Nutrient characterisation of river inflow into the estuaries of the Gouritz Water Management Area, South Africa

DA Lemley[1]*, S Taljaard[2], JB Adams[1] and N Strydom[3]

[1]Department of Botany, Nelson Mandela Metropolitan University, PO Box 77000, Port Elizabeth, 6031, South Africa
[2]Council for Scientific and Industrial Research, PO Box 320, Stellenbosch, 7599, South Africa
[3]Department of Zoology, Nelson Mandela Metropolitan University, PO Box 77000, Port Elizabeth, 6031, South Africa

ABSTRACT

This study provides an overview of the nutrient status of river inflow into the estuaries within the Gouritz Water Management Area (WMA) of South Africa. Riverine inputs are a major source of macronutrients to estuaries and the adjacent coastal environments. Long-term water quality monitoring data (dissolved inorganic nitrogen, i.e. DIN; and dissolved inorganic phosphorus, i.e. DIP), collected by the Department of Water Affairs (DWA), were used to assess historical trends of river nutrient inflow within the Gouritz WMA. The results indicate that DIP concentrations exceeded the eutrophic limits for aquatic ecosystems (DWA) in 50% of the catchments assessed. Anthropogenic activities such as agriculture, wastewater discharge, urbanisation, and afforestation were significant factors influencing nutrient levels within these rivers. For the majority of the river systems (approx. 80%) there was no significant correlation ($P > 0.05$) between inorganic nutrient levels and freshwater inflow from the catchments. Wastewater treatment plant (WWTP) data (DWA) were assessed to explore the reasons for this 'disconnect' between freshwater inflow and inorganic nutrient levels. Results indicate that the Gwaing (267.73 kg·d^{-1} DIN; 77.46 kg·d^{-1} DIP), Goukou (49.71 kg·d^{-1} DIN; 17.38 kg·d^{-1} DIP), Knysna (41.77 kg·d^{-1} DIN; 13.92 kg·d^{-1} DIP) and Hartenbos (37.73 kg·d^{-1} DIN; 21.39 kg·d^{-1} DIP) systems received the highest daily loads from WWTPs. The Gwaing and Hartenbos estuaries would be most vulnerable to increased nutrient loading because of their small size and prolonged periods of mouth closure. The study highlights the importance of water quality monitoring of river inflows into coastal ecosystems, as it is needed to assess pollution trends and identify management priorities.

Keywords: Water quality, eutrophication, inorganic nutrients, wastewater discharges

INTRODUCTION

Estuaries form the interface between the marine and freshwater environments and as a result are complex, dynamic and productive ecosystems. Estuaries provide numerous ecosystem services, such as regulating (erosion control), provisioning (food and water), supporting (nursery areas) and cultural services (recreation and tourism) (Costanza et al., 1997; Van Niekerk and Turpie, 2012). Consequently, they are one of the most heavily used and threatened ecosystems worldwide, due to their high socio-economic importance. Water quality and ecological functioning of estuaries closely reflect human activity, not only along the estuarine sector itself, but also within its entire upstream catchment (Billen et al., 2001).

Over recent history, land-use changes in drainage basins/catchments have increased significantly and have led to a considerable transformation from a natural to heavily developed landscape (Fohrer and Chicharo, 2012). This transformation is largely due to the rapid development and intensification of agricultural activities, afforestation, urbanisation, water abstraction, and industrial activities that have had a marked impact on the delicate balance between riverine and coastal ecosystems (Cloern, 2001; Fohrer and Chicharo, 2012). Similar trends are observed in South Africa where all estuaries are subjected to varying degrees and combinations of anthropogenic pressures, whether it is flow alteration, pollution, habitat loss, mining or the exploitation of living resources (Van Niekerk and Turpie, 2012). As a result, the National Biodiversity Assessment of 2011 showed that 42% of estuaries in South Africa are classified as being in fair to poor condition (Van Niekerk and Turpie, 2012).

Coastal ecosystems receive cumulative impacts from their catchments (Fohrer and Chicharo, 2012). This is an important issue to address because land cover changes are expected to have negative effects on the productivity, biodiversity, and ecological functioning of coastal ecosystems (Holland et al., 2004). Anthropogenic manipulation of freshwater affects the physical and biogeochemical balance of estuaries by altering the input, transport, and assimilation of water (i.e. quantity and quality), inorganic nutrients (i.e. N, P and Si), particulate and dissolved organic matter, toxic metals and organopollutants (Buzzelli et al., 2007). The specific responses of different estuarine types to these impacts vary with regards to the composition of the inputs and gradients in geomorphology, physical transport, and internal physical characteristics and biogeochemical cycling (Buzzelli et al., 2007; Buzzelli, 2012).

A great deal of emphasis has been placed on determining the consequences of these anthropogenic stressors on estuaries through the use of monitoring programmes (Carstensen et al., 2012). In South Africa, the Department of Water Affairs (DWA) has co-ordinated an extensive National Chemical Monitoring Programme (NCMP) since the early 1970s, which comprises in excess of 2000 monitoring stations situated in rivers, dams and lakes throughout the country (Huizenga, 2011; Huizenga et al., 2013). Chemical variables that are monitored include: inorganic nutrients (N, P and Si), pH, electrical conductivity, total dissolved solids, as well as a variety of other major chemical

* To whom all correspondence should be addressed.
☎ +27 72 063 1440; e-mail: Daniel.Lemley@nmmu.ac.za
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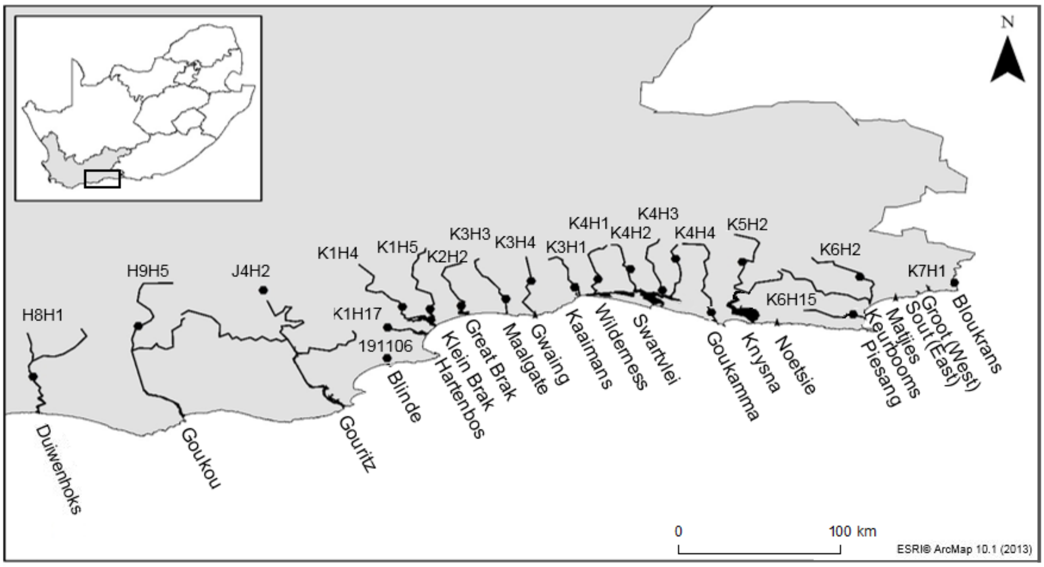

Figure 1
*Geographical distribution of the Gouritz WMA estuaries occurring along the southern coast of South Africa, and the position of the river monitoring stations assessed (*indicated by black 'dots')*

elements and compounds (i.e. sodium, calcium, potassium, magnesium, chloride, fluoride, and sulphate) (Huizenga, 2011; Huizenga et al., 2013). Despite its obvious benefits, there are numerous shortcomings associated with the NCMP, including geographical bias in site selection, tendency for intermittent datasets, and the 'inter-station' variability regarding dataset sizes (Huizenga, 2011).

