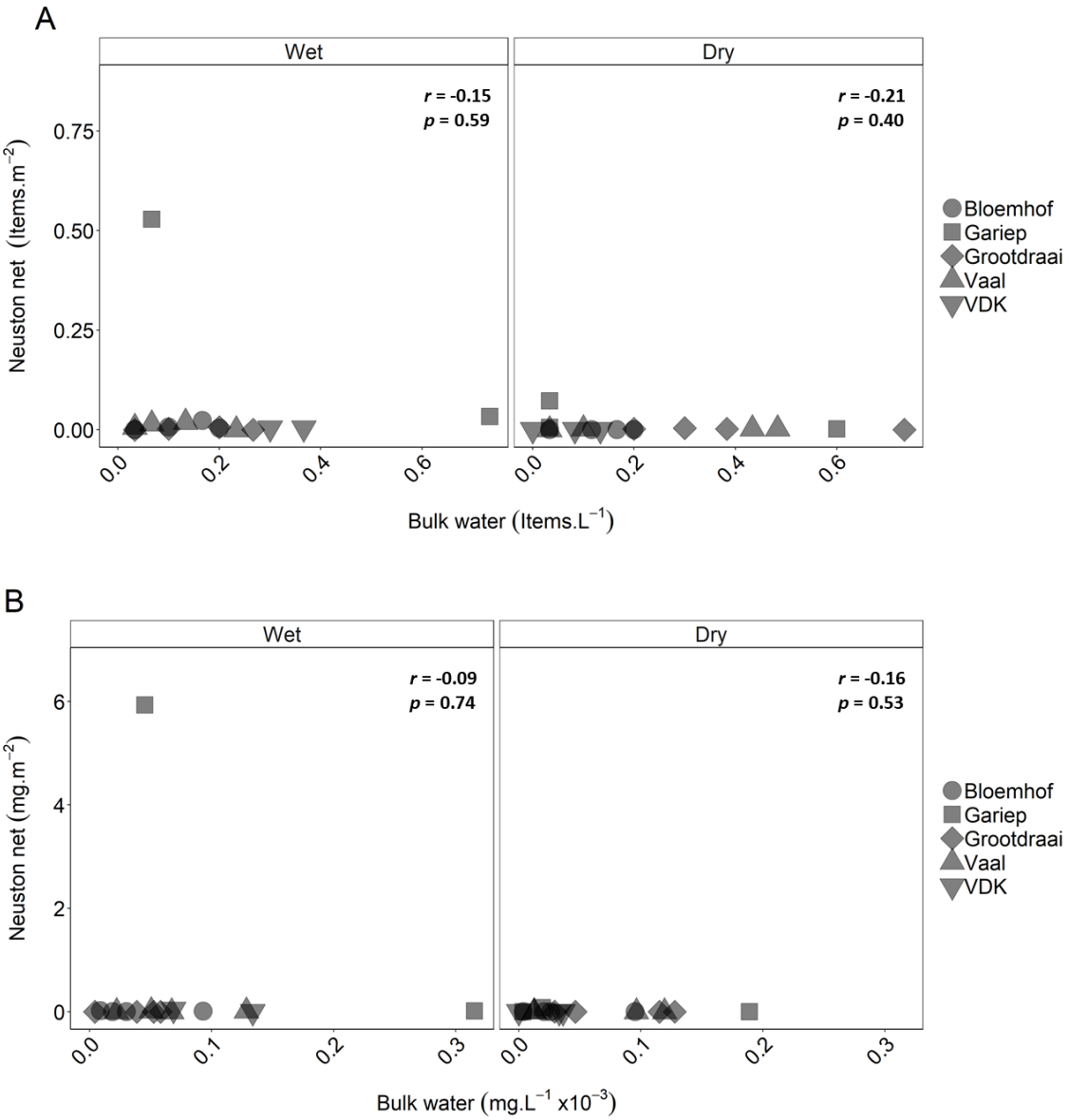


Figure 4.7: Mean (A) count and (B) mass of plastic and microfibres found in bulk water and neuston net samples per site in each dam and in each season. Correlation coefficients and their respective p-values are shown in the top right corner of each panel.

Only five plastic items were found in the neuston net samples collected by boat (overall mean = 0.003 ± 0.003 items·m^{-2}). On the Vaal Dam, only two plastic items were found, both in the second tow at the site within 500m of the dam wall and there was no significant difference between plastic concentrations in boat samples and shoreline samples (mean boat = 0.002 ± 0.004 items·L^{-1}; mean shore = 0.010 ± 0.009 items·L^{-1}; Mann-Whitney test: W = 25.5, p = 0.11). One plastic item was found in each tow at the dam wall of the Gariep Dam, and one plastic item in one of two

tows at the site farther away from the dam wall. Plastics were significantly more abundant in shoreline samples than boat samples collected on the Gariep Dam (mean boat = 0.003 ± 0.002 items·L^{-1}; mean shore = 0.478 ± 0.389 items·L^{-1}; Mann-Whitney test: W = 24, p <0.05).