In stark contrast to the NCMP, there is a lack of comprehensive long-term estuarine monitoring programmes in South Africa. The primary reasons for the slow development and implementation of such programmes have been attributed to lack of expertise, high costs, time-consuming nature of such undertakings, and a lack of integration between responsible authorities and programmes (Taljaard et al., 2003). Estuarine monitoring is largely project-specific and discontinuous, therefore providing little in the way of guiding management decisions (Taljaard et al., 2003). At present, a National Estuary Monitoring Programme is being developed for South Africa; however, only a small percentage of the country's estuaries are currently monitored on a regular basis.

This study used the DWA long-term monitoring data to determine the nutrient inputs to estuaries from rivers, using the Gouritz Water Management Area (WMA) as a case study. Seasonal profiles, nutrient fluxes, temporal trends and wastewater discharge loads were investigated to identify primary nutrient sources entering estuaries from their catchments. This is important, as knowledge about certain water quality variables can be used to predict the future structure and functioning of estuaries under various scenarios, thus assisting decision-making processes and estuary management.

MATERIALS AND METHODS

Study area

Most of the Gouritz WMA (Fig. 1) is located within the Western Cape Province of South Africa, with smaller portions in the Eastern Cape and Northern Cape Provinces (DEADP, 2011). This WMA has a total catchment area of 53 139 km^2, with its largest river (i.e. Gouritz River) accounting for approximately 41% of the total surface flow in the area (River Health Programme, 2007; DEADP, 2011). There are no inter-basin water transfers into the Gouritz WMA and approximately 70% of the available water is surface water (DEADP, 2011).

Economically important land-use activities are agricultural practices (e.g. ostriches, cattle and timber), commercial fisheries (e.g. fish and shellfish), the petrochemical industry and ecotourism (River Health Programme, 2007; DEADP, 2011). Land-use activities in the Gouritz WMA that are pertinent to the health and sustainability of aquatic ecosystems include irrigation, wastewater treatment plant (WWTP) discharges, diffuse pollution (e.g. agriculture and urbanisation), afforestation and deforestation (Table 1). These activities can have significant effects on water quality and quantity entering aquatic ecosystems.

The Gouritz WMA has 21 estuaries which have a high conservation status (River Health Programme, 2007). All estuarine types as described by Whitfield (1992) are represented in this area (Table 1). These include permanently open estuaries (POE), temporarily open/closed estuaries (TOCE), estuarine lakes, estuarine bays and river mouths. Most of the estuaries (approx. 66%) have a high Present Ecological Status (PES) and are unmodified (A) or largely natural (B) systems (Van Niekerk and Turpie, 2012). Half of the estuaries (approx. 48%) have Recommended Ecological Categories (REC) higher than their PES, which demonstrates their protected area status and importance (Van Niekerk and Turpie, 2012). Therefore, due to the socio-economic benefits they provide, it is important to monitor and manage these systems in order to ensure their continued health and sustainability. Table 1 indicates those estuaries where DWA has water quality monitoring data for the rivers that flow into the estuaries.

Assessment of historical water quality data

This study followed a similar approach to that described by De Villiers and Thiart (2007). Long-term water quality monitoring data for the river systems and WWTP discharges in the Gouritz WMA were obtained from the Department of Water Affairs (DWA) (DWA, 2014a). The water quality monitoring parameters used in this study include the dissolved inorganic nitrogen (DIN) species (NH_4^+, NO_2^- and NO_3^-) and dissolved inorganic phosphorus (DIP) (PO_4^{3-}), measured in µg·ℓ$^{-1}$. Nutrient analyses were conducted by the DWA laboratories using the photometric

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TABLE 1 Available information regarding the type, Present Ecological Status (PES), Recommended Ecological Category (REC) and significant pressures (i.e. classified as medium or high) associated with the estuaries in the Gouritz WMA (A – unmodified, B – largely natural, few modifications, C – moderately modified, and D – largely modified) (from Van Niekerk and Turpie, 2012)					
System	**Type**	**PES**	**REC**	**DWA Monitoring (Yes/No)**	**Key pressures**
Duiwenhoks	POE	B	A	Yes	Flow; fishing
Goukou	POE	C	B	Yes	Flow; pollution; fishing; habitat loss
Gouritz	POE	C	B	Yes	Flow; pollution; fishing; habitat loss
Blinde	TOCE	B	B	Yes	Pollution
Hartenbos	TOCE	D	C	Yes	Flow; pollution (WWTP); artificial breaching; habitat loss
Klein Brak	TOCE	C	C	Yes	Pollution; fishing; habitat loss
Great Brak	TOCE	D	C	Yes	Flow; pollution; fishing; artificial breaching; habitat loss
Maalgate	TOCE	B	B	Yes	-
Gwaing	TOCE	C	C	Yes	Pollution (WWTP)
Kaaimans	POE	B	B	Yes	-
Wilderness	Estuarine lake	B	A	Yes	Pollution; habitat loss
Swartvlei	Estuarine lake	B	A	Yes	Habitat loss
Goukamma	TOCE	A/B	A	Yes	-
Knysna	Estuarine bay	B	B	Yes	Pollution (WWTP)
Noetsie	TOCE	B	A	No	-
Piesang	TOCE	C	B	Yes	Pollution (industrial); habitat loss
Keurbooms	POE	A	A	Yes	-
Matjies	TOCE	B	B	No	-
Sout (East)	POE	A	A	No	-
Groot (West)	TOCE	B	A	No	Flow
Bloukrans	River mouth	A	A	Yes	-

methods as described in the Resource Quality Services methods manual (DWA, 2009). Data for total dissolved nitrogen (TN) and phosphorus (TP) are available for some of the monitoring stations (i.e. Gouritz, Hartenbos and Goukamma). However, these were not assessed as the data were intermittent (within and between systems), and provided little use with regards to assessing long-term temporal trends and inter-system comparisons. The sampling frequency varied within and among the water quality monitoring sites, from almost weekly to monthly. Furthermore, the duration of monitoring was highly variable between the different sites, ranging from 2 to 47 years of data (Table A1, Appendix).

Water quality monitoring sites situated as far downstream as possible in each catchment area were selected to estimate the quantity and quality of water entering estuaries from their respective catchments (Fig. 1). The distance of each monitoring station from the headwater reaches of the estuary was determined using the upstream estuarine boundary (5 m contour line) of each system (Table A1, Appendix). Flow data from the selected flow monitoring gauges were obtained from the DWA (DWA, 2014b). These data were utilised in the determination of annual nutrient fluxes and seasonal profiles for each catchment. Mean annual runoff (MAR), together with catchment area and median nutrient concentrations (for the entire time series), were used to calculate the annual nutrient fluxes for each catchment (Table 2). For the construction of seasonal profiles, seasonal flow patterns were attained using monthly median flows for each catchment, and subsequently overlaid onto the monthly median nutrient concentrations. Averaged annual nutrient data (i.e. DIN and DIP) were assessed to determine long-term temporal trends over the period of monitoring. A total of 20 coincident water quality and flow monitoring stations, with drainage areas ranging from 22 to more than 43 000 km^2 (Table 2), were evaluated and statistically analysed for this study.