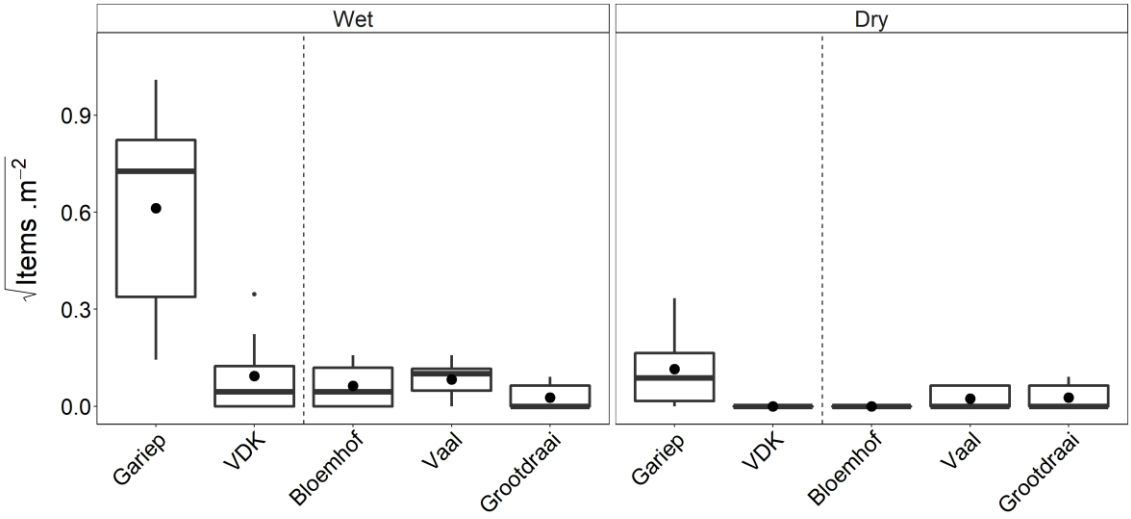


Figure 4.8: Boxplots of the number of plastic items·m^{-2} found in samples collected at each dam in the wet (left panel) and dry (right panel) season using a neuston net. For clear visualisation, the data have been transformed using a square-root transformation. Conventions as Figure 4.3.

Total contamination

The abundance of plastics found in neuston net tows at sites B1 and B2 on the Gariep Dam resulted in the average mass of plastics (bulk water and neuston samples combined) in shore samples being 5–87 times higher on the Gariep Dam than the other dams (Figure 4.9). However, due to the high variation between sites (mean CV = 180 ± 62%, range = 103–316%) and overall large number of samples where no plastic was found (41% of samples), no significant difference between dams was found in either season or when seasons were combined (Kruskal-Wallis tests, all p-values >0.05). The hurdle model overpredicted the number of samples with no plastic by approximately 10%. It showed a significant main effect of dam, with a significantly higher mass of plastic predicted at the two Orange River dams than the Bloemhof Dam (Table 4.3, Figure 4.10). However, this was driven by the

strong interaction between dam and season and was most prominent in the wet season, with little difference between dams predicted in the dry season (Figure 4.10). The effect of distance to the dam wall was not significant (Table 4.3).

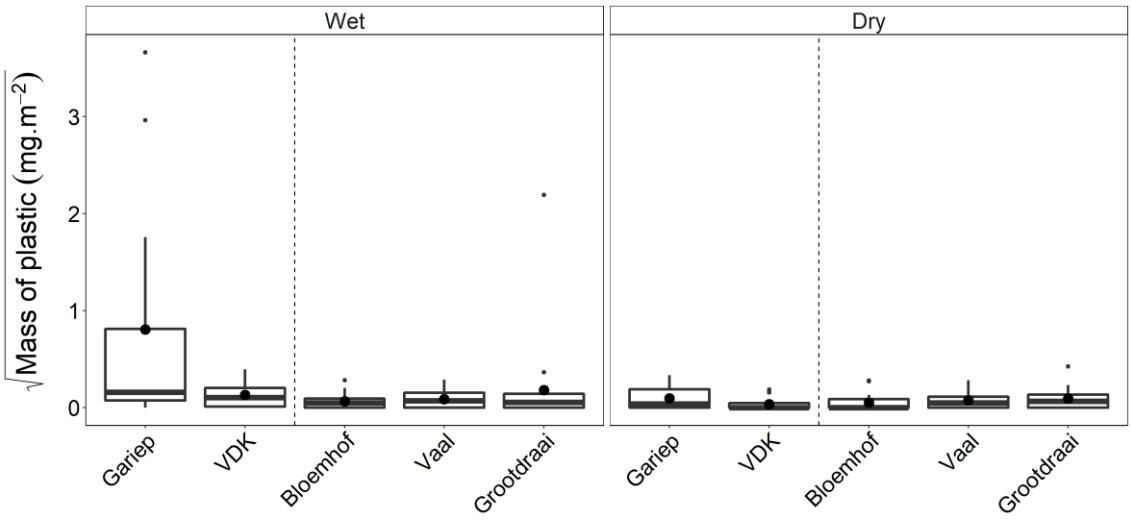


Figure 4.9: Boxplots of the total mass of plastic (mg·m^{-2}) found in samples collected at each dam in the wet (left panel) and dry (right panel) season (i.e. 10 L and neuston net samples combined). For clear visualisation, the data have been transformed using a square-root transformation. Conventions as Figures 4.3 and 4.8.