WWTPs that are situated along and currently discharging into systems in the Gouritz WMA were assessed to quantify their nutrient inputs to these systems. The operational daily flow was calculated using information on the design capacity ($M\ell \cdot d^{-1}$) and operational efficiency (%) for each WWTP obtained from the Green Drop Report (DWA, 2012). Using these operational daily flows and the median DIN and DIP concentrations ($mg \cdot \ell^{-1}$) for the entire time series, the daily loads ($kg \cdot d^{-1}$) were then calculated. Lastly, the compliance of each WWTP was assessed using the general chemical compliance limits for DIN (21 $mg \cdot \ell^{-1}$) and DIP (10 $mg \cdot \ell^{-1}$) set out in Sections 21 (f) and (h) of the General Authorisations (DWAF, 2004) which aim to control the 'discharge of waste or water containing waste into a water resource through a pipe, canal, sewer or other conduit'.

For systems with flow gauge stations present, the daily inorganic nutrient load (i.e. DIN and DIP) to the estuaries was calculated using median nutrient concentrations and MAR data obtained from the monitoring stations. Furthermore, in order to determine an estimate of the total daily inorganic nutrient load to the estuaries, the aforementioned WWTP daily loads were, where applicable, included. Therefore, in systems where the WWTP is situated between the river monitoring station and the estuary, or alternatively discharging directly into the estuary, the two loads (calculated using WWTP median nutrient values and operational daily flow data) were combined. It is, however, important to note that for systems in which the

TABLE 2
Median, minimum and maximum values (of all monthly values in time series over the sampling period) for DIN and DIP in river inflow, and annual inorganic fluxes of N and P

Catchment name	Station No.	Catchment area (km^2)	MAR ($x\ 10^6\ m^3\ yr^{-1}$)	DIN ($\mu g \cdot \ell^{-1}$) median [min; max]	DIP ($\mu g \cdot \ell^{-1}$) median [min; max]	Annual flux ($kg \cdot km^{-2} \cdot yr^{-1}$ N)	Annual flux ($kg \cdot km^{-2} \cdot yr^{-1}$ P)
Duiwenhoks	H8H1	790	85.62	158 [20; 1 110]	25 [3; 3 636]	17.12	2.71
Goukou	H9H5	228	46.71	66 [20; 1 071]	25 [5; 2 443]	13.52	5.12
Gouritz	J4H2	43 451	475.47	76.5 [20; 7 000]	28 [3; 621]	0.84	0.31
Blinde	191106	-	-	300 [300; 1 050]	25 [25; 100]	-	-
Hartenbos	K1H17	101	2.41	91 [25; 1 170]	17 [3; 1 651]	2.17	0.41
Klein Brak (Brandwag)	K1H4	215	10.03	90 [25; 2 583]	12 [3; 175]	4.2	0.56
Klein Brak (Moordkuil)	K1H5	198	16.79	80 [25; 5 095]	17 [3; 573]	6.78	1.44
Great Brak	K2H2	131	17.07	100 [20; 6 270]	25 [3; 484]	13.03	3.26
Maalgate	K3H3	145	25.92	125 [20; 4 942]	32 [3; 1 359]	22.35	5.72
Gwaing	K3H4	34	17.45	280 [20; 3 563]	29 [3; 783]	143.68	14.88
Kaaimans	K3H1	47	12.24	82 [20; 3 535]	24 [3; 536]	21.36	6.25
Wilderness	K3H5	78	13.42	68.5 [20; 1 087]	16 [3; 432]	11.79	2.75
Swartvlei (Hoekraal)	K4H1	111	24.17	80 [20; 674]	22 [3; 463]	17.42	4.79
Swartvlei (Karatara)	K4H2	22	9.65	90 [20; 1 071]	32 [3; 305]	39.47	14.03
Swartvlei (Diep)	K4H3	72	9.36	84 [20; 4 027]	12 [3; 927]	10.92	1.56
Goukamma	K4H4	-	-	69 [40; 139]	20 [8; 33]	-	-
Knysna	K5H2	133	24.72	83 [20; 3 232]	18 [3; 349]	15.43	3.35
Piesang	K6H15	-	-	168.5 [40; 526]	34.5 [10; 385]	-	-
Keurbooms	K6H2	764	84.35	63 [20; 730]	18 [3; 162]	6.96	1.99
Bloukrans	K7H1	57	25.92	63 [20; 1 778]	25 [3; 423]	28.65	11.37

WWTP is situated upstream of the river monitoring station, it was assumed that the nutrient and flow inputs from the WWTP were already accounted for in the river data, and therefore excluded.

Data analysis

Data were analysed using Statistica Version 10 (StatSoft Inc., 2010). The data were tested for normality using the Shapiro-Wilks test. Relationships between inorganic nutrient concentrations (DIN and DIP) and freshwater inflow, for each system, were analysed using either the parametric correlation coefficient or the non-parametric Spearman's rank correlation, depending on the normality of the data. A Kruskal-Wallis non-parametric analysis of variance was used for an inter-system comparison of overall DIN and DIP concentrations (using monthly median values). Lastly, long-term temporal trends in nutrient concentrations were statistically analysed for each system using linear regressions. All analyses were done at $\alpha = 0.05$.

RESULTS

Nutrient assessment of river monitoring stations

Median DIN and DIP values, together with the minimum and maximum values in the time series, for river inflow are listed in Table 2. The median DIN concentrations for the Gouritz WMA ranged from 63 to 300 $\mu g \cdot \ell^{-1}$; whilst the median DIP concentrations ranged from 12 to 34.5 $\mu g \cdot \ell^{-1}$. When assessing the maximum DIN and DIP concentrations over the entire time series, the Gouritz River showed the highest DIN value of 7 000 $\mu g \cdot \ell^{-1}$, with the Duiwenhoks River exhibiting the highest DIP value of 3 636 $\mu g \cdot \ell^{-1}$. An important aspect to note is the duration of sampling (Table A1, Appendix), as this provides a measure of reliability with regards to the observations made in Table 2. For example, the Blinde and Goukamma rivers provide little value in the way of historical assessments, due to the fact that they have only been monitored for a 2-year period.

The annual inorganic nutrient fluxes (i.e. N and P) for the various catchments assessed ranged from 0.84 to 143.68 $kg \cdot km^{-2} \cdot yr^{-1}$ N and 0.31 to 14.88 $kg \cdot km^{-2} \cdot yr^{-1}$ P. (Table 2). It is important to note that catchment area and MAR influence the determination of these fluxes, and therefore should be considered when making inter-system comparisons. For example, the similarity in catchment size and MAR (approx. 85 x $10^6\ m^3\ yr^{-1}$) of the permanently open Duiwenhoks (790 km^2) and Keurbooms (764 km^2) estuaries allows for a direct inter-system comparison of nutrient fluxes (Table 2). From this it can be seen that Duiwenhoks (17.12 $kg \cdot km^{-2} \cdot yr^{-1}$ N) receives a far greater annual N flux compared to Keurbooms (6.96 $kg \cdot km^{-2} \cdot yr^{-1}$ N). For the temporarily open/closed estuaries (TOCE), the annual N and P fluxes, with the exception of Gwaing (143.68 $kg \cdot km^{-2}\ yr^{-1}$ N and 14.88 $kg \cdot km^{-2}\ yr^{-1}$ P), which had greatly elevated values, were similar.