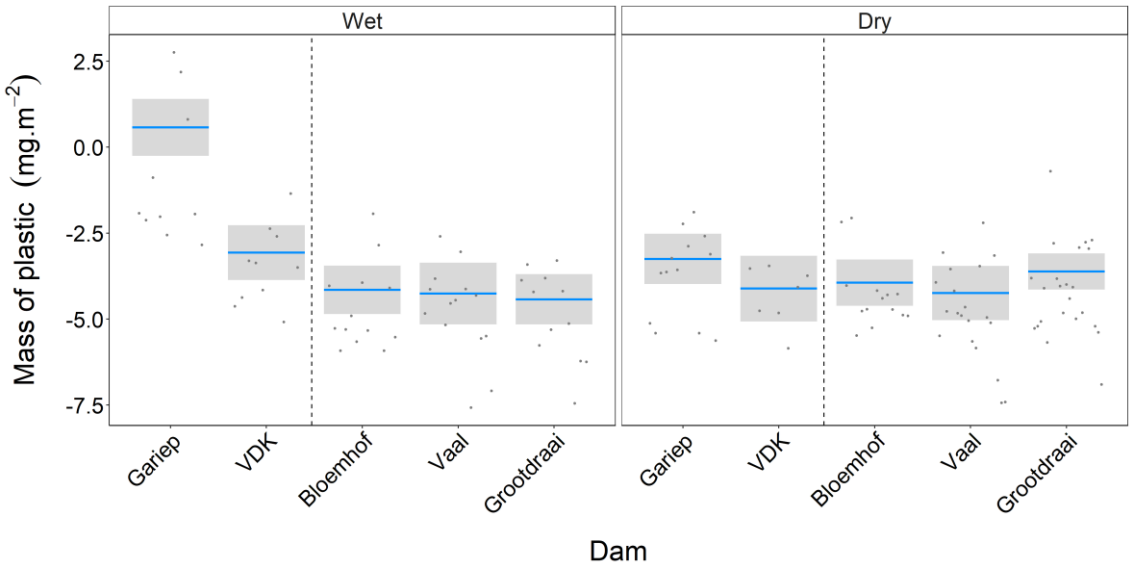


Figure 4.10: The mass of plastic and microfibres (mg·m^{-2}) as predicted by the second part of the hurdle model at the five different dams for the wet and dry seasons. The dashed vertical lines separate dams on the Orange River (Gariep and Van der Kloof [VDK]) from those on the Vaal River (Bloemhof, Vaal and Grootdraai).

Table 4.3: Hurdle model results showing the effect (± standard error) of season, dam, distance to dam wall and an interaction between season and dam on the total mass of plastic ($mg \cdot m^{-2}$) at sites where plastic was found (non-zero values) at five dams on the Orange and Vaal rivers. Estimates are presented on the log-scale.

	Estimate ± SE	**t-value**	**P**
Season (dry) [a]	0.21 ± 0.50	0.423	0.673
Distance	0.23 ± 0.17	1.387	0.168
Dam main effect [b]			
Vaal	-0.11 ± 0.60	-0.182	0.856
Grootdraai	-0.28 ± 0.52	-0.533	0.595
Gariep	4.73 ± 0.56	8.433	**<0.001**
VDK	1.08 ± 0.54	1.992	**<0.05**
Interaction effect [c]			
Vaal (dry)	-0.19 ± 0.67	-0.288	0.774
Grootdraai (dry)	0.60 ±0.67	0.890	0.375
Gariep (dry)	-4.04 ± 0.734	-5.498	**<0.001**
VDK (dry)	-1.26 ± 0.805	-1.560	0.121
Intercept	-4.03 ± 0.377	-10.674	**<0.001**

[a] Estimate indicates expected microplastic count in the dry season relative to the wet season

[b] Estimates indicate expected microplastic count at the given dam relative to the reference dam (Bloemhof Dam)

[c] Estimates indicate expected microplastic count at the given dam in the dry season relative to the same dam in the wet season

A higher mass of plastic and microfibres was found in samples collected by boat close to the Vaal Dam wall (mean = 0.25 ± 0.51 $mg \cdot m^{-2}$) than those collected 10 km away (mean = 0.03 ± 0.05 $mg \cdot m^{-2}$), but this difference was not significant (Mann-Whitney test: W = 17.5, p = 0.33). On the Gariep Dam, the mass of plastics and microfibres was highest in samples collected 5 km from the dam wall (mean = 0.21 ± 0.24 $mg \cdot m^{-2}$), but this was not significantly different to the other sites (Kruskal-Wallis test: $\chi^2 = 1.58$, p = 0.45).

Discussion

The annual flux of sediments into coastal seas has been reduced by approximately 1.4 billion tonnes·yr^{-1} due to the trapping of sediments in dam reservoirs (Syvitski et al. 2005). In the Orange River, the current annual flux of sediment into the Atlantic Ocean is estimated to be 25–100 million tonnes·yr^{-1} less than pre-anthropogenic times (Syvitski et al. 2005), suggesting that the five dams built within the Orange-Vaal River system likely retain large amounts of sediment. However, I found no conclusive evidence in this chapter to support Mani et al. (2015) or Zhang et al. (2015) who suggest that dams also trap floating plastics. While microplastics and microfibres were generally more abundant at sites above than below the dam walls, this difference was not significant and site location did not explain a significant proportion of the variation in concentrations of microplastics and microfibres. Furthermore, while boat samples on the Gariep Dam might suggest that microplastics accumulate behind the walls, there was no strong pattern in the distribution of microplastics within the dams and sites closer to the dam walls did not have higher levels of contamination in either bulk water or neuston net samples. Together, these results suggest that dams on the Orange-Vaal River system do not act as important sinks for floating microplastics or microfibers.