Daily nutrient loads to estuaries

Median DIN and DIP values, together with the minimum and maximum values in the time series, for the various WWTP discharges are listed in Table 3. Using median DIN concentrations, it was found that the chemical compliance limits for the discharge of wastewater into water resources (> 21 $mg \cdot \ell^{-1}$) were exceeded in three of the WWTPs assessed, namely the Gwaing (33.53 $mg \cdot \ell^{-1}$), Klein Brak (22.40 $mg \cdot \ell^{-1}$) and Goukou (Riversdale) (22.30 $mg \cdot \ell^{-1}$) plants. Only one of the WWTPs, namely Duiwenhoks (11.70 $mg \cdot \ell^{-1}$), had median DIP concentrations exceeding the compliance limit (> 10 $mg \cdot \ell^{-1}$). All of the WWTPs episodically exceed the chemical compliance limits if

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TABLE 3 Median, minimum and maximum values (of all monthly values in time series over the sampling period) for DIN and DIP for WWTPs, and daily inorganic loads of N and P									
System	Station No.	Station co-ordinates	Position of WWTP in relation to river station	Monitoring Period	DIN (mg·ℓ⁻¹) Median [min; max]	DIP (mg·ℓ⁻¹) Median [min; max]	Operational daily flow (Mℓ·d⁻¹)	Daily load (kg·d⁻¹ DIN)	Daily load (kg·d⁻¹ DIP)
Duiwenhoks	1-10253	34.105°S 20.9725°E	Above	2004 – 2013	12.03 [0.15; 93.6]	11.7 [0.41; 14.7]	0.37	4.44	4.32
Goukou (River)	1-10254	34.115°S 21.2823°E	Below	2004 – 2013	22.3 [0.15; 70.7]	7.15 [0.41; 210]	2.2	49.05	15.73
Goukou (Mouth)	188634	34.3872°S 21.416°E	Below (within estuary boundaries)	2005 – 2012	0.72 [0.15; 68.7]	1.8 [0.09; 69]	0.92	0.66	1.65
Hartenbos	181038	34.11389°S 22.100°E	Below (within estuary boundaries)	2004 – 2013	4.7 [0.15; 58.3]	2.67 [0.22; 18.8]	8.03	37.73	21.39
Klein Brak (Moordkuil)	1-10276	33.9576°S 22.1435°E	Above	2004 – 2012	22.4 [0.15; 116]	4.5 [0.3; 19.2]	0.09	3.25	0.86
Gwaing	1-10281	33.9987°S 22.4274°E	Below	2004 – 2013	33.53 [0.15; 77.7]	9.7 [0.23; 96.3]	7.99	267.73	77.46
Swartvlei (Diep)	188622	33.8997°S 22.6769°E	Above	2003 – 2012	10.6 [0.6; 45.3]	9.5 [1.2; 13.2]	-	-	-
Knysna	181488	34.04444°S 23.075°E	Below (within estuary boundaries)	2004 – 2013	7.2 [0.05; 60.9]	2.4 [0.03; 135]	5.8	41.77	13.92
Piesang	188633	34.0449°S 23.2307°E	Above	2005 – 2012	15.15 [1; 59.4]	9.2 [3.45; 12.5]	-	-	-

TABLE 4 Summary of the total daily flow and inorganic nutrient loads (i.e. river and WWTP inputs) entering the estuaries of the Gouritz WMA (Note: the median nutrient concentrations for Klein Brak and Swartvlei represent the median value from the combined seasonal profiles)							
System	Daily flow volume into estuary (Mℓ·d⁻¹)	Median [DIN] entering estuary (µg·ℓ⁻¹)	Median [DIP] entering estuary (µg·ℓ⁻¹)	Total daily load to estuary (kg·d⁻¹ DIN)	Total daily load to estuary (kg·d⁻¹ DIP)	Contribution of WWTP inputs to total daily nutrient loads (%)	
						DIN	DIP
Duiwenhoks	234.57	158	25	37.06	5.86	11.98	73.72
Goukou	131.09	443.63	156.98	58.16	20.58	85.47	84.45
Gouritz	1 302.67	77	28	100.31	36.47	-	-
Hartenbos	14.64	2 618.46	1 469.1	38.33	21.51	98.43	99.44
Klein Brak	73.46	84.53	14.48	6.21	1.06	52.33	81.13
Great Brak	46.76	100	25	4.68	1.17	-	-
Maalgate	71.02	125	32	8.88	2.27	-	-
Gwaing	55.79	5 039.1	1 413.43	281.11	78.85	95.24	98.24
Kaaimans	33.55	82	24	2.75	0.81	-	-
Wilderness	36.78	68.5	16	2.52	0.59	-	-
Swartvlei	118.31	81.62	23.56	9.66	2.79	-	-
Knysna	73.54	644.5	205.93	47.40	15.14	88.12	91.94
Keurbooms	231.09	63	18	14.56	4.16	-	-
Bloukrans	71.01	63	25	4.47	1.78	-	-

the maximum nutrient concentrations are considered.

The daily nutrient loads for the Piesang and Swartvlei WWTPs could not be determined due to a lack of available information on design capacity and operational efficiency. For each of the other WWTPs this ranged from 0.66 to 267.73 kg·d^{-1} DIN and 0.86 to 77.46 kg·d^{-1} DIP (Table 3). More specifically, the results indicated that the Gwaing (267.73 kg·d^{-1} DIN and 77.46 kg·d^{-1} DIP), Goukou (Riversdale) (49.05 kg· d^{-1} DIN and 15.73 kg·d^{-1} DIP), Knysna (41.77 kg DIN d^{-1} and 13.92 kg· DIP d^{-1}) and Hartenbos (37.73 kg·d^{-1} DIN and 21.39 kg·d^{-1} DIP) systems received the highest daily loads from their respective WWTPs.

An assessment of the total daily inorganic nutrient loads (Table 4) reaching the estuaries within the Gouritz WMA, showed that the systems not subjected to WWTP inputs generally experienced loads of less than 10 kg·d^{-1} DIN and 3 kg·d^{-1} DIP, respectively. The only exceptions to these observations were found in the Gouritz (100.31 kg·d^{-1} DIN; 36.47 kg·d^{-1} DIP) and Keurbooms (14.56 kg·d^{-1} DIN; 4.16 kg·d^{-1} DIP) systems; however this can be explained by the high daily freshwater inflow volumes they receive (Table 4). Alternatively, estuaries on the receiving end of WWTP inputs, either directly into the estuary or indirectly via the upstream river, had elevated total daily nutrient loads ranging from 6.21 to 281.11 kg·d^{-1} DIN and

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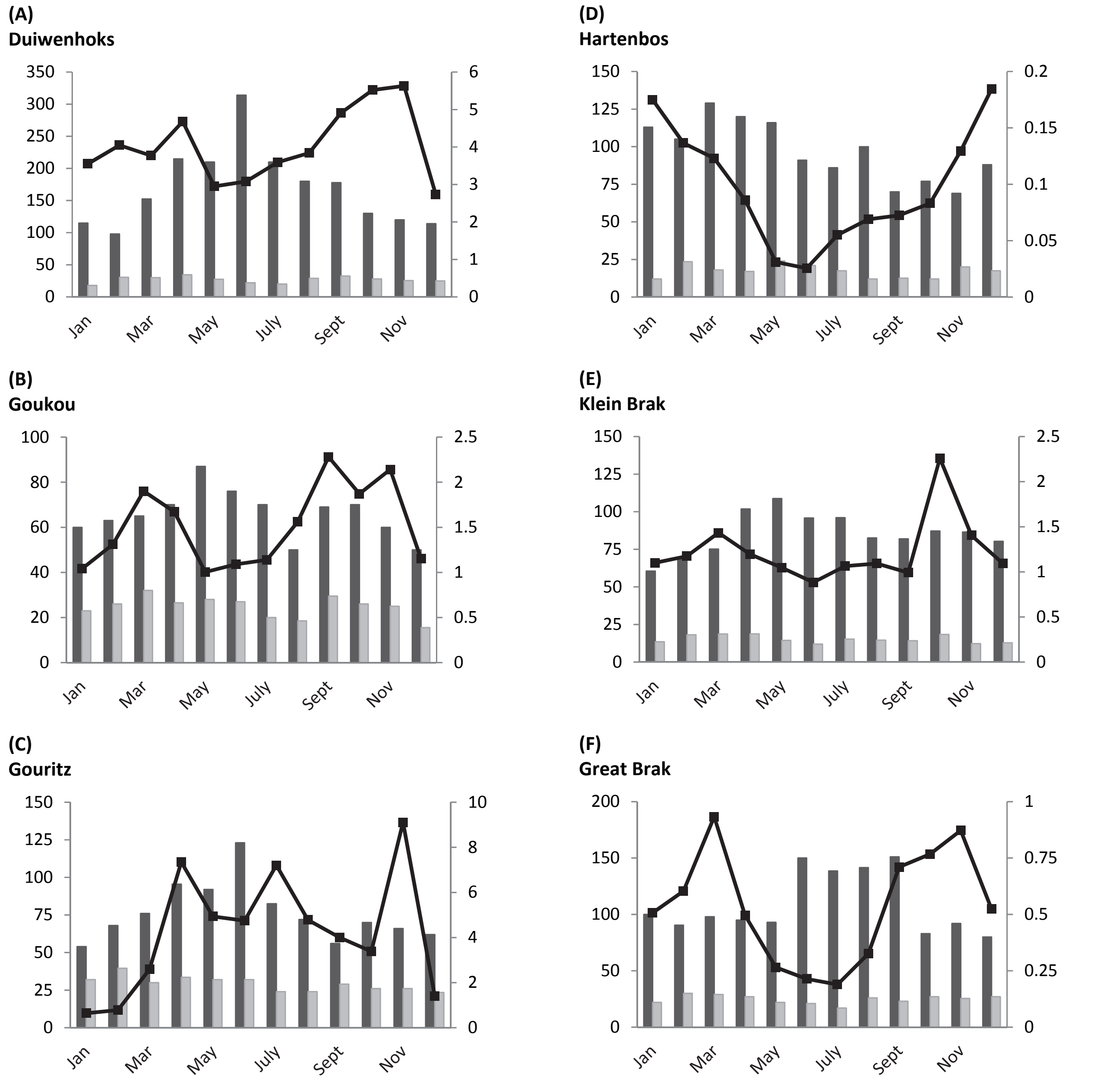

Figure 2 (above and right)

Seasonal profiles (from monthly median values over the entire monitoring period) of DIN (dark grey bars, left-hand y-axis in µg·ℓ⁻¹ N), DIP (light grey bars, left-hand y-axis in µg·ℓ⁻¹ P), and river flow volume (line, right-hand y-axis in 10^6 m³) for river monitoring stations

1.06 to 78.85 kg·d^{-1} DIP (Table 4). Furthermore, WWTP inputs contributed greatly to the total daily DIN (~11.98 to 98.43%) and DIP (~73.72 to 99.44%) loads in these estuaries (Table 4). It is important to reiterate that these total daily inorganic nutrient load values are simply being used as proxies with which to identify vulnerable systems and allow for inter-system comparisons.

Seasonal trends

Seasonal profiles of selected catchments in the Gouritz WMA are shown in Fig. 2. For ease of comparison, the seasonal profiles of the Klein Brak (Fig. 2E) and Swartvlei (Fig. 2K) systems illustrate the combined median inorganic nutrient concentrations and flow volume of each of their respective river sources. The combined seasonal profiles were calculated using the following equations:

$$V_{\text{Total}} = V_1 + V_2 + \ldots V_n$$
$$C_{\text{Resultant}} = C_1V_1 + C_2V_2 + \ldots C_nV_n \,/\, V_{\text{Total}}$$

where:
V = Volume (ℓ)
C = Nutrient concentration (µg·ℓ$^{-1}$)

River inflow was generally lowest during the winter months (i.e. May to July) and highest during early summer (i.e. October and November) within the Gouritz WMA. The MAR for the catchments varies greatly between the systems, with the lowest being that of the Hartenbos River (2.41 x 10^6 m^3·yr^{-1}) and the highest that of the Gouritz River (475.47 x 10^6 m^3·yr^{-1}) (Table 2). For the majority of the 17 monitoring stations (i.e. Blinde, Goukamma and Piesang excluded due to absence of flow data) there was no significant correlation ($P > 0.05$) between monthly