Plastic and microfibre pollution in rivers tends to be highest in densely populated and highly industrialised areas due to the prevalence of point sources such as wastewater treatment plants (WWTPs) and storm water carrying urban run-off (Eerkes-Medrano et al. 2015; Eriksen et al. 2013; McCormick et al. 2014; Yonkos et al. 2014). I therefore expected microplastics and microfibres to be most abundant in the densely populated Vaal and Grootdraai dams, particularly if microplastics and microfibres were being trapped by the dam walls. While microplastic and microfibre densities in bulk water samples were slightly higher in both these dams than the downstream Bloemhof Dam, there was little difference in the abundance of microplastics found in the neuston net or the total mass of microplastics and microfibres between the dams on the Vaal River. When compared to the rural Orange River, microplastic and microfibre contamination was higher in the Vaal and Grootdraai dams than both Orange River dams in the dry season, but the opposite was found in the wet season. Furthermore, while plastics were generally scarce in neuston net samples, they were one to two orders of magnitude more abundant at

two sites on the rural Gariep Dam than at any other site. This anomaly might be explained by converging wind and water currents which cause floating plastics to accumulate (Eriksen et al. 2013; Figure 4.1). Alternatively, both sites were less than 1 km from the town of Oviston. The high levels of contamination might be due to poor management of waste plastics or large amounts of litter left by recreational users of the area (Figure 4.1). Ultimately, the differences between the dams were small, suggesting that dams do little to trap floating microplastics or microfibres.

It is difficult to compare concentrations of microplastics and microfibres in our study system to other systems due to the lack of standardised sampling protocols. This is particularly true for microfibres because most other studies used only neuston or manta trawl nets, which tend to miss microfibres due to their coarse mesh size (Ryan et al. 2019b). What is evident from our results is that levels of microplastic and microfibre pollution relative to other dams or lakes is low. For example, Zhang et al. (2015) conducted nine trawls on the Three Gorges Dam in China (net mesh size = 112 µm, width of opening = 100 cm, sampling depth = 50 cm) and found 3.4–13.6 items·m^{-2}, two orders of magnitude more than what I detected in the dams on Orange-Vaal River system although the smaller mesh size used by Zhang et al. (2015) may explain some of this difference. Eriksen et al. (2013) conducted 21 manta net trawls (net mesh size = 330 µm, opening width = 61 cm, sampling depth = 16 cm) in the Laurentian Great Lakes system, USA, and found an average of 0.043 ± 0.116 items·m^{-2} (range = 0 – 0.450 items·m^{-2}). While this is similar to what I found, my data were strongly skewed by the high prevalence of plastics in neuston net samples collected at beaches 1 and 2 on the Gariep Dam. When these two sites are excluded, contamination in the dams on the Orange-Vaal River system is an order of magnitude lower than that found in the Great Lakes (Eriksen et al. 2013) and three orders of magnitude less than that in the Three Gorges Dam (Zhang et al. 2015).

The low concentrations of microplastics and microfibres in my samples and the high variation between samples made it difficult to detect spatial patterns within or between dams. For bulk water samples, despite doubling the number of replicates collected at each site in the dry season, the variation and standard error remained high and the data were still right-skewed. Future research aiming to quantify the effect of dams on riverine transport of microplastics and microfibres should increase

the number of replicates per site and should focus on more polluted systems where detecting such patterns should be easier.

The retention rate of microplastics and microfibres within a river or reservoir depends on the density, size and shape of the item. For example, items with much higher densities than water will be transported as bedload and are the most likely items to be trapped in a dam reservoir, although they could be carried out by sluices (Kondolf et al. 2014). On the other hand, smaller items and those with densities only slightly higher than water might remain suspended within the water column in the river (Nizzetto et al. 2016b), but then sink and accumulate in bottom sediments as the water velocity and turbulence decrease upon entering the dam (Hoellein et al. 2014; Kowalski et al. 2016; Figure 4.1). I only sampled floating microplastics and microfibres, but it is likely that microplastic debris and microfibres accumulate in the bottom sediments of dams and are thus prevented from traveling downstream (Hoellein et al. 2014; Liedermann et al. 2018). The collection of bottom sediment samples from the dams on the Orange-Vaal River system would improve our understanding of the extent to which dams trap microplastics and microfibres. It is also important to consider the size of the reservoir, the steepness of the catchment area and the amount of water flowing into the dam. For example, if the dam reservoir is small, the river catchment steep or there is a large amount of water flowing into the dam, then the turbulence and velocity of the water might remain high enough for particles to remain suspended within the water column and pass over the dam wall (Kondolf et al. 2014; Figure 4.1). Conversely, dams with larger reservoirs or those built within shallower catchments will likely have longer residence times and a higher trap efficiency (Kondolf et al. 2014).

Quantifying microplastic and microfibre loads in rivers can be difficult and time consuming given the size of many river basins. Models can therefore be useful tools to extrapolate estimated microplastic and microfibre loads based on a few samples across large spatial scales and to estimate the flux of waste plastic and microfibres into coastal seas via rivers (e.g. Lebreton et al. 2017; Schmidt et al. 2017; Siegfried et al. 2017). However, for these models to be accurate, a thorough understanding of how plastic and microfibre particles are transported and retained in a river is vital (Besseling et al. 2017). The results of this chapter suggest that dams may not act as local sinks of floating microplastics or microfibres. However, including the presence

of dams as a parameter significantly improved the fit of the models used by Lebreton et al. (2017). It is therefore likely that dams trap microplastics and microfibres, but this effect might be stronger in heavily polluted dams. Process and/or longitudinal studies comparing how the concentration of microplastics and microfibres within a dam changes over time in relation to water inflow and between different sized catchments and reservoirs are needed to improve our understanding of how dams trap microplastics and microfibres.

Chapter 5

Synthesis and conclusions

A boom on the Black River, Cape Town, installed to stop plastics flowing into the sea.

Photo: E. Weideman

Plastic production has increased dramatically over the last few decades with more than 300 million tonnes produced each year since 2015 (Geyer et al. 2017). Of all the plastic that has been produced, only 9% has been recycled, 12% has been incinerated and the remaining 79% has either been dumped into landfills or released into the natural environment (Geyer et al. 2017). Most of the plastic currently entering the marine environment is thought to originate on land (Jambeck et al. 2015), and much of the research over the last two decades has focused on quantifying land-based sources of waste plastics. However, the research to date has mainly been conducted in developed countries in the Northern Hemisphere with comparatively little field data available for Africa (Blettler et al. 2018). Due to its growing human population and increasing consumption of plastic goods, Africa is expected to become an ever more significant source of plastics into the environment (Jambeck et al. 2018). In order to implement effective mitigation strategies aimed at stopping the release of plastics into coastal seas, data on major land-based sources of plastics are required.

In this thesis I focused on quantifying two possible land-based sources of plastic pollution in South Africa. I first quantified the amounts and types of litter in urban run-off from the large coastal city of Cape Town and compared this to a similar study conducted in 1996 by Arnold & Ryan (1999). I then assessed whether the Orange-Vaal River system transports litter from the industrial heartland of South Africa to the Atlantic Ocean and investigated whether dams within this river system trap floating microfibres and microplastics. The main results of my research are as follows:

1. Urban run-off from Cape Town is a significant source of litter into Table Bay

Compared to 1996, litter loads decreased by ~50% in the residential (88% by mass) and industrial (36% by mass) catchments but increased by 355% (208% by mass) in the mixed commercial/residential area. Despite Arnold & Ryan (1999) engaging with local communities, stakeholders and government officials following the publication of their report, it appears that little has been done to prevent litter entering Cape Town's stormwater system and urban run-off is a significant source of litter into Table Bay.

When extrapolated across the area of Cape Town, I estimate that some ~200–600 tonnes of plastic litter are released from stormwater outlets each year.

2. The Orange-Vaal River system does not appear to be a major source of litter into the Atlantic Ocean

Larger plastics were most abundant in the densely populated, highly urbanised and industrialised Vaal River and largely absent at sites on the sparsely populated, more rural Orange River. While there appeared to be an accumulation of microfibres in the arid lower reaches of the Orange River during the period of low flow prior to the wet season flush, concentrations dropped significantly at the end of the dry season and remained low at the mouth of the river in the middle of the dry season. Based on the three sampling events at the river mouth, it is estimated that 2.5 (1.2–3.5) tonnes of microfibres wash out of the Orange River annually. However, only a single sweet wrapper and a single hard plastic piece were found floating at the Orange River mouth. While the Orange-Vaal River system may be a source of some microfibres into the Atlantic Ocean, it appears that larger items are retained near source in the Vaal River. Too few litter items were found at the Orange River mouth to allow accurate predictions of the annual flux of litter into the sea, but litter loads are clearly low which supports Lebreton et al. (2017) who predicted that only 0.09 (0.04–0.35) tonnes of macroplastic are released from the Orange River annually.

3. Dams do not appear to trap floating microfibres or microplastics

While microplastics and microfibres were more abundant in the densely populated Vaal and Grootdraai dams at the end of the dry season, the opposite pattern occurred in the more sparsely populated Bloemhof, Gariep and Van Der Kloof dams at the end of the wet season. When accounting for variation between sites, there was no significant difference in microplastic and microfibre concentrations at sites above versus below the dam walls nor between sites closer to the dam walls versus those further away. Together, these results suggest that dams are not important sinks for floating microplastics or microfibres, which supports the findings in Chapter

3 that microplastics and microfibres accumulate in the lower reaches of the Orange River.

How accurately do models predict the flux of litter from land to sea in South Africa?

Globally there is a large mismatch between the 5–12 million tonnes of plastic estimated to be released from land-based sources each year (Jambeck et al. 2015) and the ~250 000 tonnes found floating at sea (Eriksen et al. 2014). While some plastic sinks in the sea, this is unlikely to account for the orders of magnitude difference (Ryan 2020). Rather, global models – which are rarely based on empirical field data – likely over-estimate the amount of plastic being released from land-based sources. Indeed, the results of my thesis support this. In Chapter 2 I estimate that ~200–600 tonnes of litter are released as part of urban run-off from Cape Town annually, which is much less than the 23 000–63 000 tonnes predicted by Jambeck et al. (2015). This is further supported by beach clean-ups which suggest that <2 000 tonnes of litter wash out of Cape Town annually (Chitaka & Von Blottnitz 2019). If scaled according to population size and given that 40% of the coastal urban population and 25% of the total coastal population live in Cape Town, the total amount of litter in urban run-off from South African coastal cities would be some 3–4 times higher than the Cape Town value. This equates to only 1 000–8 000 tonnes·yr^{-1}, while Jambeck et al. (2015) that 90 000–250 000 tonnes are released from South African land-based sources annually.