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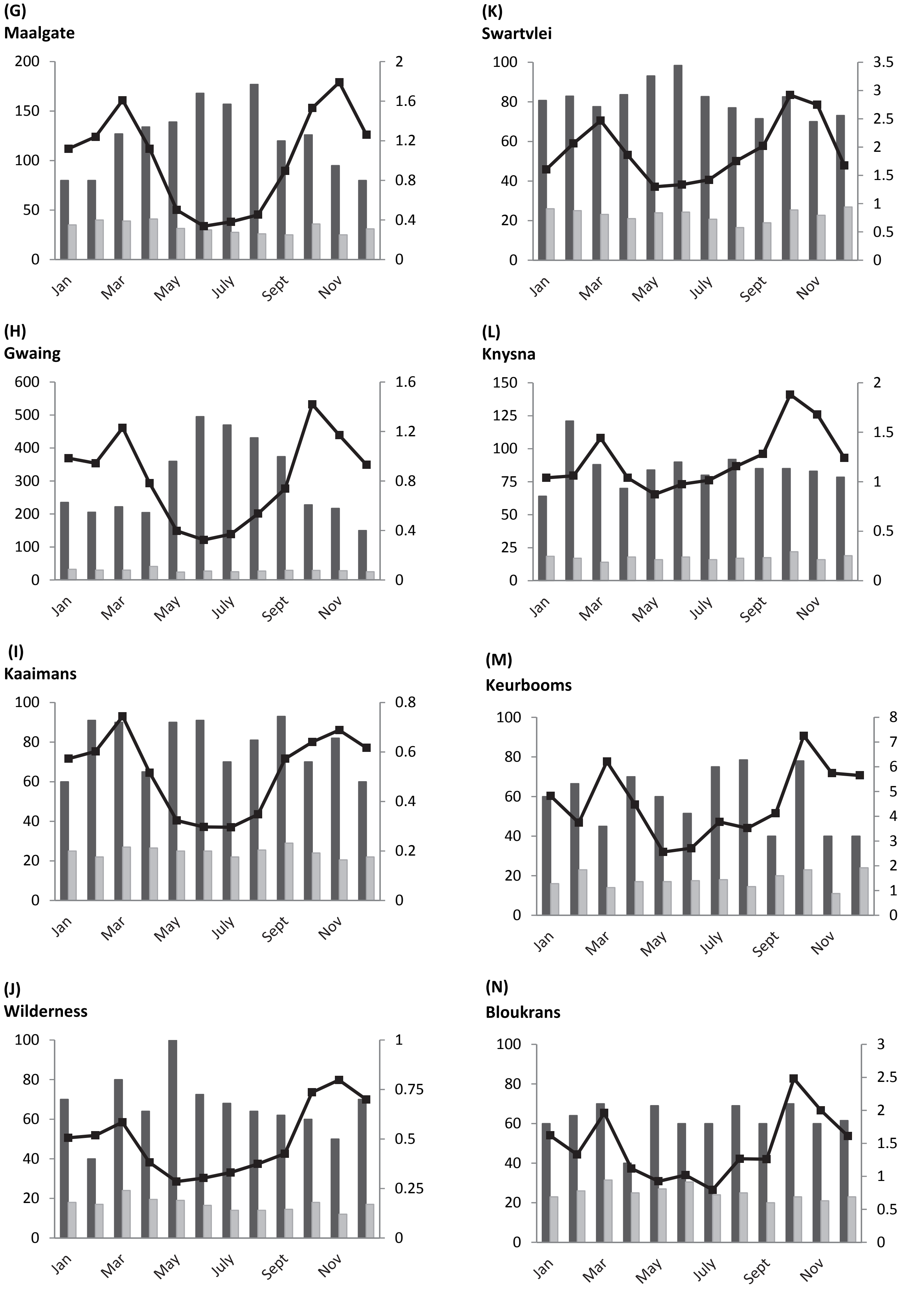

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median flows and inorganic nutrient concentrations (DIN and DIP). However, monthly median DIN values observed in the Maalgate (r^2 = 0.48) and Gwaing (r^2 = 0.66) systems were shown to significantly decrease with an increase in flow (P < 0.05). Moreover, the Great Brak River (r^2 = 0.44) was the only system in which DIP significantly increased with an increase in flow (P < 0.05).

Inter-system variation for overall DIN and DIP concentrations were investigated. The multiple rivers entering the Klein Brak and Swartvlei systems were analysed individually. It was found that the Blinde and Gwaing rivers had significantly higher DIN concentrations than the majority (> 50%) of the other systems (H = 150.04; P < 0.05; n = 234) (Table 2). The Brandwag (Klein Brak) and Diep (Swartvlei) rivers had significantly lower DIP concentrations than the majority (> 50%) of the other systems (H = 155.71; P < 0.05; n = 234).

Long-term temporal trends

A significant (P < 0.05) upward trend in DIN concentrations was observed in five of the catchments assessed: the Duiwenhoks (r^2 = 0.13; n = 471), Goukou (r^2 = 0.20; n = 324), Great Brak (r^2 = 0.11; n = 550), Gwaing (r^2 = 0.37; n = 402) and Hoekraal (Swartvlei) (r^2 = 0.41; n = 442) rivers. The Piesang river was the only catchment that demonstrated a significant (r^2 = 0.41; P < 0.05; n = 24) downward trend in DIN over the entire time series. Four of the monitoring stations studied illustrated a significant (P < 0.05) upward trend in DIP, including the: Hartenbos (r^2 = 0.11; n = 295), Hoekraal (Swartvlei) (r^2 = 0.11; n = 436), Knysna (r^2 = 0.13; n = 405) and Keurbooms (r^2 = 0.33; n = 167) rivers.

DISCUSSION

Nutrient assessment of river monitoring stations

River inflow in 50% of the systems assessed in the Gouritz WMA had median DIP values exceeding the eutrophic limits (>0.025 mg·ℓ$^{-1}$) set by the then Department of Water Affairs and Forestry (DWAF, 1996) for freshwater aquatic ecosystems. None of the systems exceeded the eutrophic limits for DIN (>2.5 mg·ℓ$^{-1}$). Perhaps more important, however, is that 45% and 100% of the systems episodically exceeded the eutrophic limits for both DIN and DIP, respectively, throughout the period of study. This is of concern as eutrophic conditions result in systems with high productivity, low species diversity, drastic shifts in community structure, bottom-water hypoxia and proliferations of nuisance and/or harmful aquatic plants (e.g. blue-green algae) (Bricker et al., 2003; Slomp, 2012).

It is important to realise that eutrophication is a highly complex and subtle issue to detect, and subsequently address. Thus, the eutrophic limits set by DWAF (1996) are only used as a preliminary approach with which to assess the status of river inflow entering the estuaries in the Gouritz WMA. The reason for this is that there are a variety of factors that control the level and extent of eutrophic symptoms, including: nutrient levels, chlorophyll *a* biomass, turbidity, residence time, tidal exchange and freshwater inflow (Bricker et al., 2003; Hilton et al., 2006). For example, with particular reference to the focus of this study (i.e. nutrients), DIP levels conducive to eutrophication are only 'problematic' provided that inorganic nitrogen and other nutrients are not limiting, and vice versa (De Villiers and Thiart, 2007). Aquatic plants require N:P ratios of between 7 and 8 (weight ratios), and associated dissolved values of > 400 μg·ℓ$^{-1}$ TN and > 30 μg·ℓ$^{-1}$ TP provide favourable conditions for eutrophication in coastal ecosystems, provided nothing else is limiting plant productivity (e.g. light levels and trace nutrient concentrations) (Swedish EPA, 2000; Camargo and Alonso, 2006; Hilton et al., 2006; De Villiers and Thiart, 2007). In freshwater systems, DIN values account for the majority of total dissolved nitrogen, and therefore the aforementioned threshold (i.e. > 400 μg·ℓ$^{-1}$ N) for eutrophication can be applied here. Conversely, DIP values only account for a fraction of total phosphorus, and therefore this study has adopted the same threshold for eutrophication, based on DIP, as proposed by De Villiers and Thiart (2007), i.e., > 20 μg·ℓ$^{-1}$ P.