Similarly, in Chapter 3 I show that very little macrolitter or larger microplastic pieces are found at the Orange River mouth, suggesting that litter loads are lower than the 0.09 (0.04–0.35) tonnes predicted to be released from the Orange River by Lebreton et al. (2017). The Orange River is, however, unusual in that most of its catchment is very sparsely populated. Rivers in more densely populated areas of South Africa, such as in the Eastern Cape and KwaZulu-Natal provinces, may release much more plastic waste into the sea. However, it is unlikely that this would account for the large discrepancy between the 90 000–250 000 tonnes predicted by Jambeck et al. (2015) and the 1 000–8 000 tonnes estimated to be released from coastal cities. It is therefore apparent that the global models used by Jambeck et al. (2015) overestimate the amount of plastic released from land-based sources by at

least an order of magnitude which helps explain the large discrepancy between the 5–12 million tonnes of plastic estimated to be released from land-based sources each year (Jambeck et al. 2015) and the ~250 000 tonnes found floating at sea (Eriksen et al. 2014).

Solutions

Irrespective of whether model estimates of plastic pollution from South Africa are inflated, there is still plenty of scope to reduce the amount of plastic and other litter in the country. The first step in implementing effective mitigation strategies to prevent the flow of litter from land to sea is to identify major land-based sources. I have shown that urban run-off via stormwater systems in Cape Town transports high litter loads into the natural environment, much of which is likely street litter. As discussed in Chapter 2, solutions to this problem include installing litter traps in catch pit openings or litter booms in rivers, as has been done in the Black River. However, litter traps can only go so far. A more effective solution would be to address the root cause of the problem and prevent litter entering the stormwater system in the first place. At a local scale, this can be achieved by incentivising people not to litter, for example the implementation of container deposit legislation (whereby people receive cash in return for used containers) has been shown to significantly reduce littering in Australia and the United States (Schuyler et al. 2018). Other solutions include placing more bins in problem areas as well as conducting campaigns aimed at educating people about the negative effects of litter on the natural environment (Marais et al. 2004; Armitage 2007). Street cleaning efforts could also be improved either by increasing the number of cleaners, increasing the number of times the streets are swept each day and ensuring that cleaners avoid sweeping litter into catch pit openings (Armitage 2007). On a broader scale, countries should aim to transition from a linear economy where plastic items are discarded after being used once, to a more circular economy where plastics are recycled and re-used multiple times (Jambeck et al. 2018).

Suggestions for future research

I have only focused on two possible land-based sources of litter in South Africa but there are a myriad more which need to be investigated. Besides Cape Town, there are another three large coastal cities where there are high densities of beach litter, namely Durban, East London and Port Elizabeth (De Villiers 2018; Ryan et al. 2018; Ryan 2020). While some of the macrolitter might be left by beachgoers or washed up from offshore sources, most appears to be transported from land either via rivers or stormwater systems. For example, recent floods in Durban washed large quantities of litter from stormwater systems onto local beaches (Somdyala 2019). Disentangling where the litter comes from, both in South Africa but also in other African countries, could be used to implement measures to stem the release of this litter into the environment.

As discussed in Chapter 3, it is important to identify point sources of litter in a river catchment. This requires sampling on a finer scale, for example litter sampling on river banks or in recreational areas would show whether dumping or littering is a major source of waste plastics (Kiessling et al. 2019). Another approach is to collect water samples for microplastics and microfibres or conduct observations for macroplastics immediately upstream and downstream of potential point sources such as wastewater treatment plants or settled areas (McCormick et al. 2016). This would be particularly helpful in the Vaal River where plastic densities were high.

It is also important to identify sinks of plastics. For example, Lebreton et al. (2017) suggested that dams might trap at least 65% of litter in rivers. While I did not find any conclusive evidence to suggest that dams in the Orange-Vaal River system trap floating microplastics and microfibres, the data were noisy which made it difficult to pick up large-scale patterns. Sampling on a finer spatial scale, both along the shorelines of the dams as well as by boat might improve our understanding of whether dams trap plastics within their reservoirs.

My thesis has focused on understanding how litter enters the marine environment. However, many communities depend on freshwater environments for resources such as food and drinking water. It is therefore important to understand how this litter effects these freshwater ecosystems before it is flushed into the sea.

For example, future research could improve our understanding of how ingested microplastics might affect freshwater fish and invertebrates (Weber et al. 2018). Large plastic items such as bags have been shown to smother marine benthic habitats (Green et al. 2015) and the same might occur in river and dam sediments. Furthermore, while entanglement in plastic debris has been well documented at-sea, little is known about entanglement of freshwater species (Blettler et al. 2018; Blettler & Wantzen 2019). The field of freshwater plastic pollution is a new and exciting field of research and there is still much research to be done, particularly in the African context.

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Appendices

Table S1: Site locations, number of wastewater treatment plants (WWTP) within 50km upstream and total human population within 50km radius of each site sampled on the Orange and Vaal rivers during the wet and dry seasons in 2018. Sites are grouped according to river section and the average number of WWTPs and human population per section are indicated in bold.

	Site Location	WWTP	Human population
Lower Orange		**0**	**4 623 ± 1 298**
Alexander Bay	28.5656°S 16.5042°E	0	3 567
Onseepkans	28.7372°S 19.3045°E	0	4 230
Vioolsdrift	28.7658°S 17.6260°E	0	6 072
Middle Orange		**1.2 ± 0.4**	**51 513 ± 43 397**
Groblaarshoop	28.8796°S 21.9864°E	1	11 128
Kanoneiland	28.6476°S 21.1014°E	2	101 071
N14 Kakamas	28.7559°S 20.6223°E	1	31 393
Prieska	29.6568°S 22.7458°E	1	18 376
Upington	28.4646°S 21.2426°E	1	95 599
Upper Orange		**2.3 ± 1.7**	**30 183 ± 20 265**
Aliwal North	30.6855°S 26.7051°E	4	61 456
Douglas Orange	29.1617°S 23.6946°E	0	15 006
Orania	29.7901°S 24.4099°E	3	22 448
R48/Orania	29.9107°S 24.6363°E	2	12 973
Senqu	29.2902°S 28.9898°E	0	39 035
Lower Vaal		**2.2 ± 1.3**	**161 986 ± 129 030**
Barkly West	28.5506°S 24.5331°E	4	279 809
Christiana	27.9161°S 25.1741°E	1	147 464
Douglas Vaal	29.0496°S 23.7689°E	1	17 155
Schmidtsdrift	28.7086°S 24.0755°E	3	58 929
Windsorton	8.3291°S 24.7189°E	2	306 570
Middle Vaal		**5.4 ± 1.9**	**2 943 307 ± 2 636 506**
Boundary Road	26.7706°S 27.6777°E	7	1 652 170
Garthdale	26.4213°S 28.0950°E	6	8 391 004
Golf Road	26.6646°S 27.9548°E	7	3 776 022

Heidelberg	26.5021°S 28.3712°E	6	4 403 021
Kammandodrift	27.4863°S 26.2246°E	3	180 808
Lapa on Vaal	26.6661°S 27.9829°E	7	3 854 413
Orkney	27.0160°S 26.6942°E	3	505 658
S713	26.9707°S 27.2100°E	3	329 650
Vereeniging	26.6825°S 27.9384°E	7	3 397 016
Upper Vaal		**3.0 ± 1.8**	**222 805 ± 53 663**
FR Standerton	26.8548°S 29.4809°E	4	303 129
FR Vaal Dam	26.9919°S 28.7298°E	1	162 581
Paul Kruger	26.7702°S 29.9230°E	2	179 286
R65 Vaal	26.5693°S 30.2127°E	3	215 115
Robbersdrift	27.0159°S 29.0247°E	2	207 675
Standerton	26.9604°S 29.2434°E	6	269 042

Table S2: The location of each sampling site on the five major dams in the Orange-Vaal River system and the average number of microplastics and microfibres found at each site in bulk water and neuston net samples in the wet and dry seasons.

		Bulk water ($N \cdot L^{-1}$)		Neuston net ($N \cdot m^{-2}$)	
Site	**Location**	**Wet**	**Dry**	**Wet**	**Dry**
Overall		**0.23 ± 0.26**	**0.21 ± 0.28**	**0.07 ± 0.21**	**0.01 ± 0.02**
Bloemhof					
B1	27.6782°S 25.6832°E	0.03 ± 0.06	0.17 ± 0.27	0	0
B2	27.6768°S 25.6191°E	0.10 ± 0.10	0.03 ± 0.08	0.01 ± 0.01	0
B3	27.7233°S 27.7233°E	0.20 ± 0.20	0.12 ± 0.10	0.004 ± 0.01	0
B4	27.6310°S 25.6718°E	0.17 ± 0.15	0.20 ± 0.17	0.02 ± 0.003	0
Below wall	27.6639°S 25.6154°E	0.10 ± 0.10	0.15 ± 0.12	-	-
Gariep					
B1	30.7463°S 25.7756°E	-	0.03 ± 0.05	0.87 ± 0.21	0.01 ± 0.01
B2	30.6920°S 25.7795°E	0.07 ± 0.12	0.03 ± 0.05	0.53 ± 0.03	0.07 ± 0.06
B3	30.5858°S 25.5176°E	0.73 ± 0.49	0.6 ± 0.26	0.03 ± 0.02	0.01 ± 0.003
Below wall	30.6241°S 25.5034°E	-	0.03 ± 0.05	-	-
Grootdraai					
B1	26.9466°S 29.3973°E	0.27 ± 0.31	0.30 ± 0.13	0	0.004 ± 0.01
B2	26.8889°S 29.3090°E	0.20 ±0.17	0.20 ± 0.40	0.01 ± 0.003	0.002 ± 0.003
B3	26.8954°S 29.3251°E	0.03 ±0.06	0.38 ± 0.16	0	0.002 ± 0.003
B4	26.9169°S 29.3245°E	0.10 ± 0.00	0.73 ± 0.48	0.002 ± 0.003	0
Below wall	26.9170°S 29.2929°E	0	0.3 ± 0.21	-	-
Vaal					
B1	27.0095°S 28.3591°E	0.03 ± 0.06	0.48 ± 0.40	0.01 ± 0.003	0.002 ± 0.003
B2	26.9427°S 28.3976°E	0.13 ± 0.12	0.03 ± 0.05	0.02 ± 0.01	0
B3	26.9938°S 28.2154°E	0.07 ± 0.06	0.1 ± 0.15	0.02 ± 0.003	0.002 ± 0.003
B4	26.8685°S 28.2833°E	0.23 ± 0.21	0.43 ± 0.28	0	0.002 ± 0.003
Below wall	26.8743°S 281163°E	0	0.12 ± 0.18	-	-
Van Der Kloof					
B1	30.0763°S 24.8651°E	0.3 ± 0.26	0	0.004 ± 0.01	0
B2	29.9792°S 247463E°	0.37 ± 0.38	-	0.004 ± 0.01	-
B3	29.9946°S 24.7358°E	-	0.13 ± 0.1	0.09 ± 0.05	0
B4	30.2210°S 24.8728°E	-	0.08 ± 0.13	0	0
Below wall	29.9910°S 24.7280°E	-	0.12 ± 0.16	-	-

Table S3: The abundance and colour of litter items found in neuston net samples collected at five dams on the Orange and Vaal rivers in April and November 2018.