Using these values as proxies for the occurrence or probability of inducing eutrophication in receiving estuarine systems, it was found that median DIN values in river inflow exceeding 400 μg·ℓ$^{-1}$ N were not observed in any of the systems (Table 2). Seasonal DIN profiles (Fig. 2) showed the Gwaing River as the only system with values exceeding 400 μg·ℓ$^{-1}$ N for short periods (i.e. June to August) during the year. However, maximum DIN values (Table 2) for each catchment indicated that values exceeding 400 μg· ℓ$^{-1}$ N were observed intermittently in all of the river systems, except for Goukamma, as it had a very short monitoring period of only 2 years. Median DIP values greater than 20 μg·ℓ$^{-1}$ P were observed in 13 of the 20 catchments assessed. Seasonal DIP profiles showed that all of the systems had values exceeding 20 μg·ℓ$^{-1}$ P during periods (i.e. ranging from 1 to 12 months per year) of the year. The results of this study indicate that the catchments in the Gouritz WMA, in terms of monthly median inorganic nutrient concentrations, are in a good to fair condition, with Gwaing being the only system consistently vulnerable to periods of eutrophication. However, the other systems in the Gouritz WMA are also prone to episodic events of eutrophication and therefore need to be managed correctly in order to prevent these events from becoming more frequent, and potentially impacting the estuaries downstream. It is important to note that studies done at the primary catchment scale provide only conservative estimates of eutrophication probability (De Villiers and Thiart, 2007). At finer scales of assessment (e.g. in vicinity of point sources) the likelihood of nutrient enrichment is expected to increase.

Wastewater treatment plants

This study identified systems subjected to direct or indirect (i.e. via river inflow) effluent discharges from WWTPs, and consequently assessed the impacts of these point sources on the respective systems. Assessing the effectiveness and compliance of WWTPs is essential to providing a complete and holistic overview of water quality inputs into the estuaries of the Gouritz WMA. Effluents from WWTPs are discharged to surface waters, after varying degrees of treatment to remove toxic contaminants and excessive nutrient loads (Withers and Jarvie, 2008). In the Gouritz WMA the size (i.e. ranging from micro to large) of the treatment plants and the method of treatment used vary depending on the size of the population that the plant serves, as well as the level of industrial activity in the area. The various methods of treatment used in the Gouritz WMA include anaerobic digestion, biological (trickling) filters, activated sludge, biological nutrient removal, composting, screw/belt press dewatering, solar/thermal drying beds, reed beds and chlorination (DWA, 2012). Despite the array of treatment methods implemented, all of the WWTPs in the Gouritz WMA exceeded the effluent chemical compliance limits for both DIN and DIP, at least episodically throughout the period of study.

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Perhaps most concerning is the Gwaing WWTP, which introduced daily loads of 267.73 kg·d^{-1} DIN and 77.46 kg·d^{-1} DIP into the surface waters of the Gwaing River (Table 3). This translates to a 95.2 and 98.2% contribution of WWTP inputs to the total daily loads of DIN and DIP entering the estuary, respectively (Table 4). Furthermore, the Gwaing WWTP is situated below the river monitoring station (Table A1, Appendix) that already exhibits potentially eutrophic conditions (i.e. due to industrial activities) with regards to its DIN and DIP levels. Therefore, it can be expected that during closed mouth phases the small Gwaing Estuary will be highly eutrophic and degraded due to these inputs.

Based on river monitoring data, it was found that the Hartenbos River was in a fairly good condition as it had low annual nutrient fluxes and low to moderate median nutrient concentrations (DWAF, 1996). However, the estuary received significant daily loads of 37.73 kg·d^{-1} DIN and 21.39 kg·d^{-1} DIP from the Hartenbos WWTP situated below the river monitoring station (i.e. within estuarine boundaries). Putting this into context, the Hartenbos WWTP is responsible for introducing 98.4 and 99.4% of the total DIN and DIP daily loads, respectively, to the Hartenbos Estuary. Therefore, due to the small size and low freshwater inflow associated with the Hartenbos Estuary, as well as closed mouth conditions, it can be expected that eutrophic conditions will occur in this system. Observations from a similar system were made along the KwaZulu-Natal south coast (South Africa) in a study by Perissinotto et al. (2002). It was shown that the temporarily open/closed Mpenjati Estuary, on the receiving-end of a WWTP discharge into the estuary, showed clear signs of eutrophication with microphytobenthic chlorophyll *a* concentrations found to be as high as 616 mg·m^{-2} chl *a*.

The Duiwenhoks system provides an example where the WWTP is situated above the river monitoring station, and therefore can be detected and monitored, to a degree, using data from the river monitoring station downstream. Results showed that the Duiwenhoks system exhibits an elevated median DIN value (158 µg·ℓ$^{-1}$) and annual N flux (17.12 kg·km^{-2}·yr^{-1}) in comparison to the other high-flow systems (i.e. monthly median flow > 2 x 10^{6} m^{3}) in the Gouritz WMA. From these findings, however, it is apparent that dilution of WWTP effluents occurs in the Duiwenhoks system (i.e. below eutrophic limits) possibly due to factors such as high flow volumes, biological uptake and adsorption (e.g. sedimentary processes). Furthermore, WWTP inputs account for only 11.98 and 73.72% of daily DIN and DIP inputs, respectively, which suggests the prevalence of other anthropogenic nutrient sources. Poor agricultural practices within the Duiwenhoks catchment, highlighted by the 2011 National Biodiversity Assessment, provide a possible explanation for these inputs (Van Niekerk and Turpie, 2012). Although observed only intermittently, these impacts are highlighted by the maximum DIP value recorded in Duiwenhoks (approx. 3 600 µg·ℓ$^{-1}$), i.e., 3 times higher than those observed in the hypernutrified Colne Estuary (approx. 1 200 µg ℓ$^{-1}$) in the United Kingdom (McMellor and Underwood, 2014).

Overall, this study found that the WWTPs in the Gouritz WMA are generally not operating to an acceptable standard and, as a result, are releasing unacceptably high levels of inorganic nutrients into the rivers along which they are situated. Similar observations were made in the Olifants River catchment during a study by Dabrowski and De Klerk (2013), thus indicating that inadequate management and maintenance of WWTPs is a national issue, and not only limited to the Gouritz WMA.

Patterns and trends

Seasonal trends relating inorganic nutrient concentrations to freshwater inflow rates provide a useful tool with which to assess the primary inorganic nutrient sources from the catchment. For example, it is known that under natural conditions rivers dominate the input of phosphorus (P) to the marine environment (Slomp, 2012; Statham, 2012); whilst the natural introduction of N to estuaries is dominated (more equally than P) by rivers, groundwater and atmospheric deposition (Jickells and Western, 2012; Voss et al., 2012). It is important to understand that the significance of these allochthonous sources, of both N and P, to a given aquatic system depends on the source composition and proximity, as well as the chemical, biological, and physical characteristics of the receiving system (Voss et al., 2012).

When examining the relationship between flow and inorganic nutrients during this study, it was shown that the vast majority of systems in the Gouritz WMA demonstrated an apparent disconnect between these two variables. This suggests that point sources determine the inorganic nutrient concentrations within the majority of the Gouritz WMA systems. As proposed by De Villiers and Thiart (2007), point sources provide a relatively constant input of DIN and DIP throughout the year, which results in seasonal concentration profiles that have no relation to flow. The most prominent activity responsible for such occurrences in the Gouritz WMA is WWTP inputs.

The Gwaing, however, exhibited an inverse relationship whereby DIN values decreased significantly with an increase in flow. This inverse relationship reflects the high volume of WWTP effluent entering this system. Similar observations were also made in a study by Scharler and Baird (2003), in which nitrate concentrations decreased with increased flow in the Sundays River. Therefore, the DIN reductions observed in this study can be attributed to the dilution effect that freshwater inflow can have on nutrient inputs from such sources in aquatic ecosystems. The Maalgate also demonstrated this inverse relationship, but in this instance it could not be linked to point-source inputs. Rather it was attributed to the effect of extended periods of low flow (monthly median flow <1 x 10^{6} m^{3}) that facilitate the accumulation of DIN via biological processes (e.g. nitrification and ammonification) during these periods (McMellor and Underwood, 2014). As a result, when these systems experience high flow periods, elevated nutrient levels are diluted by the introduction of high-velocity (in relation to base flow conditions) freshwater inputs.

The Great Brak was the only system that showed DIP concentration profiles coincident with river runoff, where DIP levels increased with an increase in flow. This observation could indicate that the primary source of DIP to this system is of riverine origin, or alternatively it could suggest the facilitation of diffuse pollution (e.g. fertilisers) via increased runoff (Hilton et al., 2006; De Villiers and Thiart, 2007). As a result, prominent land-use practices such as irrigated agriculture, afforestation and urbanisation in this catchment could potentially be responsible for producing the strong seasonal DIP concentration profiles observed (River Health Programme, 2007). Perhaps more likely is the influence of the upstream dam (i.e. Wolwedans Dam), which may be acting as a sink that 'feeds' the estuary with DIP during high flow events (i.e. overflow from dam) (Camargo et al., 2005).

With regards to the annual nutrient fluxes calculated for the systems in the Gouritz WMA, it is important to realise that these values are reliant, to a degree, on the MAR and area of each catchment. However, the annual nutrient flux of a catchment gives a good indication as to land use in the area (Hilton et al., 2006). For example, De Villiers and Thiart (2007) showed that catchments with greater than 20% agricultural land-use all exhibited high N and P fluxes, of > 8 $kg·km^{-2}·yr^{-1}$ N and > 0.47 $kg·km^{-2}·yr^{-1}$ P, respectively. When comparing the findings of this study, it was found that approximately 70% and 90% of the catchments followed the same trends for annual inorganic N and P fluxes, respectively. This ultimately suggests that the land-use practices previously identified by the River Health Programme (2007), such as agricultural activities (e.g. irrigation and livestock), urbanisation, industrialisation and afforestation, are the primary source of nutrients from the catchments in the Gouritz WMA. This is, however, a conservative comparison due to the generally reduced catchment area, annual runoff and land-use extent of the catchments assessed in this study.

The elevated inorganic nutrient fluxes observed for the majority of the catchments demonstrates the extreme pressures placed on small systems, especially temporarily open/closed estuaries, in South Africa. Further evidence for this was illustrated in this study, whereby 5 of the catchments demonstrated significant increases in DIN over the monitoring period, and a further 4 catchments showed similar increasing patterns for DIP in river waters. These increasing trends are a cause for concern as eutrophication resulting from increased nutrient loading is regarded as one of the critical issues facing water quality management in South Africa (Dabrowski and De Klerk, 2013). More specifically, because the Gouritz WMA is dominated by smaller, low-flow systems that are more vulnerable to degradation by anthropogenic activities, the potential for drastic environmental changes and socio-economic losses are significant. Therefore, the identification of nutrient sources is an important step with regard to managing eutrophication at the catchment level (Dabrowski and De Klerk, 2013).

CONCLUSION

This study highlighted the benefit of long-term water quality monitoring datasets to estimate inputs to coastal ecosystems, as these are needed to assess trends, identify management priorities, as well as identify shortcomings associated with such assessments. The river systems in the Gouritz WMA were generally shown to be in fairly good condition; however, the risk of introducing nutrients to receiving coastal ecosystems is ever-present in the majority of the catchments. Furthermore, the general non-compliance with targets/limits for WWTPs and freshwater quality guidelines emphasizes the potential threats to the natural ecological functioning and sustainability of downstream estuarine systems in the Gouritz WMA. Effective management and maintenance of WWTPs is needed in order to reduce the high inorganic nutrient loading in catchments.

One way in which this problem has been addressed is through the implementation of tertiary treatments (‘P stripping’), in situations where the receiving waters are highly sensitive to eutrophication (Withers and Jarvie, 2008; McMellor and Underwood, 2014). Although an effective management tool in freshwater systems, reducing only P loads can result in exacerbated eutrophication symptoms (i.e. adsorbed P released under saline conditions) downstream (Conley et al., 2009). Consequently, it is sensible to implement a ‘dual-nutrient-reduction’ strategy to prevent eutrophication in estuaries (Conley et al., 2009). Wetlands, natural or constructed, can also be used to remove nutrients associated with diffuse and point sources (De Villiers and Thiart, 2007). It has been shown that the nutrient retention of wetlands is more than efficient (i.e. 3 000 to 285 000 $kg·km^{-2}·yr^{-1}$ N and 100 to 71 000 $kg·km^{-2}·yr^{-1}$ P) with regards to reducing nutrient inputs to aquatic ecosystems (Fink and Mitsch, 2004; De Villiers and Thiart, 2007). This highlights the importance of wetlands, and illustrates a technique that could be implemented more consistently in South Africa in order to improve the overall status of its aquatic ecosystems.

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APPENDIX

TABLE A1
Supplementary river sampling station information

River catchment	Sampling period	Sampling point	Distance from estuary (km)	Latitude (°S)	Longitude (°E)
Duiwenhoks	1967 – 2012	Dassjes Klip	0	34.25167	20.99194
Goukou	1969 – 2012	Farm 216	24	34.09250	21.29417
Gouritz	1965 – 2012	Zeekoedrift	48.9	33.98028	21.65333
Blinde	2009 – 2011	Farm 316 - PetroSA	6.1	34.18038	21.9838
Hartenbos	1973 – 2012	Hartebeestkuil	10.3	34.09694	22.01028
Klein Brak (Brandwag)	1973 – 2012	Brandwacht	7.8	34.03167	22.05278
Klein Brak (Moordkuil)	1976 – 2012	Banff	0.7 km within estuary boundaries	34.03944	22.13222
Great Brak	1970 – 2012	Wolvedans	0.4 km within estuary boundaries	34.02861	22.22194
Maalgate	1971 – 2012	Knoetze Kama	4.7	34.00667	22.35028
Gwaing	1967 – 2012	Blanco	15.2	33.95111	22.42250
Kaaimans	1971 – 2012	Upper Barabierskraal	2.9	33.97111	22.54750
Wilderness	1969 – 2012	Farm 162 (Touw River)	4.9	33.94528	22.61444
Swartvlei (Hoekraal)	1969 – 2012	Eastbrook	1.7 km within estuary boundaries	33.97982	22.79950
Swartvlei (Karatara)	1971 – 2012	Karatara Forest Res.	13.9	33.88083	22.83778
Swartvlei (Diep)	1971 – 2012	Woodville Forest Res.	9.3	33.91250	22.70778
Goukamma	1998 – 1999	Buffels Vermaak	4 km within estuary boundaries	34.0325	22.93972
Knysna	1971 – 2012	Milwood Forest Res.	17.3	33.89111	23.02944
Piesang	1996 – 2008	Piesang River	0	34.06167	23.35639
Keurbooms	1967 – 2008	Newlands (Flow ~ K6H19)	9.8	33.93844	23.36730
Bloukrans	1967 – 2012	Lotterings Forest Res.	2.8	33.95556	23.63861

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