Type and colour of litter	n	% of total
All items	**827**	
Hard pieces	**703**	**85**
Clear	272	39
White	243	35
Blue	102	15
Turquoise	25	4
Green	18	3
Pink	13	2
Yellow	6	<1
Red	6	<1
Black	4	<1
Brown	4	<1
Orange	4	<1
Mixed	2	<1
Grey	2	<1
Purple	2	<1
Flexible packaging	**76**	**9**
Clear	34	45
White	15	18
Blue	13	17
Mixed	6	8
Grey	4	5
Turquoise	3	4
Black	1	1
Polystyrene	**28**	**3**
White	22	79
Grey	2	7
Brown	2	7
Black	2	7
Fibres	**10**	**1**
White	4	40
Blue	3	30
Clear	1	10
Green	1	10
Orange	1	10
Twine	**4**	**<1**
Blue	2	50
Turquoise	1	25
Green	1	25
Foam	**3**	**<1**
Grey	3	100
Paper	**2**	**<1**
White	2	100
Industrial pellets	**1**	**<1**
Clear	1	1

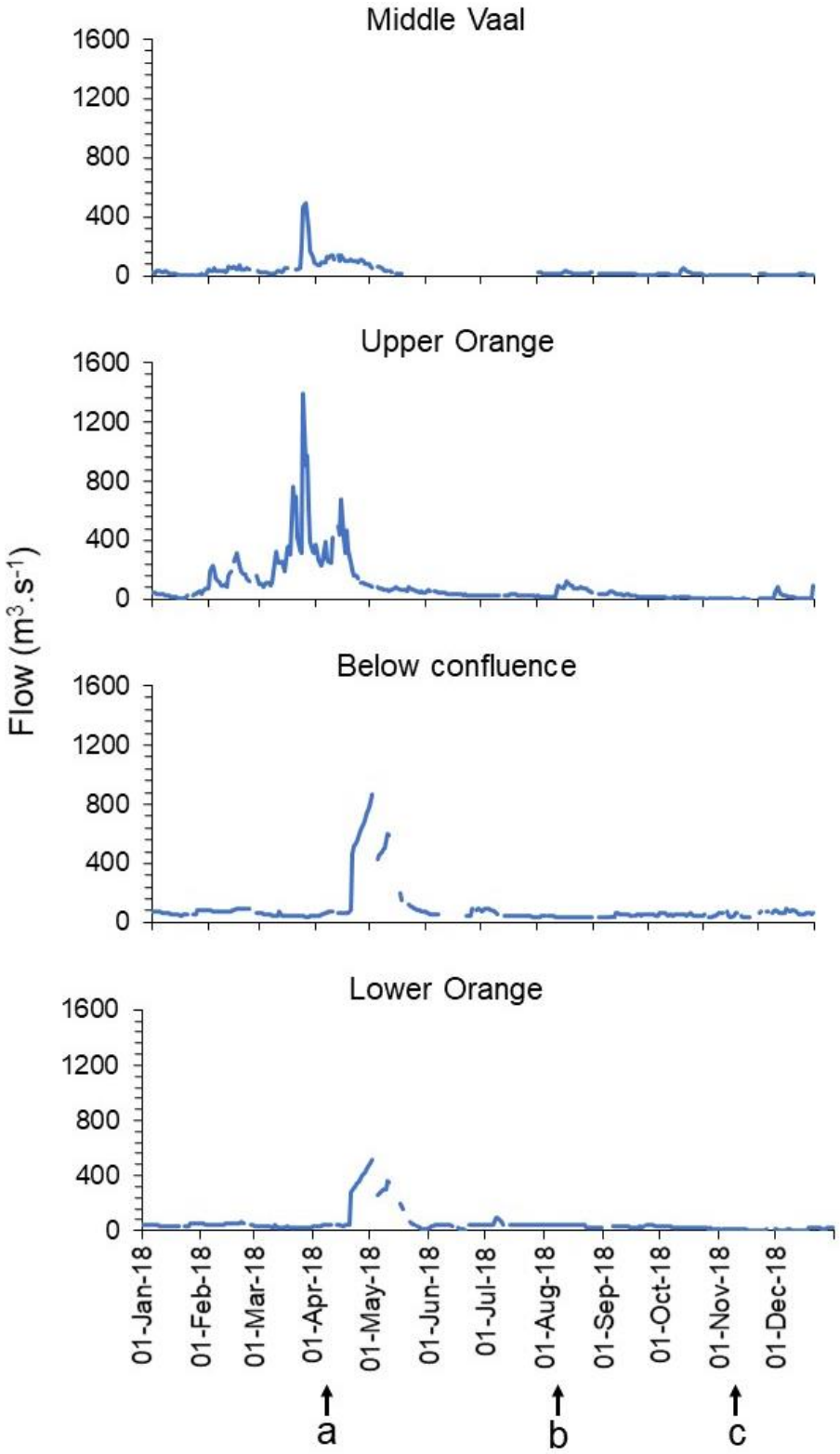

Figure S1: Flow rate ($m^3 \cdot s^{-1}$) in different sections of the Orange and Vaal rivers for 2018 when sampling was conducted. Sampling events at the Orange River mouth are indicated by letters where a = end of the wet season, b = middle of the dry season and c = end of the dry season. Flow rate data were obtained from the Daily State of the Dams website (DWS 2019a).